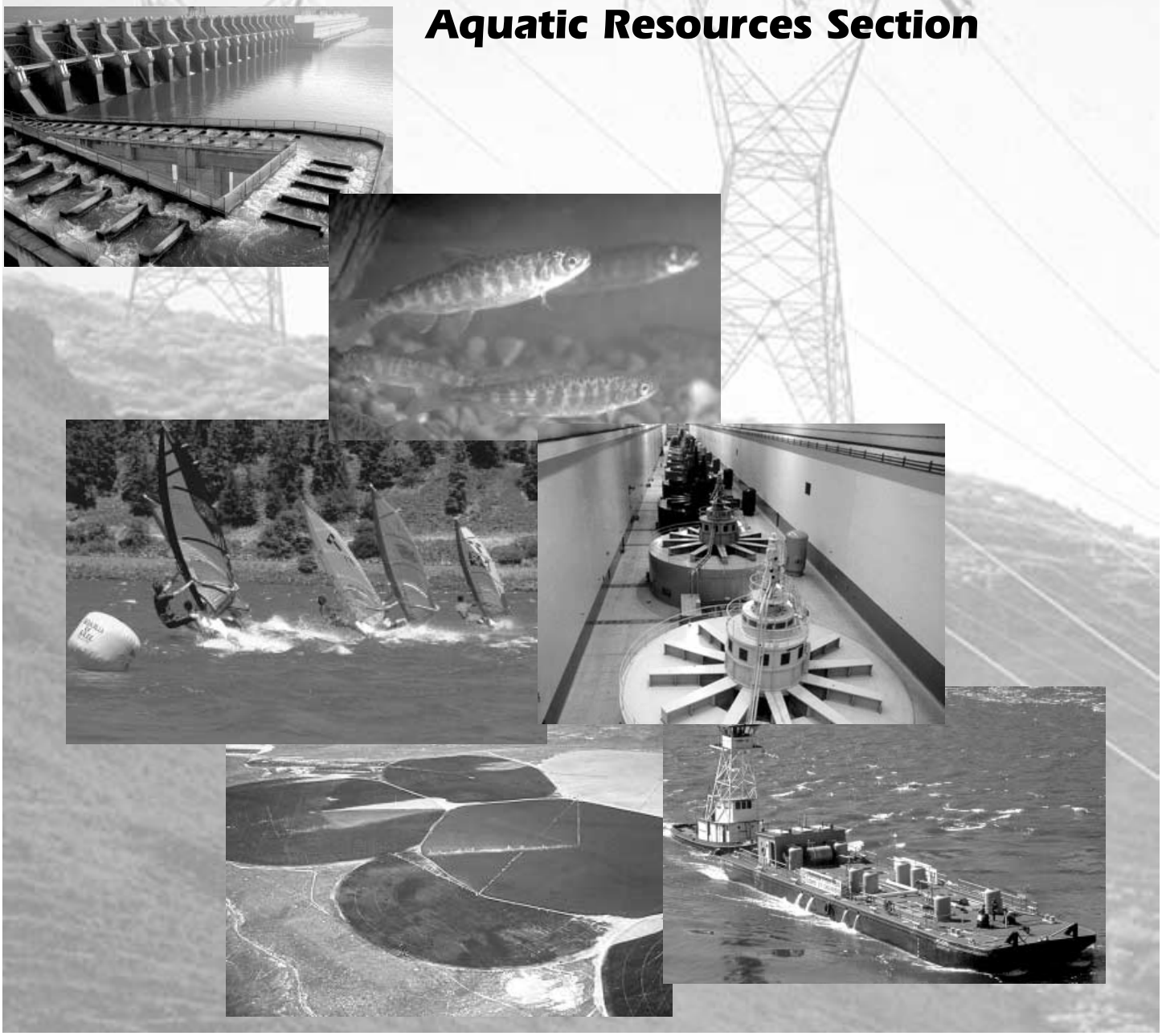
US Army Corps of Engineers ®
Portland District

Salmon Recovery through John Day Reservoir

John Day Drawdown Phase I Study

Biological/Environmental Technical Appendix
Aquatic Resources Section



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SECTION 1. Introduction

This technical appendix section documents the results of the aquatic resources evaluation for the John Day Drawdown Phase I Study. This Phase I Study is a reconnaissance-level evaluation of the potential consequences and benefits of the proposed drawdown of the John Day Reservoir. This technical appendix section supplements the main report, which describes more fully the alternatives, purpose, scope, objectives, assumptions, and constraints of the study.

SECTION 2. Background of the Project

In 1991, the National Marine Fisheries Service (NMFS) proposed that Snake River wild sockeye, spring/summer chinook, and fall chinook salmon be granted "endangered" or "threatened" status under provisions of the Endangered Species Act. Natural resource agencies believe that the drawdown of the 76-mile John Day Reservoir may provide substantial improvements in migration and rearing conditions for juveniles by increasing river velocity, reducing water temperature and dissolved gas, and restoring riverine habitat. It is also speculated that drawdown may improve spawning conditions for adult fall chinook by restoring spawning habitat and the natural flow regimes needed for successful incubation and emergence.

As a result, the NMFS Reasonable and Prudent Alternative Action #5 of its' Biological Opinion on Operation of the Federal Columbia River Power System (FCRPS), and subsequent reports recommended that USACE investigate the feasibility of lowering John Day Reservoir. In compliance with appropriation conditions, only two alternatives were to be evaluated: reduction of the current water surface elevation 265 to the level of the spillway crest that would vary between elevations 217 and 230, or reduction to natural river level elevation 165. Both alternatives were proposed by NMFS. These two alternatives were then expanded to consider each alternative with 500,000 acre-feet of flood storage and without such storage. Flood storage and hydropower are the current approved authorizations for the John Day project.

SECTION 3. Description of the Study Area

The Columbia River originates in Canada and flows for 300 miles through eastern Washington to Oregon and continues west to the Pacific Ocean, as shown in Figure 1. The adjoining region is mostly open country, with widely scattered population centers. The climate of the region is semiarid. Agriculture, open space, and large farms are prevalent. Lands adjacent to the reservoir are used to grow grains and other crops. The reach of the Columbia River under consideration in this report extends from John Day Lock and Dam at river mile (RM) 215.6, to McNary Lock and Dam RM 291. The body of water impounded by John Day Dam, Lake Umatilla, is referred to as the John Day Reservoir throughout this report. The John Day is the second longest reservoir on the Columbia River, extending 76 miles upstream to McNary Dam.

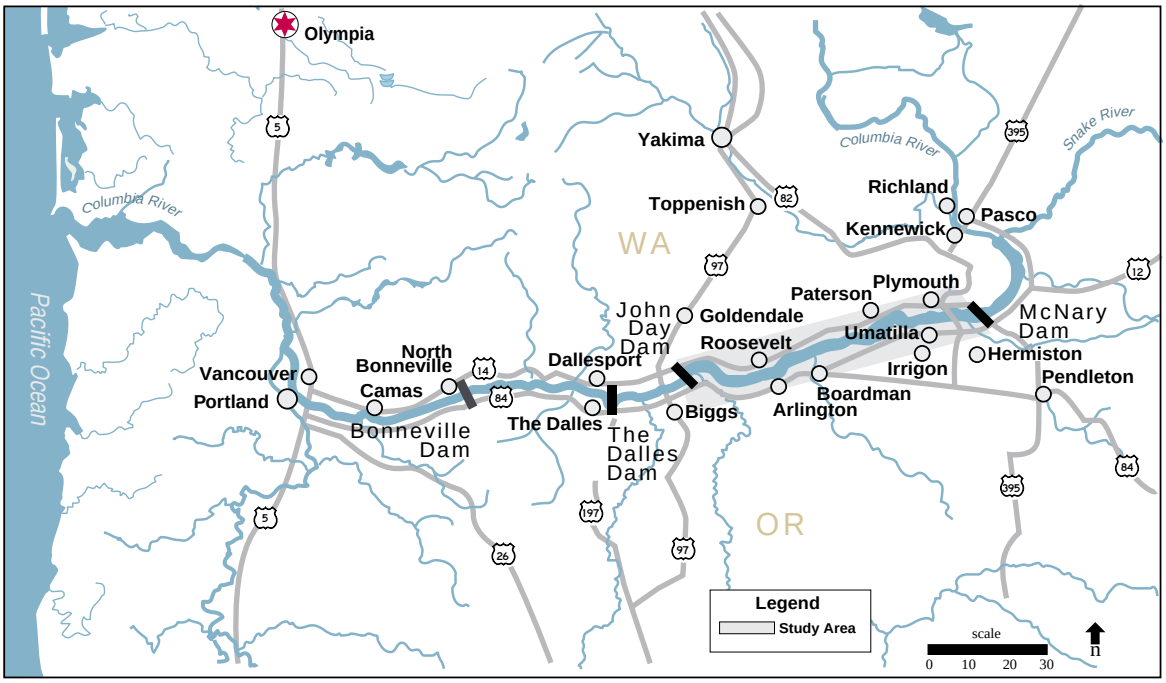


Figure 1. John Day Drawdown Phase 1 Study Area

John Day Dam and Reservoir are part of the Columbia-Snake Inland Waterway. This shallow-draft navigation channel extends 465 miles from the Pacific Ocean at the mouth of the Columbia River to Lewiston, Idaho. The entire channel consists of three segments. The first is the 40-foot-deep water channel for ocean-going vessels that extends for 106 miles from the ocean to Vancouver, Washington. The second is a shallow-draft barge channel that extends from Vancouver to The Dalles, Oregon. Although this section is authorized for dredging to a depth of 27 feet, it is currently maintained at 17 feet. The third section of the channel is authorized and maintained at a depth of 14 feet and extends from The Dalles to Lewiston. In addition to the main navigation channel, channels are dredged to numerous ports and harbors along the river.

The middle Columbia River area is served by a well-developed regional transportation system consisting of highways, railroads, and navigation channels. Railroads and highways parallel the northern and southern shores of the reservoir. Interstate 84 (I-84), a divided multilane highway, runs parallel on the south shore with the Columbia River from Portland, Oregon, to points east. Washington State Route 14 (SR-14) also parallels the Columbia River from Vancouver to McNary Dam on the north shore. Umatilla Bridge at RM 290.5, downstream from McNary Dam, is the only highway bridge linking Oregon and Washington across the Columbia River in the John Day Reservoir.

The study area includes lands directly adjacent to the reservoir as well as those directly and indirectly influenced by the hydrology of the reservoir (e.g., irrigated lands). It includes the reservoir behind the John Day Dam, and adjoining backwaters, embayments, pools, and rivers.

SECTION 4. Alternatives

The Phase 1 Study includes a preliminary evaluation of the impacts of the drawdown scenarios relative to the "without project condition," which is defined as the condition that would prevail into the future in the absence of any new federal action at John Day. The four alternatives are summarized below. One of the most important constraints on the alternatives is the requirement to pass fish for river flows up to the 10-year flood flow of 515,000 cfs. Under the four alternatives, John Day Reservoir would be drawn down at a rate of one foot per day. For greater detail, please refer to the main report, *John Day Drawdown Phase 1 Study*, and *John Day Drawdown Phase 1 Study, Engineering Technical Appendix, Structural Alternatives Section*.

4.1 Spillway Drawdown without Flood Control (Alternative 1)

The first drawdown alternative is based on requirements for improved downstream fish passage conditions during both low and flood flow conditions on the Columbia River. The existing 20-bay spillway will be operated differently from current operations, but without any structural modifications. All project inflows will be directly passed through the dam spillway with the spillway gates fully opened in free overflow condition, resulting in a pool elevation that will vary from elevation 217 to 230. Impacts downstream from John Day Dam were not studied.

4.2 Spillway Drawdown with Flood Control (Alternative 2)

The second study alternative is based on requirements for improved downstream fish passage conditions during low flow periods, while maintaining authorized flood control for the John Day Project. The existing 20-bay spillway will be operated differently from current operations, but without any structural modifications. During low flow periods, project inflows will be directly passed through the dam spillway with the spillway gates set in fully open, free overflow condition. During a flood event, however, the spillway gates will be controlled to reduce downstream flood flows based on using 500,000 acre-feet of allocated project storage space. Ponding will occur upstream from the dam. Impacts downstream from John Day Dam were not studied.

4.3 Natural River Drawdown without Flood Control (Alternative 3)

The third study alternative is based on a natural river drawdown for fish passage "without flood control" condition. Natural river conditions pertain to an opening at the John Day Dam that permits acceptable upstream fish passage conditions. The size of the total dam opening must conform to two criteria based on an invert elevation at the dam of 135. The first criterion is that the opening must be sufficiently large to meet maximum allowable stream velocity criteria for sustained swim speed for the weakest salmon species, which is estimated to be 10 feet per second (fps). The second criterion is that fish passage for this opening must correspond to the 10-year annual flood peak (515,000 cfs). This alternative will require extensive modifications to John Day Dam even beyond modification of the 1,228-foot long spillway structure. Impacts downstream from John Day Dam were not studied.

4.4 Natural River Drawdown with Flood Control (Alternative 4)

This fourth study alternative is based on natural river conditions for fish passage and includes the "with flood control" condition. It requires natural fish passage conditions for both upstream and downstream directions at the dam and includes a requirement for full authorized flood control. The calculated width of the total dam opening will correspond to that previously calculated for natural river conditions without flood control (Alternative 3). Impacts downstream from John Day Dam were not studied.

SECTION 5. Fisheries Analysis Description

This assessment was undertaken based on the NMFS Reasonable and Prudent Alternative Action #5 of its 1995 Biological Opinion (BiOp), for the operation of the Federal Columbia River Power System. This action was recommended based on the status of ESA[1]-listed anadromous fish species in the Snake River Basin. It also serves as a preliminary action to a feasibility study of potential long-term alternative configurations of lower Columbia River projects to improve survival of ESA-listed species, as proposed in the 1998 Steelhead BiOp prepared by NMFS. Benefits, if realized, could be extended to the mid- and upper Columbia River stocks as well.

[1] Endangered Species Act

The premise of evaluating drawdown as a stock recovery tool is based on the assumption that drawdown will: 1) improve juvenile salmonid in-river survival by decreasing their travel time past the John Day project; 2) increase the quality and quantity of in-river spawning habitat; 3) create a more natural shoreline and benthic community structure, thereby improving fish rearing habitat for native fishes; and 4) for natural river drawdown, will eliminate mortality associated with dam passage. The drawdown alternatives could improve juvenile migration and rearing by: 1) increasing river velocities through the John Day Reservoir Reach, 2) narrowing the period of high water temperature regimes, and 3) decreasing total dissolved gases. Drawdown could also improve spawning conditions, for primarily fall chinook salmon and white sturgeon, by restoring spawning habitat and natural flow regimes for successful embryo incubation. Migration for both juvenile and adult fish would be improved by removal of the John Day Dam as an obstacle to passage.

In addition to decreasing travel times and increasing spawning habitat, drawdown is anticipated to provide other benefits to anadromous salmonid fishes. Recent discussions concerning changes in hydro system operations have focused on restoration of salmonid populations through ecosystem-based approaches such as the "Normative River." The Independent Scientific Group (IFG) put forth this concept, calling for re-establishment of a riverine character in key stream reaches of the Columbia River Basin. For example, drawdown may lead to reservoir conditions similar to the unimpounded Hanford Reach of the Columbia River that supports a healthy and productive mainstem fall chinook salmon stock. Under drawdown scenarios at John Day, it is presumed that the composition of resident fish species would more closely resemble pre-impoundment fish assemblages.

The benefits to be gained, however, may be offset by the effects of other modifications in the system. Biological, chemical, and physical alterations including reservoir habitat changes, changes in predation rates or periods of predator/prey overlap, lower juvenile passage survival through inefficient turbine operation, delayed downstream mortalities and the like, require a complete evaluation to judge the response to the four alternatives. The ultimate prediction of benefit is the anticipated change in the number of returning adults and the contribution toward NMFS recovery goals for these stocks. As such, a full life-cycle assessment is summarized or assumed for each stock in a subsequent section of this report. The intent of the life-cycle modeling is to assess if the potential benefits of drawdown to juvenile outmigrants will result in potentially greater numbers of adult fish to the natal streams.

This appendix addresses the influences of drawdown on:

- Anadromous salmonid fishes
  - Juvenile rearing, migration and passage
  - Adult spawning
- Resident fish communities
- Site-specific habitat changes

With respect to anadromous salmonids, ESA-listed species and specific stocks influenced by drawdown at John Day Dam were evaluated. The assessment occurred on a life-history stage basis. Differences in life-history characteristics between the stocks were summarized in

relation to drawdown effects. The result for each stock was synthesized by a life-cycle assessment into the total anticipated change in returning adult escapements with the drawdown alternatives.

Juvenile anadromous salmonid fish rearing, migration and dam passage were evaluated, and the results are presented in 0. This section also incorporates the life-cycle modeling assessment and estimates the number of returning adults with each drawdown alternative. Adult spawning changes with drawdown are provided in SECTION 7. Habitat modifications and resident fish community alterations are estimated in Section SECTION 8. Resident fish communities are evaluated in light of anticipated changes in primary and secondary productivity in the reservoir, and alterations to benthic and pelagic community structures. Estimated changes in salmonid predator abundance, population dynamics and shifts in locations and spatial and temporal overlap with juvenile salmonids were evaluated with anticipated changes in reservoir habitat conditions.

5.1 Listed Species of Concern

Salmonid fish species in the Columbia River system listed under the ESA that would potentially benefit from drawdown activities at John Day Dam are provided in Table 1, below. Other ESA-listed salmonid species in the region originating from areas located downstream of John Day Dam are unlikely to directly benefit from drawdown at John Day and will not be evaluated in this preliminary assessment. Thus, the evaluation species and specific stocks considered herein include up-river components of Snake and Columbia River chinook and sockeye salmon, steelhead and bull trout, as listed in Table 1.

Table 1.
ESA-listed Salmonid Fish Species Potentially Influenced with Drawdown at the John Day Project; Including Status and Date of Listing

Species	ESU[1]	Stock/Race	Listing Status	Date
Chinook Salmon *(Oncorhynchus tshawytscha)*				
	Snake River	Spring/Summer	Threatened	Apr 92
	Snake River	Fall	Threatened	Apr 92
	Upper Columbia	Spring	Endangered	Mar 99
Steelhead (*O. gairdneri*)				
	Snake River	Summer	Threatened	Aug 97
	Upper Columbia	Summer	Endangered	Aug 97
	Middle Columbia	Summer	Threatened	Mar 99
Sockeye Salmon (*O. nerka*)				
	Snake River		Endangered	Nov 91
Bull Trout (*Salvelinus confluentus*)				
	Columbia River (DPS)[2]		Threatened	June 98

[1]ESU = Evolutionary Significant Unit
[2]DPS = Distinct Population Segment

5.2 Salmonid Life-Histories

Salmonid fishes have adapted life-history strategies and occupied specific habitat features to maximize their production while minimizing competition with other species for resources (niche and guild theory; adaptive management). Life-history adaptations include variations in adult and juvenile migration timing, spawning segregation (temporal and spatial), habitat preferences, prey selection, lengths of freshwater or ocean residence, and/or predator avoidance mechanisms, to name a few. Understanding the differences between species behavior is critical to assessing the potential benefits or impacts of the proposed actions of the various stocks in question. The following subsection highlights life histories of the species with particular emphasis on the strategies where they differ in relation to the proposed action.

5.2.1 Chinook Salmon (Oncorhynchus tshawytscha)

Chinook salmon are the largest of all Pacific salmon, and can weigh over 100 pounds; the average weight is close to 22 pounds. Chinook salmon are an anadromous fish, spending part of their life history in freshwater and part in saltwater. Adults spend one to several years in the ocean where they grow and mature before returning to their natal stream to spawn. The eggs are deposited into a nest of clean gravel, referred to as a "redd." Adult salmon die after spawning.

The eggs remain buried in the gravel for one to four months, depending on stream temperatures. Chinook eggs require 882 to 991 temperature units on average before hatching (1 temperature unit = 1 degree C above freezing for 24 h) (Beauchamp et. al., 1983). The alevins, or yolk-sac fry, remain in the gravel for two to three weeks after hatching (Wydoski and Whitney, 1979). After emerging from the gravel, the young salmon can remain in the natal stream for days, months, or several years depending on the race, and stock before migrating down stream to the ocean.

Historically, chinook salmon entered the Columbia River continually from early spring through late fall. Due to overharvest and the construction of dams without fish passage, segments of the run were eliminated (Chapman, 1994a, 1995a). Timing of peak counts of adults passing upstream of dams and segregation of spawning areas are used today to divide the continuum into separate stocks. A window of time for egg deposition exists in each spawning area based on water temperature, and the timing of upstream migrating adults matches this window (Miller and Brannon, 1982). Therefore, adults that spawn in the upper reaches of tributaries, in the middle and lower reaches of tributaries, and in the mainstem rivers and lower reaches of tributaries, can be divided into three races.

Adults of the race that spawn in the highest reaches of the tributaries enter the Columbia River first and migrate past mainstem dams in spring and are known as spring chinook. Similarly, the race that spawns in the middle and lower reaches of tributaries generally return past mainstem dams in the summer, and are known as summer chinook. Those spawning in lower tributaries and mainstem rivers arrive in the fall, are known as fall chinook salmon (Meekin, 1963; French and Wahle, 1965; Chapman et. al., 1982; Mullan, 1987).

This classification is arbitrary. For example, the administrative cutoff date for separating spring and summer chinook is June 10th and for separating summer and fall chinook is

August 1st at the John Day Dam. These cutoffs are not necessarily reflective of the origin of the adults (Chapman et. al., 1995a). Spring-run chinook pass the John Day dam from late April though early June. Spawning begins during late July and continues through September. Peak spawning varies among tributaries (Chapman et. al., 1995a; Peven, 1992). Summer-run chinook return to freshwater in June through mid-August, with peak spawning occurring in October (Peven, 1992). Columbia River fall chinook runs typically return to the river in mid-August through October, and spawn within a few weeks. Peak spawning occurs in November (Myers, 1998, from WDF et. al., 1993, Kostow, 1995).

Within the three races, chinook salmon have been classified into two life-history strategies; "ocean-type" populations that migrate to seawater as fry in their first year of life (subyearlings) and spend most of their oceanic life in coastal waters, and "stream-type" populations that migrate to sea as yearlings and often make extensive oceanic migrations (Myers et al., 1998). Since the maturation process is slower in the cold headwater tributaries, most spring chinook juveniles do not reach sufficient size and developmental maturity to migrate during the first year. These fish winter-over in the freshwater environments and migrate seaward during the second spring, in the late March through May time period. They are referred to as "stream-type" fish. Conversely, summer and fall chinook rear in warmer waters and primarily outmigrate during their first year, mostly late May through July. They are referred to as "ocean-type" fish.

Within the overall dominant strategy for each race of chinook salmon, there are many possible variations in juvenile life history. A small component of each population will vary in their outmigration style. The distinctions result from variability in the pattern of juvenile freshwater rearing (Reimers, 1973). Environmental cues such as streamflow reductions, food supply, changes in photo-period, and temperature increases are all factors that lead to the evolution and expression of particular juvenile outmigration timing (Myers et al., 1998).

Five examples of juvenile chinook salmon life histories suggested by Reimers (1973) are as follows:

- Emergent fry move directly downstream and into the ocean within a few weeks
- Juveniles rear in the main river or remain in tributaries until early summer, then emigrate into the estuary for a short period of rearing and enter the ocean before the improved growth in late summer
- Juveniles rear in the main river or tributaries until early summer, then emigrate into the estuary for extended rearing during the period of improved growth in late summer and enter the ocean in autumn
- Juveniles rear in the tributary streams (or, rarely, in the main river) until autumn rains, then emigrate to the ocean
- Juveniles remain in the tributary streams (or rarely in the main river) through the summer, rear in the river until the following spring, and enter the ocean as yearlings

Juvenile chinook may pass through estuaries quickly or may reside in the estuary for up to 189 days (Emmett et al., 1991). Chinook typically spend three to four years in the ocean before returning to their natal stream to spawn.

5.2.1.1 Known Occurrences in the Project Vicinity.

NMFS has identified 15 evolutionarily significant units (ESU) of chinook salmon in the western United States. The ESU designations were based on a combination of genetic data, life-history behavior, and geographic isolation that support the likelihood of reproductive isolation between populations. Seven chinook ESUs were identified in the Columbia River basin, five of which may use the John Day Reach during some part of their life history, as described below.

- **Mid-Columbia River Spring-Run ESU.** The Mid-Columbia River Spring-Run ESU includes natural spring-run chinook salmon that exhibit a stream-type life history strategy. They spawn in Columbia River tributaries from the Klickitat River Basin upstream to and including the Deschutes, John Day, and Yakima River Basins, excluding the Snake River Basin. The chinook salmon from this ESU generally migrate to the ocean as yearlings. The dominate age-class of adults returning to spawn is age 4 in most streams except for populations from the upper tributaries of the Yakima that primarily return at age five. Chinook salmon from this ESU do not appear in appreciable numbers in any ocean fisheries and are assumed to migrate far offshore.
- **Upper Columbia River Summer-and Fall Run ESU.** The Upper-Columbia River Summer and Fall Run ESU is comprised of fall- and summer-run "ocean-type" chinook salmon in the Columbia River and tributaries upstream of the confluence of the Snake River to Chief Joseph Dam, excluding Marian Drain, an irrigation collection canal to the Yakima River. The chinook salmon in this ESU mature at an older age than the ocean type chinook salmon in the Lower Columbia and Snake Rivers. This ESU includes the fish that managers have termed upriver bright (URB) fall chinook from the Hanford Reach of the Columbia. The Hanford Reach of the Columbia River is the only unimpounded section of the Columbia River remaining in this ESU.
- **Upper Columbia River Spring Run ESU (Endangered)**. The Upper Columbia River Spring Run ESU is comprised of spring-run, stream type chinook salmon in the Columbia River upstream of the Yakima River to Chief Joseph Dam including its tributaries. These fish spawn above Rock Island Dam in the Wenatchee, Entiat, and Methow river systems (Chapman et al., 1995a; Mullan, 1987). They predominately migrate to the ocean as yearling smolts, and are rarely recovered from coastal fisheries.
- **Snake River Fall-Run ESU (Threatened)**. The Snake River Fall-Run ESU is comprised of "native" populations of fall-run, ocean type chinook salmon. Hatchery origin fish in the ESU are excluded from the listing status. Per final NMFS decision (Fall, 1999), this ESU includes the entire Snake River Basin upstream of its confluence with the Columbia River. The fall run is primarily composed of "ocean-type" chinook salmon
- **Snake River Spring-and Summer-Run ESU (Threatened).** The Snake River Spring- and Summer-Run ESU is comprised of all natural populations of spring- and summer-run stream-type chinook salmon in the Snake River and its tributaries, excluding the Clearwater River. These fish migrate to the ocean as yearling smolts, are rarely recovered from coastal fisheries. They typically return to the ESU to spawn at age four or five years.

5.2.1.2 Chinook Population Status under the ESA.

NMFS listed the Snake River Spring/Summer and Fall-Run ESUs as threatened in April, 1992 and the Upper Columbia River Spring-Run ESU of chinook salmon in Washington as endangered under the ESA on March 24, 1999 (64 FR 14308). The Upper Columbia River Summer/Fall Run ESU population is healthy and quite robust. Numbers of returning adults have been increasing dramatically since the, 1980s in spite of commercial harvest and hydroelectric influences and the greater summer overlap with predators than other salmonid races/species.

5.2.2 Sockeye Salmon (*O. nerka*)

Anadromous sockeye salmon exhibit a variety of life history patterns. They use lakes for juvenile rearing more than other species of salmon. There are four major sockeye life history patterns: lake-type, river-type, sea-type, and land-locked.

Lake-type represent the most common and typical life history form or pattern, they spawn in either inlet or outlet streams of lakes, or along lakeshores. The juveniles typically spend one to three years in lake habitat before migrating to the ocean. Sockeye that rear in rivers for one to two years (**river-type** sockeye) are less common than the lake-dependent sockeye life history forms, and hence, little is known about them. River type sockeye that migrate as fry to saltwater, or lower river estuaries in the same year as emergence are termed "**sea-type**" sockeye (Gustafson et al., 1997). Anadromous sockeye, regardless of the life-history type, spend one to four years in the ocean before returning to their natal lake or river system to spawn. The **landlocked** or resident form, called kokanee, spend their entire life in fresh water, and do not migrate to the ocean (Burgner, 1991).

Wydoski and Whitney (1979) report adult sockeye reaching lengths of 33 inches and a weight averaging between 3.5 and 8 pounds. They are very accurate in their return to their natal stream, which is believed to help ensure that proper rearing habitats are available for juvenile sockeye. Many adult sockeye make long migrations, requiring higher stored energy reserves. Delays in migration can be very damaging to the success of spawning (Hart, 1988).

Sockeye spawn on lake shores at outwash fans of tributaries, along beaches where the substrate is free of fine sediment and the eggs are oxygenated by wind-driven water circulation, or in streams with suitable spawning habitat. Sockeye typically select spawning areas that contain upwelling of oxygenated water through sand and gravel substrate. Spawning occurs from August to December, usually peaking in October. Spawning typically occurs in water temperatures ranging from 3 to 7ºC (Emmett et al., 1991). For a given fish size, sockeye salmon have the highest number of eggs, and the smallest egg size of the Pacific salmon (Gustafson et al., 1997).

In general, spawning occurs during periods of declining temperatures, incubation occurs throughout the cold winter temperatures, and hatching is associated with rising water temperatures (Burgner, 1991). The length of sockeye egg incubation is temperature dependent, and is generally longer than the other salmon species (Burgner, 1991), taking from approximately 50 days to more than five months depending on temperature (Hart, 1973, Scott and Crossman, 1973). Reiser and Bjornn, 1979 suggest optimum incubation temperatures lie between 4 and 14°C.

After emergence (January to June), juvenile sockeye will migrate to rearing lakes, use river and estuarine habitat for rearing, or migrate directly to the sea (Birtwell et al., 1987; Wood et al., 1987; Heifitz et al., 1989; Murphy et al.1991; Eiler et al., 1992; Levings et al., 1995; and Wood, 1995). Initially upon emergence, juvenile sockeye exhibit photonegative response, traveling at night. This behavior is believed to be an anti-predator adaptation (Burgner, 1991). Smolt outmigration to the ocean also occurs during darkness, beginning in March and extending through early July, when temperatures rise to 4 to 7°C (Emmett et al., 1991).

5.2.2.1 Known Occurrences in the Project Vicinity.

Sockeye adults enter the Columbia River in May and occur at John Day dam through mid-August. Adults typically reach the spawning grounds in July through September and spawn September and October (Mullan, 1986). Sockeye juveniles migrate as smolts after spending one to three years in nursery lakes, passing the lower Columbia River dams between mid-April through late May, (Bjornn et al., 1968; Chapman et al., 1995b). The mean size of wild sockeye salmon outmigrating past the dams has been reported to range from about 95 to 154 mm (Peven, 1992). Sockeye from the following three ESUs may use the John Day Reach during some part of their life history:

- **Snake River ESU (Endangered).** The Snake River ESU comprises sockeye salmon originating solely from the Snake River basin. Currently, only a remnant population from Redfish Lake occurs. Since their listing in, 1991, there have been several years without adult returns to Redfish Lake. As modification, a captive brood stock program was developed that is apparently having some success. Approximately 22 sockeye were counted during the, 1999 summer migration past Ice Harbor dam, headed for Redfish Lake.
- The Snake River basin lies on the extreme southern edge of the historic sockeye distribution. Populations on the fringe of their historic range and sockeye in particular have a wide variability in returning numbers of spawners.
- **Okanogan River ESU.** The Okanogan River ESU is comprised of sockeye salmon from Lake Osoyoos. The Okanogan River sockeye salmon typically spawn in the Okanogan River above Lake Osoyoos in Canada. Sockeye are occasionally observed spawning downstream of Lake Osoyoos, and below the Enloe Dam on the Similkameen River. These fish are believed to be strays from the Okanogan River population and are therefore included in this ESU. Resident sockeye, kokanee salmon, (*O. nerka*) observed spawning with the anadromous sockeye on the Okanogan River are considered part of this ESU.

The Okanogan River ESU has been distinguished from other ESUs by genetic, environmental, and life history information. There are differences in both the juvenile migration and adult return timing between Okanogan and Wenatchee sockeye populations. Okanogan juveniles typically migrate one month earlier than Wenatchee-run fish. Okanogan River adult sockeye typically return to the Columbia river after one to two years in the ocean, at the age of three or four years old. There are a large percentage of three-year-old returns to Okanogan. In comparison, very few three-year-olds return to Lake Wenatchee. After entering the Columbia River sockeye salmon adults bound for the Okanogan River system can be delayed up to three weeks as a result of water temperature. Adults congregate at the

confluence of the Okanogan and Columbia Rivers until the Okanogan water temperature falls below 21.1°C (Major and Mighell, 1966, Allen and Meekin, 1980, Chapman et al., 1995b). The Okanogan River sockeye typically spawn in October, peaking about the third week of October (Chapman et al., 1995b).

- **Lake Wenatchee ESU.** The Lake Wenatchee ESU is comprised of sockeye salmon from Lake Wenatchee that spawn in tributaries above the Lake. Resident sockeye, kokanee salmon, (*O. nerka*) observed spawning with the anadromous sockeye on the White and Little Wenatchee Rivers are considered part of this ESU. Lake Wenatchee sockeye typically spawn from mid-September to the beginning of October, peaking in the third week of September (Chapman et al., 1995b). There are very few three-year-old adult returns to Lake Wenatchee.

5.2.2.2 Sockeye Population Status under the ESA.

Snake River Sockeye salmon is currently the only population of sockeye salmon listed under the ESA that use the study area during any part of their life history. In, 1991 the Snake River Sockeye salmon were listed as endangered under the ESA (56 FR 58619). During the most recent sockeye salmon status review in Washington and Oregon, the NMFS did not find the Okanogan River and Lake Wenatchee ESU's warranted listing (64 FR 14529). However, they remain concerned about the two ESUs and will closely monitor the population status.

5.2.3 Steelhead (*O. mykiss*)

Steelhead (*O. mykiss*) exhibit one of the more complex life histories of pacific coast salmonids. There are two distinct life history forms of this species, an anadromous form "steelhead" trout and a resident form, "rainbow" trout. The inter-relationship between the two life forms is poorly understood. Currently, only the anadromous steelhead life from is listed under the ESA.

Steelhead typically spend two to three years in the ocean before returning to their natal stream to spawn. There are two reproductive ecotypes of steelhead "stream maturing" and "ocean maturing" commonly called summer and winter steelhead, respectively. The ecotypes are based on their state of sexual maturity at the time of river entry, and the duration of their spawning migration. Summer steelhead typically enter freshwater from May-October requiring several months in the river before spawning. Winter steelhead typically enter fresh water from November-April, spending only a few months in the river before spawning. The two runs are differentiated by the timing of adult returns, however, they frequently share common juvenile life-history behavior patterns (Barnhart, 1991). Although both life-history strategies are present in the Columbia River, only summer run fish are influenced by the John Day action.

Steelhead typically spawn between December and June (Bell, 1990; Busby et al., 1996). In general, steelhead differ from spawning chinook and coho salmon by their use of faster, shallower, and higher gradient locations in mainstem or tributary streams (Everest and Chapman, 1972).

Incubation rates vary with water temperature, and fry emergence usually occurs within 40 to 80 days after spawning. Everest and Chapman (1972) found age-0 steelhead residing over cobbles in water velocities of <0.15 m/s (<0.5 fps) in depths of 0.15 to 0.30 m (0.5 to 1.0 ft).

Juvenile steelhead will utilize stream margins and submerged rootwads, debris and logs to provide shelter and cover while rearing in their natal streams (Bustard and Narver, 1975).

Juvenile steelhead rear in freshwater for one or more years before migrating to the ocean (Barnhart, 1991). Chapman et al. (1994) found a positive relationship between migration speed of active migrants and fish size. Juvenile downstream migration for steelhead smolts occurs from April through June, with peak migration in general occurring in mid-April (Wydoski and Whitney, 1979).

Steelhead are capable of repeat spawning, although the incidence is relatively low. Steelhead will rarely spawn more than twice before dying. Repeat spawning in Washington ranges from 4.4 to 14.0 percent of total spawning runs (Wydoski and Whitney, 1979).

5.2.3.1 Known Occurrences in the Project Vicinity.

Steelhead from the following three ESUs may use the John Day Reach during some part of their life history:

- **Upper Columbia River ESU (Endangered).** The Upper Columbia River ESU is comprised of steelhead from the Columbia River and its tributaries upstream from the Yakima River to the United States-Canada border. Steelhead in this ESU are summer-run steelhead and contain some of the oldest smolt ages for steelhead, up to seven years. However, most fish smolt and outmigrate at age two and three (Chapman et al., 1994b).
- **Snake River Basin ESU (Threatened).** The Snake River Basin ESU is comprised of summer steelhead in the Snake River Basin. This ESU is composed of two groups to summer steelhead, an A-run and a B-run. The classification of the two runs is based on migration timing, ocean-age, and adult size.
- **Middle Columbia River ESU (Threatened).** The Middle Columbia River ESU is comprised of steelhead from the Columbia River and its tributaries from the Wind River upstream to and including the Yakima River excluding the Snake River Basin. The majority of steelhead in this ESU are summer run fish, and only summer run fish are influenced by the John Day action. They typically smolt at two years and spend one to two years in the ocean before returning to the Columbia River.

5.2.3.2 Steelhead Population Status under the ESA.

In August, 1997, the NMFS listed the Upper Columbia River ESU as endangered and the Snake River Basin ESU as threatened under the ESA. Most recently, on March 25, 1999 the Middle Columbia River ESU was listed as threatened.

5.2.4 Bull Trout (Salvelinus confluentus)

Columbia River bull trout exhibit three life-history patterns, resident, fluvial, and adfluvial. Resident populations of bull trout confine their migrations to within their natal stream. Fluvial populations usually migrate between their natal streams used for spawning and early juvenile rearing and large rivers used for adult rearing. Adfluvial populations usually migrate between their natal streams used for spawning and early juvenile rearing and lakes or reservoirs used for adult rearing (Buchanan et al., 1997). Anadromy is not currently found in

Columbia River bull trout populations Bond (1992). It is believed anadromy historically occurred.

Spawning in most bull trout populations occurs from August to November during periods of decreasing water temperatures. Spawning is generally initiated as stream temperatures decline below 10ºC. Most populations of stream resident fish spawn every year (Armstrong and Morrow, 1980). Studies conducted throughout the species range indicate that spawning sites are characterized by low gradient, shallow water from 20 to 60 cm (8 to 24 inches) deep, uniform flow, a gravel substrate between size 6 mm to 51 mm [0.25 to 2 inches] (Wydoski and Whitney, 1979; Brown, 1992; Fraley and Shepard, 1989). Spawning generally occurs between water temperatures of 5 to 9ºC. Groundwater influence and proximity to cover also are reported as important factors in spawning site selection (Fraley and Shepard, 1989). Spawning often occurs in small headwater reaches fed by cold streams, or near other sources of cold groundwater (Pratt, 1992).

Embryos incubate for approximately 100 to 145 days, and hatch in late winter or early spring (Pratt, 1992). Goetz (1989) and Rieman and McIntyre (1993) indicate that optimum incubation temperatures are between 2 and 4ºC. The alevins remain in the streambed, absorbing the yolk sac, for an additional 65 to 90 days (Pratt, 1992). Emergence from the streambed occurs in late winter/early spring (Pratt, 1992).

Fry are usually found in shallow, slow backwater side channels and eddies (Elliott, 1986). Young-of-the-year (YOY) bull trout are found primarily in side channel areas and along stream margins (Fraley and Shepard, 1989) similar to that reported for other species of salmonids. Juveniles (less than 100 mm, 4 inches in length) are primarily bottom dwellers and are found among coarse substrate (Thompson and Tufts, 1967; Fraley and Shepard, 1989; Pratt, 1992). Goetz (1989) suggested optimum water temperatures for fry rearing of about 7°C to 8°C. Larger juveniles and adults are often found in deep stream pools or in lakes in deep water with temperatures less than 15°C (Pratt, 1992).

Bull trout are opportunistic feeders, size and life history are the primary determining factors for prey selection. Resident and juvenile migratory bull trout prey on terrestrial and aquatic insects, macro-zooplankton, and small fish (Boag, 1987; Goetz, 1989; Donald and Alger, 1993) including sculpin, salmon fry, and other bull trout (Pratt, 1992). Adult migratory bull trout are primarily piscivorous, feeding on a variety of fish species.

The WDFW lists the following limiting factors for the bull trout in the Columbia River basin: stream temperatures exceeding the normal spawning and incubation temperature range; lack of spawning and rearing habitat; and a high percentage of fine sediment in spawning gravels (WDFW, 1998). Because of their close association with the bottom, bull trout are sensitive to changes in the streambed (Fraley and Shepard, 1989). Bull trout also readily interbreed with non-native brook trout *(Salvelinus fontinalis),* and brook trout competitively exclude bull trout (Buckman et al., 1992; Dumbacher et al., 1992).

5.2.4.1 Known Occurrences in the Project Vicinity.

Use of the study area by bull trout is unlikely. However, some straying of adfluvial migrants may occur. Bull trout have been infrequently observed in some mid- and lower Columbia River reservoirs. For example, migrating adults bull trout are annually caught from mid-May to mid-October at the Powerdale Dam trap on the lower Hood River near the Columbia

River. One of these fish was recaptured in the Columbia River approximately 11 km (6.8 miles) downstream of the mouth of Hood River (Buchanan et al., 1997). Bull trout were also observed in the past at fish ladders within the mid-Columbia River (Dell et al., 1975). Given these examples, it is possible bull trout from the following population may occasionally stray into the John Day Reach.

5.2.4.2 Columbia River Bull Trout DPS (Threatened).

Bull trout were listed as one distinct population segment (DPS) throughout the Columbia River basin. It is comprised of 141 subpopulations. These sub-populations are generally reproductively and geographically isolated, residing in restricted habitats typically in the upper reaches of tributaries to the Columbia and Snake river (Buchanan et al., 1997). Five subpopulations occur in the headwaters of the John Day and Umatilla River basins.

5.2.4.3 Bull Trout Population Status under the ESA.

In June, 1998, the Columbia River Bull Trout DPS was listed as Threatened under the ESA (63 FR 31647). The Columbia River DPS is represented by relatively widespread, geographically isolated subpopulations throughout the entire Columbia River basin within the United States and its tributaries, excluding bull trout found in the Jarbidge River, Nevada (63 FR 31647).

5.3 Normative River Concept

In their assessment of the recovery process for restoring healthy fish runs in the Columbia River System the Independent Scientific Group (ISG, 1995) stated that the runs can withstand a substantial amount of abuse in terms of harvest, hydroelectric impediments and hatchery influences, if sufficient quality spawning and rearing habitat exist. They point to the example of the upriver bright (URB) fall chinook stock in the Hanford Reach that exists within the hydro system yet they are increasingly successful in natural production. The ISG also discuss the importance of a meta-population, or a stable core population that supports development of other adjacent habitats. It is likely that URB populations from the Hanford Reach would extend itself to the John Day area if sufficient spawning and rearing habitat were available.

5.4 Resident Fish Communities

Historically, fish communities in the Columbia River Basin were dominated by coldwater salmonid and cottid fishes (Li et al., 1987). However, water resource development modified aquatic habitats and altered native biotic communities in rivers throughout the basin. Fish assemblages changed in response to these habitat modifications. They were further modified by the proliferation of introduced warmwater and coldwater fishes, such as percids and centrachids, in littoral and sub-littoral habitats of mainstem reservoirs (Li et al., 1987; Poe et al., 1994).

To assist in the assessment of potential drawdown influences on resident fish communities, the USGS examined trends in three years (1984,1985 and, 1995) of existing data collected from various shallow water littoral sites in the John Day Reservoir (Barfoot et al., 1999; Attachment A). The results of the fish community analysis are presented in Section 9.1. The

authors identified 30 distinct resident fish species representing nine families that were collected during the three years of sampling. Over half of the collected species were non-native fishes introduced over time in the Columbia River Basin. The prevalence of non-native fish has increased more than an order of magnitude over the last decade. An increasing abundance of introduced species more typically associated with lake-like habitats is characteristic of post-impoundment patterns of fish community changes described by Li et al. (1987) for impounded areas of the Snake and Columbia River Basins.

5.5 General Site-Specific Habitat Description

The John Day Reservoir was created when the dam was completed in, 1968, impounding 123 km (76 mile) of riverine habitat upstream to McNary Dam. It has a surface area of about 20,000 ha (49,500 acres), representing the second largest impoundment on the Columbia River. The converted river has the main channel area plus a number of backwater and wetland areas. These "converted" aquatic habitats provide a diversity of opportunities for aquatic organisms not available in the original river channel. Altered habitat conditions have favored development of biota preferring shallow and quiescent waters.

The Corps of Engineers data indicate a relatively flat reservoir from the dam upstream to River Mile (RM) 280. The pool elevations rise more sharply upstream of this point, creating conditions somewhat resembling a free-flowing river for approximately two miles to the base of McNary Dam. The upper third of the reservoir contains extensive shallow water habitat in main-channel and backwater areas, especially in the Blalock Island area (Figure 2). Further downstream the riverbanks become steep and shallow littoral habitat is limited. In a previous study, Li et al. (1981) characterized the reservoir into three velocity zones; 1) forebay (John Day Dam upstream to RM 247; 2) transitional (RM 247 upstream to RM 282) and 3) tailrace (RM 282 upstream to McNary Dam). The substrate in the reservoir is primarily mud, sand, gravel and cobble (Parsley et al., 1993). Gilbreath et al. (1999; Attachment B) found the majority of upper reservoir sediments in the shallow littoral zones were comprised of silty sand with a median grain size between 0.1 and 0.2 mm (0.004 to 0.008 in.). The fraction of silt and clay varied in the samples between 11 and 31 percent.

Primary aquatic production is dominated by phytoplankton in the reservoir. Reservoir turnover rate is relatively rapid, however, limiting the production of planktonic organisms in the water column. As a result, zooplankton are similarly limited in production.

Growth of aquatic macrophytes, primarily water milfoil (*Myriophyllum* spp.), is seasonally abundant in shallow littoral areas, particularly in shallow embayments and backwater habitats. Aquatic macrophytes have apparently greatly increased in abundance in the last decade (Barfoot et al., 1999).

Several studies have been conducted to examine the abundance and distribution of macroinvertebrates in John Day Reservoir. These studies and others in reservoir habitats in the Columbia and Snake River system support the generalization that shallow water habitats have higher biomass of benthic macroinvertebrates than deep-water habitats. Although biomass is high, the majority of the benthic community consists of burrowing forms like oligochaetes and chironomids (Bennett, 1999; Attachment C). These data appear to represent an apparent shift in the benthic macroinvertebrate community, since earlier studies revealed a large number of invertebrates were abundant in the backwaters of the reservoir.

The general effect of mainstem reservoirs on water temperatures has been to delay the occurrence of maximum temperatures in late summer as well as to delay early autumn cooling. The surface waters of the John Day Reservoir during the summer months are warm, peaking around 22°C in August and staying above 20°C into October, annually. Bottom temperatures in the reservoir are only slightly cooler, 0.5 to 0.8°C, than the surface measurements (Gilbreath et al., 1999). Backwater areas can be 1 or 2°C warmer than the mainstem during any month (Gilbreath et al., 1999).

5.6 Ongoing Biotic Trends

Changing macrophyte densities and altered resident fish community structures may eventually influence juvenile salmonid rearing in shallow water habitats of the John Day Reservoir. Aquatic vegetation could play a role in altering shallow water trophic dynamics in some areas of the reservoir by changing water circulation patterns, DO, pH and the local water temperatures. Macrophyte beds can also create structurally complex spawning and rearing habitats for many of the introduced fish species present in the reservoir. This shift may in part provide an explanation for the increased abundance of introduced taxa (e.g., sunfishes and yellow perch) in 1995 compared to the mid-1980s. Many exotic fish species now abundant in shoreline and backwater areas of the reservoir could prey upon or directly compete with juvenile salmonids. The anticipated changes in predation and competition with the various drawdown scenarios are presented in Section 9.3.

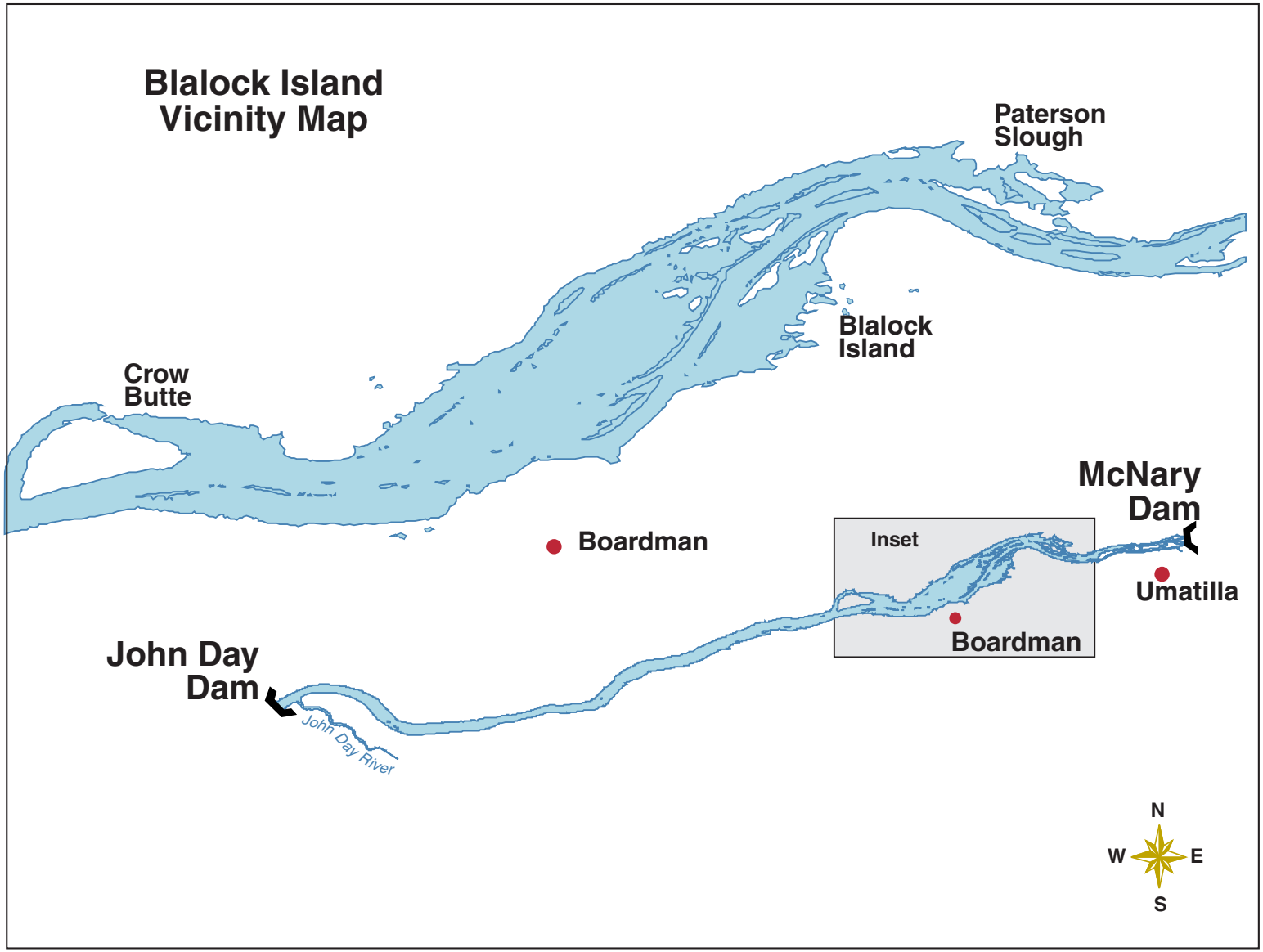


Figure 2 Blalock Island vicinity map.

SECTION 6. Analysis of Effects on Juvenile Salmonids

The objective of this section is to review the effects of various drawdown scenarios on juvenile salmonid fishes by evaluating: 1) juvenile fish passage through both the reservoir and the dam, 2) the timing and duration of reservoir rearing, 3) total project smolt survival and the potential numbers of returning adults and 4) extent of stock recovery for each of the evaluation species. Life-stage history differences between the evaluation species have been summarized in Section 4.3.3.14.1. The important aspects related to passage behavior and survival of the various species are the outmigration timing, the life-history strategy (ocean-type vs. stream-type) of the fish and the size of the juveniles when they enter the John Day Reservoir.

By the time juvenile salmonid outmigrants reach the John Day Project, most are actively migrating smolts in later stages of smoltification. They represent yearling or older fish, exhibiting stream-type characteristics. Steelhead and sockeye are generally the largest outmigrants, followed by spring chinook. These fish move quickly through the reservoir (three to eight days), generally offshore in fast water. Migration is purposeful and direct. The rate of downstream progress has been shown to be directly related to river flow, stream temperatures and the level of smolt development (Giorgi, 1993). Fall chinook salmon are the only evaluation species exhibiting mostly ocean-type characteristics, with a majority of smolts migrating as sub-yearling fish. These fish are thought to rear in the shallow littoral zones of the reservoir as they slowly make their way downstream. Migration speed is sporadic and generally unrelated to river flow. It is believed the Hanford Reach stock and upstream Columbia River stocks exhibit these characteristics. Snake River fall chinook stocks act in similar fashion in their natal streams and upstream in the mainstem Snake River. However, it is thought by the time smolts from these stocks reach John Day, they are actively migrating through the system similar to yearling or older outmigrants (J. Petersen, USGS pers. comm., 1999). If accurate, this hypothesis is important from an ESA perspective, since the Snake River fall chinook stock is one of the ESA-listed species of concern for the John Day Project, while the Hanford Reach and upper Columbia River stocks are not.

The influence of various drawdown alternatives at John Day dam on juvenile salmonid fishes is a combined factor of changes in both: 1) juvenile passage past the dam and 2) juvenile passage through the reservoir commensurate with different reservoir elevations, river discharges and current velocity. Losses as a result of dam passage are a function of passage route selected by the fish (turbines, spill, bypass system, locks or ladders) and the degree of injury inherent with each route. River discharge, water temperature, the amount of spill and the size of the outmigrant can all affect the passage success rates. Reservoir-related mortalities are assumed to be primarily a function of passage timing (speed and season of the year), and the resulting degree of exposure to salmonid predators.

The migratory behavior of subyearling juvenile fall chinook results in them having lower collection efficiencies into bypass systems at mainstem Columbia River and Snake River dams and higher dam passage-related mortalities than yearling migrants. This race of chinook is also slowest to navigate the reservoir due to the preponderance of in-reservoir rearing. Given their early summer migration through the reservoir, fall chinook are thought

to have more exposure to predatory fishes due to greater spatial and temporal overlap with predator activity than the other salmonid species.

The following sections summarize the available knowledge base concerning juvenile salmonid fish passage through the reservoir and past the John Day dam, including a discussion of the potential for the production of rearing salmonid life stages in the reservoir.

6.1 Reservoir Rearing

Reservoir rearing is important to understand, not only for juvenile growth and well being, but also for timing and duration of the fish in the reservoir related to availability to predators and ultimately to passage success. In this section the distribution of the juveniles in the reservoir, the seasonal timing, the habitats utilized and an estimate of the change in preferred rearing habitats are evaluated for each species and drawdown scenario.

The importance of mainstem reservoir habitat for rearing of salmonid fishes varies by species. It is generally believed that yearling migrants tend to migrate rapidly through the reservoir in mid-channel areas, exhibiting little or no reservoir rearing time (Ledgerwood et al., 1991b; Zook, 1983). However, little information is available concerning use of reservoir rearing habitat by very early life stages (i.e., fry) of naturally produced fall chinook salmon. This habitat may be very important to the productivity of naturally produced fall chinook. In spite of the facts that Upriver Bright (URB) fall chinook are more susceptible to predation-related losses, to dam passage-related mortality, and to high ocean and inriver harvest rates in comparison to other Columbia River and Snake River salmonid stocks as discussed in this report, Upriver Bright fall chinook is the most robust anadromous stock in the Columbia River Basin above Bonneville Dam. Use of rearing habitat in the John Day pool may be critical to the productivity of this stock.

6.1.1 Juvenile Fish Distribution and Habitats Utilized in the John Day Reservoir

6.1.1.1 Spring Chinook Salmon.

Rearing, growth and overwintering of juvenile spring chinook salmon generally occurs in headwater tributary streams. Juvenile spring chinook primarily use the John Day Reservoir as a migration corridor. Within-reservoir movements have been monitored in the immediate vicinity of the John Day Dam forebay. Graphical representation of juvenile spring chinook migration data indicates that as fish migrate from the upstream reservoir to the forebay they encounter the John Day River plume. They either avoid or are shunted away form the turbid, warm water which is most pronounced along the Oregon shore. Giorgi et al. (1985) found that fish abundance and distribution in the forebay was directly correlated with increasing water clarity and that in no case was the distribution of spring chinook correlated with either water velocity or water temperature. The speed of migration through the reservoir is rapid for this stock and additional rearing time in the reservoir is not apparent.

6.1.1.2 Summer Steelhead.

Ninety percent of the steelhead rearing production occurs in hatcheries in the region (Chapman et al., 1994c). The balance of the rearing production occurs in tributary streams,

although some minor amount of reservoir rearing may occur during overwintering or if individual fish residualize. Giorgi et al. (1985) tracked the distribution of steelhead smolts in the forebay of the reservoir. Steelhead smolts migrated along the middle of the channel and passed through the forebay at or faster than the mean water particle travel time (WPTT), suggesting they did not spend substantial time to rear in the forebay area.

6.1.1.3 Sockeye Salmon.

Although sockeye could conceivably rear in the reservoirs, the rapid flushing rate, low primary productivity and lack of abundant zooplankton limit production potential (Chapman et al., 1994b). Sockeye smolts are presumed to migrate quickly through the reservoir at a similar rate to steelhead and spring chinook. Reservoir rearing is thought to be limited for this species.

6.1.1.4 Fall Chinook Salmon.

Fall chinook use nearshore littoral habitat in the reservoir for rearing in late spring and early summer (Chapman et al., 1994a; Burley and Poe, 1994; Barfoot et al., 1999 [Attachment A] Gilbreath et al., 1999 [Attachment B]). Unlike yearling stocks, fall chinook subyearlings remain most abundant along shoreline stations during outmigration, in comparison to mid-reservoir stations in the forebay as determined by purse seine collections (Giorgi et al., 1985). Recently emerged fall chinook fry drift downstream into John Day Reservoir and rear throughout the shallow, low velocity areas of the reservoir in April through June, annually (Gilbreath et al., 1999; Barfoot et al., 1999). This type of rearing behavior occurs commonly among other ocean-type chinook salmon populations on the Pacific Coast. Hayman et al. (1996) found the greatest densities of subyearling chinook along stream margins in a large mainstem river in Western Washington. They were especially dense in backwater areas. Kiefer and Forster (1989) also found off-channel ponds connected to the Crooked River in Idaho, offered the greatest densities of Snake River chinook juveniles.

After reaching approximately 50 mm in size, chinook move slightly offshore into faster flowing water during days and typically establish feeding territories along the river bottom (Lister and Genoe1, 1970; Everest and Chapman, 1972; Hillman et al., 1987; Rondorf and Gray, 1987; Campbell and Eddy, 1988; Chapman et al., 1994a). At night they have a tendency to move back inshore to shallow, quiescent resting locations (Campbell and Eddy, 1988). Roper et al. (1994) found subyearling, on their way to becoming yearling chinook, heavily concentrated in pool habitats in a large tributary of the south Umpqua River in Oregon and that chinook densities in stream reaches were a function of the quantity of pool habitat in each reach. Chinook may feed on limnetic species (free-swimming invertebrates in the water column of lake-like environments) when available but prefer benthic macroinvertebrates (bottom-dwelling insects) drifting in the current. (Chapman et al., 1994a). Based on these observations, it appears that most suitable chinook rearing habitat is located in the upstream portion of the reservoir, where river velocities are greater and the substrates coarser (less fine sediment) than downstream in the reservoir. However, no surveys have been done in the reservoir to determine habitat preferences for rearing areas.

Gilbreath et al. (1999) found 0-age juvenile fall chinook salmon (42 to 98 mm in size; surmised to be upriver bright of the Hanford Reach stock) along the nearshore habitats of the upper reservoir from May to mid-July annually during, 1994 and, 1995. Less than one

percent of the chinook fry catch occurred in early July and juvenile chinook were absent in the catch from late July through November, annually. Reservoir water temperatures reach 20°C by late July. Water temperatures above, 19°C are suboptimal for juvenile chinook growth and metabolism (Raleigh et al., 1986; Bell, 1990; Reiser and Bjornn, 1991), and may account for the lack of nearshore catches after mid-July. Barfoot et al. (1999) noted similar seasonal distributions of chinook fry in, 1984, 1985 and, 1994 in the upper John Day Reservoir. A few outmigrants were collected offshore in the forebay area during August, 1985. The observation of chinook in August was not consistent from year to year and may be a factor of the annual variability of annual water conditions.

6.1.1.5 Bull Trout.

Bull trout in the mainstem Columbia River are usually associated with tributaries that contain resident populations (Mongillo, 1993). Although bull trout populations in Columbia River tributaries are not considered anadromous (Mongillo, 1993), they may be present year-round in some of the reservoirs (Dell et al., 1975; Welsh et al., 1994). They may occur as lake resident adults, exhibit adfluvial life history forms (rearing in lakes and spawning in tributaries to the lake) or they may simply stray on occasion from upstream tributaries to Columbia River reservoirs (Beak, 1995). Fry are usually found during early spring in shallow, slow backwater side channels and eddies (Elliot, 1986). Young-of-the-year bull trout are found primarily in side channel areas and along stream margins (Fraley and Shepard, 1989) similar to that reported for other salmonid species. Juveniles (less and 100 mm in length) are primarily bottom dwellers and are found among coarse substrate (Thompson and Tufts, 1967; Fraley and Shepard, 1989; Pratt, 1992). In reservoirs, the fish will move out of the shoreline areas and into deep, cool water as thermoclines (temperature gradients) develop (Pratt, 1992).

No native char have been observed to date during sample collection in the John Day Reservoir. The use of the reservoir for char rearing is uncertain.

6.1.2 Estimated Change in Preferred Habitat Area

6.1.2.1 Yearling Outmigrants.

There will be little to no effect of drawdown on reservoir rearing of species exhibiting primarily yearling or older juvenile outmigration strategies (stream-type fish) like spring chinook, steelhead and sockeye because they do not rear in the reservoir. Benefits or impacts to these species will primarily arise with decreased travel time through the reservoir as discussed in the subsequent section.

6.1.2.2 Sub-yearling Outmigrants.

Juvenile fall chinook salmon that primarily emigrate as subyearling (ocean-type) fish are known to utilize the nearshore habitat of the John Day Reservoir for rearing during the spring and early summer months. Consequently, drawdown of John Day Reservoir could substantially affect fall chinook production. Drawdown to spill crest level (Alternative 1) or to the natural river level(Alternative 3) will alter the aquatic environment from slower flowing to faster flowing conditions. The surface area of water in the reach will change dramatically with drawdown as shown in Table 2 (USGS 1999).

Table 2.
Changes in Surface Area and Volume of the John Day Reservoir at 300 kcfs under Existing Conditions and Four Drawdown Alternatives; Flood Control at 500 kcfs (WEST, 1999).

Drawdown Alternative	Surface Area (acres)	Pool Volume (acre-ft)
Existing Conditions	51,409	2,396,332
Alternative 1, Spill Crest without Flood Control	33,307	1,229,691
Alternative 2, Spill Crest with Flood Control[1]		1,706,426
Alternative 3, Natural River without Flood Control	26,505	534,395
Alternative 4, Natural River with Flood Control[1]		700,137

1) When flood control is imposed (>515kcfs), the pool would increase in elevation by almost 6m (20 ft) in each case (WEST, 1999). A flood release period of, 19 to 26 days would be required at the maximum flow rate before the flood control volume is reduced to zero again.

Drawdown to natural river level would result in the loss of nearly 25,000 acres (39 mi^2) of reservoir surface area under Alternative 3. Drawdown to spillway crest (Alternative 1) would reduce the surface area of the pool by 18,100 acres (28 mi^2) or approximately 72 percent of the pool reduction under Alternative 3.

The effect of reduced rearing habitat currently in use in the reservoir on the production of fall chinook is uncertain.

The up-river bright (URB) stock of fall chinook presently supports a robust and increasing population of salmon in the Columbia River. Use of nearshore habitat in the John Day Reach by fall chinook juveniles for spring and early summer rearing has been documented and it is possible that the extensive shallow-water rearing environment occurring presently in the upper John Day Reservoir may provide considerable benefit to the production of this stock. With the loss of historic habitat in the Columbia River estuary area, the substitution of habitat provided in the McNary and John Day reservoirs may be very important for fall chinook natural production. Researchers have noted that backwaters, off-channel ponds, and nearshore stream margins with low water velocities may comprise the best habitat conditions for small rearing juveniles (Lister and Genoe, 1970; Everest and Chapman, 1972; Hillman et al., 1987; Rondorf and Gray, 1987; Campbell and Eddy, 1988; Kiefer and Forster, 1989; Chapman et al., 1994a; Hayman et al., 1996). Given that shallow water, nearshore habitat has been shown to be important for fall chinook rearing and life history diversity in other areas, that the use of existing nearshore habitat in the upper end of the John Day pool by fall chinook has been documented, and that the current status of the URB stock is robust, we conclude that substantial risk may exist in modifying the habitat of this healthy stock without a better understanding of the importance of this habitat to the stock's productivity.

Potential subyearling fall chinook rearing habitat was estimated using habitat suitability information in a hydraulic model for the John Day Reach as described in Battelle and U.S. Geological Survey (in review). The USGS used data collected in the Hanford Reach of the Columbia River (Key et al., 1996) to construct a simplified model incorporating velocity, depth and distance from shore as a means to assess the quantity of potential rearing habitat available for juvenile fall chinook salmon under existing reservoir and under drawdown conditions. Water depths greater than 1.6 m (5.2 feet), velocities greater than 1.21 m/sec (4.0 fps) and distances from shore greater than 7.6 m (25 feet) offshore were deemed not characteristic of quality rearing habitat (Key et al., 1996).

No consideration of additional habitat parameters such as substrait, presence of vegetation or structure, or water temperature was incorporated into the model as related data was generally not available. The modeling results represent only a rough estimate of the change in potential rearing habitat available under the drawdown alternatives. The modeling approach delineated rearing habitat for subyearlings as a discriminate probability function that classifies fish presence or absence in each 10 m^2 (108 ft^2) habitat cell and the probability of a correct classification. High probability habitat was defined as habitat cells with greater than or equal to a 70 percent probability that 10 or more juvenile chinook salmon (1.00 fish/m^2) were present. Cells with less than 70 percent probability were defined as low probability habitat.

The total acres of high and low probability habitats were calculated for three drawdown scenarios (normal operating pool, spillway crest (Alternative 1), and natural river [Alternative 3]) at three river discharge levels (100, 156, and 300 kcfs). The results of the 300 kcfs flow model are reported in Table3 as this level of flow most closely represents normal conditions found in John Day Reservoir during the peak occurrence of rearing fall chinook salmon juveniles in May and June annually. Modeled results between alternative drawdown scenarios is generally consistent with Table3, regardless of the river discharge modeled.

Table 3.
Acres and Probability of Habitat Cells Containing 10 or More Subyearling Chinook Salmon under Current, Spillway Crest and Natural River Drawdown Alternatives at 300 kcfs in John Day Reservoir.

Parameter	Normal Operating Pool (Acres)	Spillway Crest (Acres)	Natural River (Acres)
Total Acres	49,406	32,025	25,486
Not likely used for Rearing	48,007	30,842	24,052
Total Potentially Useable	1,399	1,305	1,434
Probability > 70%	686	628	785
Probability < 70%	713	554	649

The total acres of high probability rearing habitat for subyearling chinook salmon was highest under the natural river alternative for all river flows modeled. The model analysis shows changes in river morphology resulting from a reservoir drawdown to natural river provides slightly more rearing habitat area compared to current conditions. Thus, even with

a loss of approximately 25,000 acres of surface area, there may be no net loss, and perhaps a small increase, in available rearing habitat for fall chinook salmon under drawdown to natural riverine conditions.

Since riverine environments produce more diverse and higher quality food supplies for fall chinook (Becker, 1973; Dauble et al., 1980) than reservoir habitats (Muir and Emmett, 1988; Rondorf et al., 1990), the natural river alternative may offer greater production potential for juvenile fall chinook salmon than either current conditions or the spillway crest alternative. Higher water velocities found in riverine systems promote a diversity of substrates and microhabitats that result in a richer, more abundant invertebrate community than found in reservoir systems. Currently, riverine conditions between John Day and McNary dams can be found in the upper reaches of the reservoir. Additional directed study, including field investigations over a multi-year period, would be needed to more thoroughly assess the likely impacts of John Day drawdown on the productivity of fall chinook salmon.

6.2 Reservoir Passage

Adverse effects of reservoirs on outmigrating juvenile salmonids are thought by some to be much less of an influence than passage at dams. Iwamoto et al. (1994) and Muir et al. (1995) indicated that virtually all of the mortality of yearling outmigrants measured in the mainstem Snake River was attributed mainly to fish passing thought the hydroelectric structures and that the reservoirs themselves were quite benign. Conversely, many researchers believe reservoir-related mortalities as a function of predation, can equal or exceed dam passage losses for subyearling outmigrants (Beamesderfer et al., 1990; Rieman, 1990, NMFS, 1999).

Reservoir impoundment can create increased rearing area and provide overwintering habitat for juvenile anadromous salmonids. It can also affect the outmigration of anadromous salmonid juveniles by causing extended travel times, residualization and decreased survival rates. Juveniles, when exposed to extended travel times and increased water temperatures, can residualize (become residents) and fail to migrate to the ocean that year.

6.2.1 Rate of Migration

Raymond (1968, 1969, 1979) and Bently and Raymond (1976) estimated that juvenile anadromous salmonids move through the Snake River and Lower Columbia River impoundments one-half to one-third slower than they would through free-flowing river sections of the same length. The minimum time it takes a particle of water (water particle travel time; WPTT) to traverse the John Day Reservoir is a function of pool elevation and river discharge and velocity (Figure 3 and Table 4). If the rate of fish outmigration is passive with respect to river velocities, than the potential benefit under drawdown would be equal to the change in WPTT. At 300 kcfs, a typical river discharge during the outmigration period, the expected change in the minimum WPTT is between 1.3 and 1.8 days, for drawdown Alternatives 1 and 3 respectively.

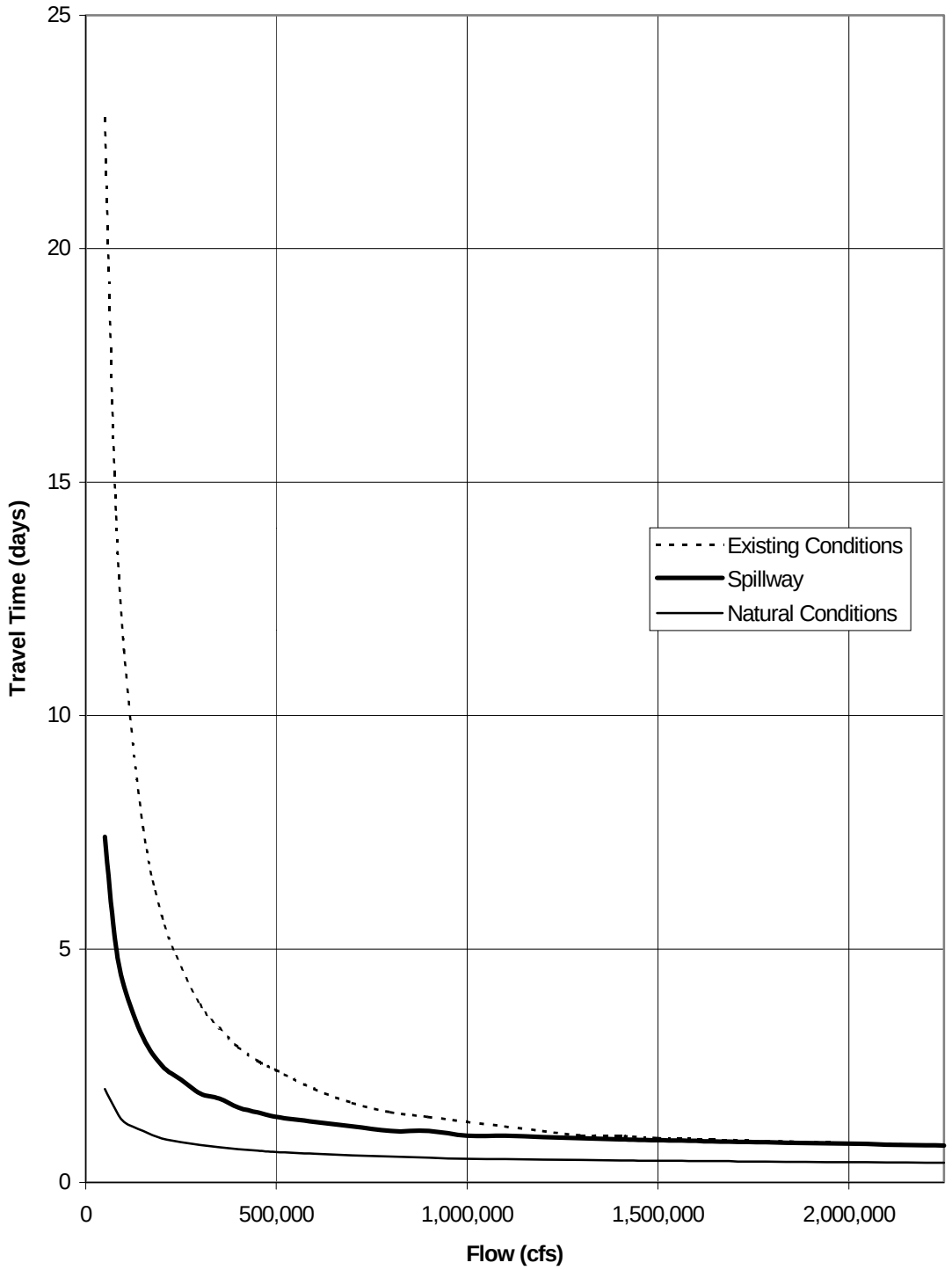


Figure 3. Flow vs. travel time – John Day Reservoir.

Table 4.
Average Water Particle Travel Time (WPTT) in the John Day Reservoir under Various Drawdown Alternatives and River Discharges (after WEST, 1999).

	WPTT in Days			
Drawdown Alternative	100 kcfs	200 kcfs	300 kcfs	400 kcfs
Existing Conditions	11.4	5.7	3.8	2.9
Alternative 1, Spill Crest without Flood Control	4.2	2.5	1.9	1.6
Alternative 2, Spill Crest with Flood Control[1]	NA	NA	NA	NA
Alternative 3, Natural River without Flood Control	1.3	0.9	0.8	0.7
Alternative 4, Natural River with Flood Control[1]	NA	NA	NA	NA

1) When flood control is imposed, the pool would increase in elevation by as much as 6m (20 ft) under Alternative 2 or 17m (55 ft) under alternative 4 (WEST, 1999). WPTT will be slower (increasing WPTT by approximately 40%) for these alternatives compared to their counter part for a period of, 19 to 26 days during the flood release period before the flood control volume is reduced to zero again.

Under normal conditions during the outmigration period (300 kcfs), a benefit of approximately two to three days faster travel time could be realized with drawdown to spillway crest and natural river, respectively. As river discharge increases above 300 kcfs, the level of change from current conditions diminishes. The level of WPTT change is faster as river discharges decrease below 300 kcfs. The benefit of drawdown is considerable only during extreme low flow years, as shown at 100 kcfs when the change would be in the range of 7 to 10 days faster with drawdown. A flow level of 100 kcfs rarely occurs during the spring runoff period. It represents more than a 98.5 percent exceedence value (occurs less than 1.5 percent of the time; a frequency of less than once in 67 years) during the months of April to June (USGS gauge 14105700, flow duration curves for April – June, 1974 to, 1998). Typical flow rates during the outmigration period range between 200 and 400 kcfs. WPTT benefits during outmigration would typically range between one and five days depending of river discharge and the drawdown scenario. Based on hydrograph frequencies the benefit would normally range between two and four days.

6.2.1.1 Existing Smolt Travel Time.

PIT Tag Data. Skalski and Townsend (1999) [Attachment D] summarized information regarding juvenile fish travel time through the John Day pool based on analysis of PIT-tag data (i.e., McNary Dam PIT-tag detector to John Day Dam PIT-tag detector). Available information is largely limited to baseline data beginning in, 1998, since either detection probabilities were too low or detection facilities were unavailable to estimate travel times prior to, 1998. The travel time results are summarized in Table 5.

Table 5.
Current Average Travel Time Statistics Based on PIT-tag Data for Various Species Migrating through the John Day Reservoir (after Skalski and Townsend, 1999).

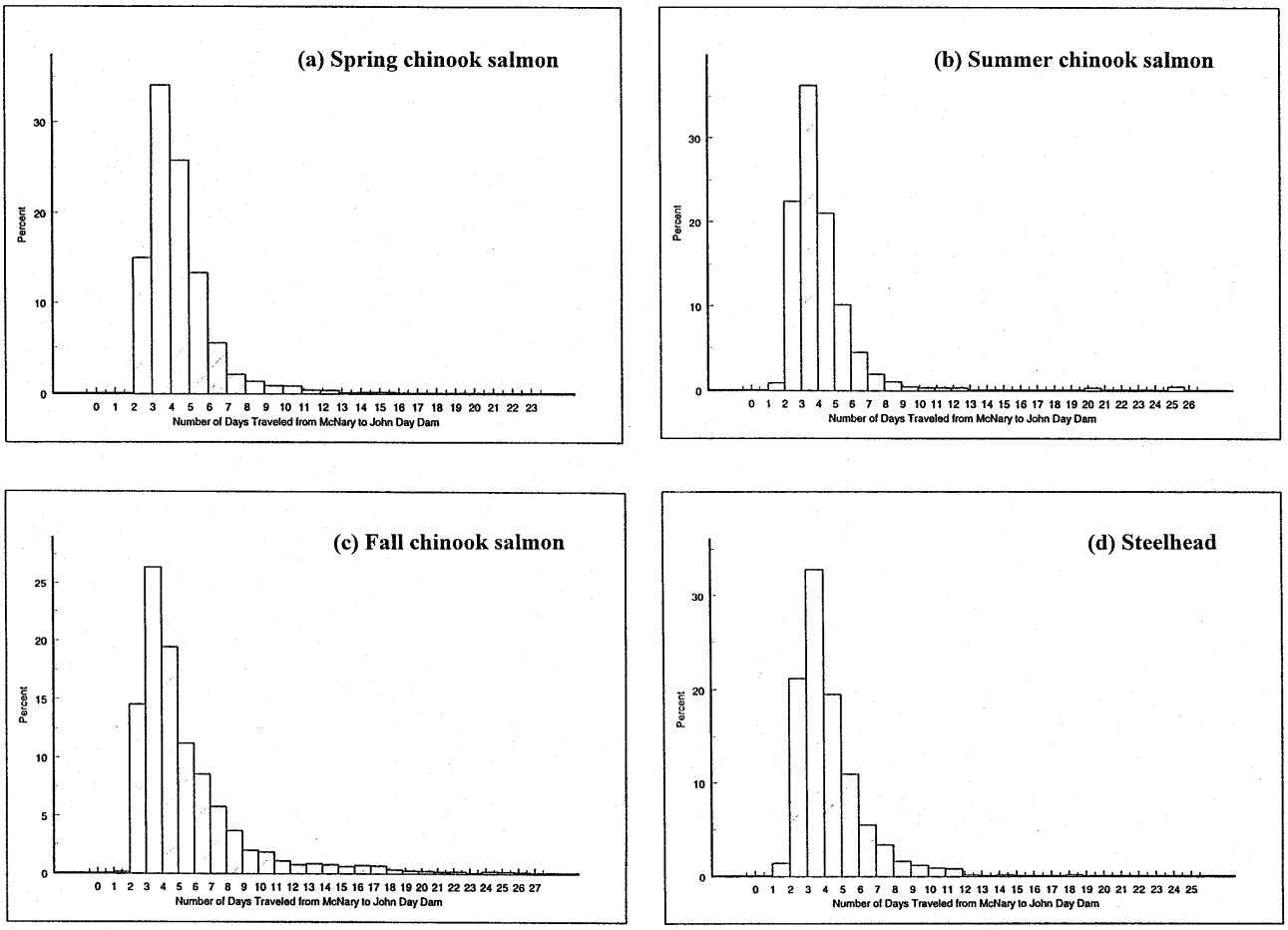
Species/Year Class	Arithmetic Mean Travel Time[1] (days)	Mean Migration Rate (miles/day)
Spring Chinook –, 1998	3.9	20
Summer Chinook –, 1998	3.7	21
Fall Chinook –, 1998	5.1	15
Steelhead –, 1997[2]	2.3	33
Steelhead –, 1998	3.9	20

1) See Attachment D for statistical details.

2) The, 1997 water year was very wet resulting in higher stream flows and faster reservoir currents than normal.

Travel time curves based on the PIT-tag data for the various fish runs are shown in Figure 4. As expected, fall chinook traveled significantly slower than the other three salmonid stocks in, 1998. The PIT-tag information provided herein for fall chinook represent the results of hatchery released fish. These fish are slightly larger than natural reared fall chinook and they may be ready to actively migrate seaward. The data may not reflect the rearing characteristics or migration rate of smaller fall chinook from naturally produced stocks. It is assumed the migration rate for fall chinook fry (ocean-type) rearing in the reservoir is slower than reported in Table 4. It is further assumed changes in WPTT with drawdown will have less effect on modifying the outmigration rates of these stocks compared to their hatchery counterparts.

Figure 4. Histograms of travel time frequencies at the John Day Dam project during, 1998 for (a) spring chinook, (b) summer chinook, (c) fall chinook, and (d) steelhead



Travel times through the John Day project in, 1998 were correlated with ambient river conditions to determine if possible relationships exist that might influence travel times. River flows ranged between 100 and 400 kcfs, while spill ranged from 0 to 225 kcfs, turbidity was quite variable between 0.5 and 4.0 NTU and temperatures steadily increased from around 10°C to 23°C, during the migration season (April 16th through August 23rd). All of these factors showed considerable covariance between each other. Skalski and Townsend (1999) statistically verified that as flow and spill volumes increased, mean travel time decreased on a daily basis for all the stocks during, 1998. Temperature was also found to be correlated with travel time. In three of four cases, travel times decreased as temperatures increased, which may be directly related to the thermal regime or it may be a factor of enhanced smolt development later in the migration season. Turbidity showed less of a consistent relationship with travel times.

The authors concluded that the mean travel speeds of smolts through the John Day project were quite fast; with the arithmetic means ranging between 33 miles/day for steelhead in, 1997 to 15 miles/day for fall chinook salmon in, 1998. Steelhead mean travel time was significantly shorter in the John Day project than chinook stocks.

Radiotelemetry Data. Giorgi et al. (1985) radiotagged and tracked a small number of spring chinook and steelhead smolts released nearly 6.4 km (4 miles) above John Day Dam in, 1983 and, 1984. Their data indicate spring chinook moved in the forebay at an average rate of 0.18 m/sec (0.59 fps) absolute progress toward the dam. The rate was highly variable ranging from 0.04 to 0.40 m/sec (0.12 to 1.28 fps). Steelhead moved slightly faster (14 percent) than spring chinook in the forebay averaging 0.20 m/sec (0.67 fps) progress toward the dam. Steelhead were 10 percent larger and the river discharge was four percent higher than during the spring chinook monitoring.

Similarly, Sheer et al. (1995) tracked 100 spring chinook released in seven groups between May 2nd and June 8th, 1995 at river flows between 210 and 296 kcfs with spillway discharges ranging between 8 and 14 kcfs (4 to 5percent spill). The corresponding water particle travel time (WPTT) through the reservoir at these discharges would have ranged between 3.8 and 5.7 days during the spring outmigration period. The median time for spring chinook smolts from the release point at Towal, WA, 8 km (5 miles) to John Day dam was 5.7 hours (1.4 km/h at 0.04 m/sec. [1.3 fps], or 21 miles per day).

The average, 1995 migration rate for spring chinook was slightly faster than the water particle travel time through the reservoir. It was substantially faster (2.2 x) than the travel time measured by radiotelemetry techniques in, 1983 and, 1984, even though the river discharges during the monitoring in, 1995 were less.

Based on recent radiotelemetry and PIT-tag data, the mean travel time for all stocks traversing the John Day pool at river flows considered typical during the outmigration season, range between 2.3 and 5.1 days (at a corresponding migration rate of 15 to 33 reservoir miles per day). The water particle travel time (WPTT) through the existing reservoir at different flow rates is shown in Table 6. For river flows between 200 and 400 kcfs, a particle of water will traverse through the reservoir in 2.9 to 5.7 days at a mean velocity of 0.8 to 1.6 fps or 13 to 26 miles/day. Mean smolt travel times for all stocks in, 1983, 1984, 1995, 1998 fell within or were slightly faster than this range.

Under natural river conditions across the John Day Reservoir the WPTT would range between 0.7 days at 400 kcfs and 0.9 days at 200 kcfs. Under the same river discharges, the WPTT for the drawdown scenario to spillway crest would range between 1.6 and 2.5 days. If fish migration is passive with river velocities, average transit times based on current PIT-tag data could be reduced from no benefit for steelhead up to 3.5 days for fall chinook under drawdown to spillway crest (Alternative 1) and from 1.4 for steelhead up to 4.4 days for fall chinook under the natural river scenario (Alternative 3) depending on river discharge (Table6).

Table 6.
Estimate Reduction in Travel Time (days) if Migration Speed Is Assumed to Be a Direct Function of WPTT (Based on Current PIT-tag Data; Skalski and Townsend, 1999).

	Alternative 1 Spillway Crest		**Alternative 3 Natural River**	
Salmonid Race/Year Class	**(200 kcfs) (days)**	**(400 kcfs) (days)**	**(200 kcfs) (days)**	**(400 kcfs) (days)**
Spring Chinook/1998	-1.4	-2.3	-3.0	-3.2
Summer Chinook/1998	-1.2	-2.2	-2.8	-3.0
Fall Chinook/1998	-2.6	-3.5	-4.2	-4.4
Summer Steelhead/1997	0.0	-0.7	-1.4	-1.6
Summer Steelhead/1998	-1.4	-2.3	-3.0	-3.2
Overall Benefit	0.0 to -2.6	-0.7 to 3.5	-1.4 to 3.0	-1.6 to 4.4

Although several studies indicated that water velocity is a primary determinant of juvenile migration speed (Smith, 1982; Buettner and Brimmer, 1995; Berggren and Filardo, 1993) other studies suggest factors other than flow may be affecting the dynamics of outmigration (Mains and Smith, 1964; Giorgi et al., 1985; Beeman and Rondorf, 1992; Achord et al., 1994; Chapman et al., 1994; Skalski and Townsend, 1999). Migration speeds increase later in the outmigration season as water temperatures increase. There is also strong correlation between the level of smolt development and the time of year. Migration speed is faster at higher levels of smolt development. As discussed previously, active rearing in the reservoir can also slow down the rate of migration as evidenced by fall chinook results. The literature suggests changes in WPTT in the reservoir under drawdown will primarily benefit yearling smolt migration rates early in the spring, especially during extreme low flow periods.

6.2.2 Predation Estimates

Increasing migration rates through the John Day Reach with drawdown may reduce the exposure of juvenile migrants to predator fishes and decrease juvenile losses. Reducing juvenile fish travel time up to four days at river flows that normally occur during the outmigration period could reduce predation-related mortalities of salmonid fishes in the John Day Reach. This factor was incorporated into the fish passage model of Snake River chinook salmon as discussed in a subsequent section. The estimated change in travel time through the reservoir for fall chinook, the species with the greatest overlap and exposure to predators, resulted in an estimated increase in in-river survivals of two to three percent for Alternatives 1 and 3, respectively.

6.2.3 Reservoir Survival Estimates

Reservoir survival is assumed to be a function of travel time; primarily related to predation rates. According to Poe (1992), the primary mechanism responsible for juvenile mortality associated with downstream reservoir migration is predation by piscivores (fish eating fishes). Faster migration could mean less exposure to predatory fishes. Yearling outmigrants moving past John Day during spring are rapidly migrating through the reservoir. Their temporal presence generally does not overlap the peak activity period of predators that occurs during warmer summer months. They migrate primarily through the fast moving reservoir water located offshore, also minimizing overlap with piscivorous fishes (Ledgerwood et al., 1991b and Zook, 1983). Conversely, fall chinook migrating during the summer months as subyearlings utilize the nearshore waters for rearing during their outmigration through the reservoir (Chapman et al.1994a; Burley and Poe, 1994). Their temporal and spatial presence overlaps more directly with predator fishes. The slower rate of reservoir travel and later timing of migration exposes fall chinook stocks to greater levels of predation than other stocks.

Attempts have been made to apportion juvenile downstream migration mortality between dam and reservoir passage. Chapman et al. (1994a) state that reservoir-passage mortality for juvenile yearling chinook has been estimated at 5 to 10 percent and that the majority of reservoir-related mortality appears to occur in the tailrace below dams where predators congregate. In their Habitat Conservation Plans (HCPs) under Section 10 of the Endangered Species Act (ESA), the mid-Columbia PUDs (1998) have assumed reservoir mortalities for juveniles of all the salmonid species at their projects are averaging approximately 4 percent. Ongoing survival studies at each of the mid-Columbia projects are designed to address this issue.

Studies in the Snake River (Muir et al. 1995) estimated total survival rates for yearling chinook range between 82 and 92 percent per project. The authors did not attempt to apportion passage mortality between the reservoirs and dams. If it is assumed dam passage-related mortalities are near 10 to 15 percent per project, this work indicates reservoir-passage mortality of yearling chinook may be low. It agrees with the work of Iwamoto et al. (1994) that indicated reservoir mortality of yearling chinook salmon was a small fraction of the total project mortality in the Snake River hydroelectric projects. Conversely, Snake River fall chinook salmon experience higher rates of mortality through impounded reaches than other species. Summer PIT tag data suggest survival ranges between 62 and 95 percent and averages 75 percent per project (NMFS, 1999).

PIT Tag Data. From the, 1998 PIT-tag baseline data for the John Day Reach, Skalski and Townsend (1999) were able to estimate total project survival past the dam. The estimates were unable to differentiate differences between reservoir effects and dam passage effects. The average survival for the spring, summer and fall chinook salmon passing through the John Day Reach ranged from 83 to 89 percent (95% CI = 74% to 100%) with no statistically significant differences detected between the stocks. Steelhead survival was significantly higher than for the chinook stocks in, 1998, with the average project smolt survival approaching 100 percent (95% CI = 97% to 100%).

The relationship between smolt survival and ambient river conditions was investigated using the available, 1998 PIT-tag data. Skalski and Townsend (1999) found no statistical correlation between the five-day average survival probabilities and river operations (spill) or ambient conditions (river flow, temperature and turbidity) in 96 percent of the analyses.

Spring chinook survival in one analysis was found to be positively correlated with turbidity levels in the reservoir.

The mean yearling chinook salmon survival [i.e., 0.826 (SE = 0.030)] through the John Day project in 1998 (Table7) was generally less than mean survival rates reported by the National Marine Fisheries Service (NMFS) for yearling chinook salmon at upper Snake River hydro projects during, 1994-97. Mean steelhead survival [1.043 (SE = 0.038)], however, was higher than mean values reported by NMFS for Snake River projects.

Table 7.
Average Travel Time and Survival Estimates for Various Salmonid Stocks Passing John Day Dam Based on, 1998 PIT Tag Data (Skalski and Townsend, 1999; Attachment E).

Species	Reservoir Travel Time (Days)	Survival McNary Past John Day (%)
Spring Chinook/1998	3.9	83
Summer Chinook/1998	3.7	87
Fall Chinook/1998	5.1	83
Summer Steelhead/1997	2.3	NA
Summer Steelhead/1998	3.9	100

The survival rates for fall subyearling chinook salmon passing through the John Day Project in, 1998 averaged 83 percent (SE = 0.0033). However, these hatchery fall chinook were predominately spring releases from the mid-Columbia and Snake River regions and may not be representative of wild stocks especially from the Snake River. Hatchery fish represent larger, more mature, smolts than wild subyearlings. Hatchery fish may have a tendency to actively migrate.

Insufficient PIT–tag data currently exist to estimate survival rates for wild stocks of salmonids in the John Day project. A careful assessment of smolt outmigration dynamics at John Day project will require additional years of tagging data before mean responses can be characterized and relationships to ambient river conditions can be thoroughly explored. However, based on currently available information from the, 1998 PIT-tag database, smolt survival through the John Day project ranged from 83 to 100 percent depending upon salmonid stock. Smolt survival for any stock was not statistically correlated with river flow.

Although travel time through the John Day Reservoir during, 1998 was correlated with river flow, spill volumes and water temperatures, survival was not found to be statistically correlated with river operations (Spill) or ambient conditions (River Flow, Water Temperature or Turbidity) in 96 percent of the analyses. In one case, spring chinook survival was positively correlated with turbidity levels in the reservoir (Skalski and Townsend, 1999).

6.3 Dam Passage

6.3.1 Passage Routes

Sheer et al. (1995) estimated that 76 percent of the spring chinook passed downstream through the John Day powerhouse either through the juvenile bypass system or the turbines,

and 24 percent passed via the spillway during the, 1995 spring outmigration period. Other seldom used passage routes include the John Day Dam navigation lock and the two fish ladders.

The passage routes taken by outmigrating juveniles and the passage efficiencies are highly variable depending on the daily river discharge and dam operation, especially the volume of spill.

Direct assessments of survival via the various passage routes are generally lacking for various conditions experienced at the John Day Project. Literature-based assessments of similar systems in the Columbia and Snake River systems suggest a wide range of variability exists in the passage and survival estimates that can be used to approximate the losses of juvenile salmonids under the various operational systems. For the purposes of this report the overall, 1998 PIT tag survival data, that does not differentiate survival estimates between passage routes, will be used for comparing alternative management options.

6.4 Total Project Juvenile Salmonid Survival (In-River Survival)

Anderson et al. (1999) [Attachment E] estimated the changes in juvenile survival through the hydro system [lower Snake and Columbia Rivers; from Lower Granite Dam (LGD) to Bonneville Dam (BON)], under various alternatives, including drawdown scenarios at John Day Dam. This effort was consistent with the Plan for Analyzing Testable Hypotheses (PATH) process developed for the Snake River drawdown assessment. The method involved utilizing juvenile salmonid passage models to characterize survival through the hydro system. Anderson et al. (1999) used the Columbia River Smolt Passage (CRiSP) model for the detailed analysis of drawdown actions at the John Day project on fish travel time and smolt survival. The model tracks release groups of juvenile salmonids as they migrate through the hydro system. Mortality is attributed to: 1) direct dam mortality (as fish pass through turbines, bypass systems or spillways of dams), 2) predation (predator temperature response functions, predator consumption rates, and predator abundance indices in the forebays, tailraces and main bodies of the reservoirs), and 3) gas bubble disease resulting from total dissolved gas (TDG) supersaturation. Fish migration rates are modeled in terms of river velocity, date in the season, and length of time in migration and water temperature (Zabel and Anderson, 1997; Zabel et al., 1998). The model was run with and without options for fish collection at several dams and subsequent transportation to below Bonneville Dam. The current levels of transported chinook stocks arriving below Bonneville Dam are shown below in Table8.

Table 8.
Percent of Chinook Stocks Arriving below Bonneville Dam that were Collected Upstream and Transported in Barges Past John Day Dam.

Chinook Stock	Percent of Run Transported	Percent of Run In-River
Snake River		
Spring Chinook	93%	7%
Fall Chinook	97%	3%
Upper Columbia River		
Spring Chinook	0%	100%
Fall Chinook	55%	45%

Three John Day drawdown scenarios, Alternatives 1, 3 and 4, were modeled. The measures of performance for the management actions include: fish travel time (FTT), in-river survival, and total hydro system survival for both in-river and transported smolts.

Because considerable uncertainties exist concerning the effects of drawdown on fish survival and how observed survivals are partitioned throughout the life-cycle; a number of hypotheses are included in the modeling analyses and systematically tested. Results are highly dependent upon the assumptions made and the hypotheses selected as the modeling input parameters. Results reported here for juvenile migrants from Lower Granite Dam to below Bonneville Dam represent an average of results obtained across the full set of modeled hypotheses and assumptions. These results do not consider subsequent potential delayed mortality occurring below Bonneville Dam, which is discussed later.

6.4.1 In-River Travel Time and Survival Results

The mean in-river travel time estimates in days and survival estimates for Snake River spring and fall chinook from Lower Granite Dam to Bonneville Dam are provided in detail in Attachment E and summarized in Table 9 and Table 10, respectively.

Table 9.
Mean Fish Travel Time Estimates (LGD to BON) from CRiSP Modeling Analysis for John Day Drawdown Alternatives (after Anderson et al., 1999; Attachment E).

	Fish Travel Times (Days) Snake River Stocks	
Management Actions	**Spring Chinook**	**Fall Chinook**
Existing Conditions (including transportation)	-	-
Existing Conditions (without transportation)	18.0	24.0
Alternative 1; Drawdown to Spillway Crest	17.0	22.9
Alternative 3; Drawdown to Natural River	15.9	21.5
Alternative 4; Drawdown to Natural River w/ Flood Control	16.4	NA[1]

1) No model run for Alternative 4 was performed for fall chinook, since flood waters are released prior to the onset of the fall chinook migration from the Snake River.

Table 10.
Mean Survival Estimates (LGD to BON) from CRISP Modeling Analysis for John Day Drawdown Alternatives (after Anderson et al., 1999; Attachment E).

	Survival Estimates Snake River Stocks	
Management Actions	**Spring Chinook**	**Fall Chinook**
Existing Conditions (including transportation)	81%	78%
Existing Conditions (without transportation)	44%	28%
Alternative 1; Drawdown to Spillway Crest	47%	30%
Alternative 3; Drawdown to Natural River	50%	31%
Alternative 4; Drawdown to Natural River w/ Flood Control	49%	NA1

Spring chinook salmon travel times are estimated to decline approximately one to two days with drawdown resulting in an estimated mean total survival increase of three to six percent compared to full pool operation. Fall chinook salmon travel times are estimated to decrease

one to four days under the various drawdown alternatives, resulting in an estimated total survival increase between two to three percent compared to existing conditions.

On a relative basis, the mean benefit of drawdown (percent increase in survival rate) under Alternatives 1, 3 and 4 compared to full pool is summarized as in Table11 which follows:

Table 11.
Change in Survival Rate after Drawdown

Change from Full Pool	**Spring Chinook**	**Fall Chinook**
Alternative 1; Drawdown to Spillway Crest	3%	2%
Alternative 3; Drawdown to Natural River	6%	3%
Alternative 4; Drawdown to Natural River w/ Flood Control	5%	NA[1]

1) No model run for Alternative 4 was performed for fall chinook, since flood waters are released prior to the onset of the fall chinook migration from the Snake River.

The estimated mean smolt survival rates, presented above, provided for a direct comparison of survival through the reservoir with and without the benefit of transportation under existing conditions. However, under current hydro system operations, 93 percent of the Snake River spring chinook and 97 percent of the fall chinook arriving below Bonneville Dam are transported.

Snake River chinook are primarily captured and barged or trucked to below Bonneville Dam with higher survivals than fish migrating through the lower Snake and Columbia River hydroelectric system (Marmorek et al., 1998). The present deep-draft barges used for transportation are too deep to navigate the reservoir under the contemplated drawdown conditions (USACE, 1999). If transportation by barge is lost due to John Day drawdown, then survival of juvenile spring chinook smolts from Lower Granite Dam to Bonneville Dam is estimated to decrease dramatically as shown in Table12 below. Since most Snake River fall chinook (90%-95%) are transported by truck between late June and October, changes in their survival with drawdown of John Day Reservoir, only, would be far less than for spring chinook.

Table 12.
Change in Survival from Existing Condition w/ Transportation

	% Decrease in Survival Rate	
	Spring Chinook	**Fall Chinook**
Alternative 1; Drawdown to Spillway Crest	-34%	-2% to –5%
Alternative 3; Drawdown to Natural River	-31%	-2% to –5%
Alternative 4; Drawdown to Natural River w/ Flood Control	-32%	NA[1]

1) No model run for Alternative 4 was performed for fall chinook, since flood waters are released prior to the onset of the fall chinook migration from the Snake River.

6.4.1.1 Reduced Travel Time, Improved In-River Survivals under Current Transportation Operation (Alternatives 1 and 2 only).

The maximum improvement in the in-river survival estimates for Snake River chinook stocks is estimated to be an increase of six percent and three percent for spring chinook and fall chinook salmon, respectively.

6.4.1.2 Reduced Travel Time, Improved In-River Survivals without Transportation (Alternatives 3 and 4 only).

The maximum in-river survival estimates for Snake River chinook stocks are estimated to decrease 34 percent and from 2 to 5 percent under the natural river drawdown scenario for spring chinook and fall chinook salmon, respectively. These changes could represent a decrease of approximately – 1,360,000 spring chinook smolts (0.34 x 100% of LGD run size of 4.0 million smolts) and approximately – 28,000 fall chinook smolts (0.035 x 100% of LGD run size of 0.8 million smolts).

The implications of either increased or reduced in-river survivals on adult returns is explored in the life-cycle analysis section below.

6.4.1.3 Adult Recruitment (Life-Cycle) Estimates.

Results of the juvenile fish passage model were incorporated into life-cycle models to characterize the effect of John Day drawdown alternatives on returning adult population levels (Anderson et al., 1999; Attachment E). Two modeling approaches were used: 1) a Bayesian life-cycle model to assess the probability of complying with NMFS standards for survival and recovery of the stocks (in accordance with PATH, Marmorek et al., 1998; Peters et al., 1999) and 2) a deterministic life-cycle model to assess equilibrium levels of spawning populations and total returns (i.e., spawners plus harvest) under management to obtain maximum sustainable yield (MSY) for each alternative as a means to estimate gains or losses of spawning adults and total returns when comparing two separate management actions.

These models are based on the life cycle of salmon (Figure 5) with generally four important stages; 1) freshwater spawning and production stage (extending from eggs in redds [spawning nests], to the beginning of smolt migration); 2) freshwater smolt migration stage (from Lower Granite Dam to Bonneville Dam); 3) estuarine and oceanic survival stage (Bonneville Dam to Bonneville Dam) and 4) upstream adult migration stage (Bonneville Dam to the spawning grounds). The number of adult spawning recruits (returning adults to the spawning grounds) is expressed as the product of survival at each of the four life stages above with the inclusion for ocean and river harvest mortalities (Refer to Attachment E for details of the models).

A great deal of uncertainty exists over what survivals will be in free-flowing river segments after reservoirs are drawn down to natural river levels. To characterize the range of uncertainty, PATH estimated the upper and lower bounds of survival based on existing PIT-tag data in free-flowing areas of the Snake River (Peters et al., 1999) as follows in Table13.

Table 13. Upper and Lower Bounds of Survival (Based on Existing PIT-tag Data)		
	Survival	
Snake River Stock	**Upper Estimate**	**Lower Estimate**
Spring Chinook	96%	85%
Fall Chinook	90%	61%

The, 1998 PIT-tag data for the impounded John Day Reach suggested a mean survival of 83 percent for both spring and fall chinook (Skalski and Townsend, 1999). Current passage survival estimates in the impounded reach for chinook lie within the range of uncertainty for survival in free-flowing reaches. The potential benefit of altering from impounded to a free-flowing reach then would range for spring chinook between 2 and 13 percent. The maximum benefit would be a seven percent improvement in survival of fall chinook.

Actual survival estimates as modeled under the fish passage model for the John Day Reach are in the range of half these maximum benefit assumptions.

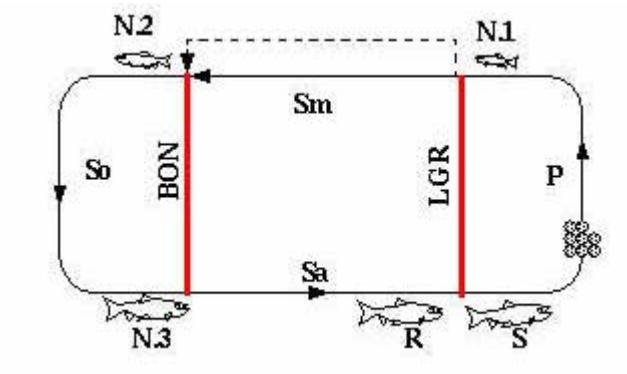


Figure 5. Life cycle of salmon extending from freshwater production stage, P, to hydrosystem survival, Sm, from Lower Granite Dam (LGR) to Bonneville Dam (BON), which includes in river and transport passage, to ocean survival, So, to upriver adult migration survival, Sa. S spawners produce R recruits.

6.4.1.4 Bayesian Life-Cycle Model.

This analysis was conducted with the PATH Bayesian life cycle analysis, which provides probabilities of meeting 24-year survival and 48-year recovery goals (Marmorek et al., 1998). A number of hypotheses were explored on the in-river survival of smolts, the linkages between survival in the freshwater and smolt passage stages and survival in the ocean and the survival of adults migrating up river.

The analysis showed a range of probabilities of survival and recovery depending on the action and hypotheses used as input parameters. However, general trends are apparent in survival and recovery probabilities relative to drawdown alternatives as shown in Table 14.

Table 14. The Estimated Changes in the Range of Survival and Recovery Probabilities with the Addition of John Day Drawdown Actions to other Management Actions in the Snake River (after Anderson et al., 1999).		
Chinook Stock	**Change in Probability of Complying with NMFS 24-yr Survival Standard**	**Change in Probability of Complying with NMFS 48-yr Recovery Standard**
Snake River		
Spring Chinook	-0.2 to 0.5%	2 to 3%
Fall Chinook	-0.1 to 0.0%	0 to 4%

The results indicate the ranges in survival probabilities with the addition of John Day drawdown to natural river are small and do not add measurably to improved probabilities of complying with adult survival or recovery standards for either stock compared to other proposed management actions like enhanced transportation or drawdown of Snake River dams. Under some of the assumptions incorporated into the model, drawdown at John Day could decrease the probability of complying with the survival standard slightly.

A variety of important assumptions drove the results obtained from life-cycle modeling. Among these, assumptions regarding "delayed mortality" to transported Snake River chinook salmon and "extra mortality" in the ocean were the most important.

It has been hypothesized that additional post-transportation mortality occurs below Bonneville Dam to juvenile chinook salmon transported from the Snake River. In the life-cycle modeling analysis performed by PATH, a "D-factor" was used to impart additional "delayed mortality" to transported verses non-transported Snake River migrants. Mechanisms responsible for this differential mortality have not been substantiated, and there is disagreement among biologists as to the magnitude of any delayed mortality that may occur. Assumptions regarding the size of this factor greatly affect results concerning the benefits of Snake River and John Day drawdown. We have used modeling results that maximize the potential benefits associated with John Day drawdown. Actual benefits are likely to be less, and could be far less than those reported in this document.

In addition, three alternative hypotheses concerning "extra mortality" occurring in the ocean were formulated and modeled under the PATH analytical approach. One hypothesis postulated that extra mortality in the ocean was attributable to hydrosystem effects in the Snake River. The estimated benefits presented in this report are based on modeling results obtained under this assumption because they maximize potential benefits from drawdown. Alternative hypotheses attribute extra mortality in the ocean either to ocean environmental conditions or to reduced "fitness" of Snake River chinook as a result of hatchery program influences. Under these alternative hypotheses, benefits from drawdown would be substantially less than those reported.

6.4.1.5 Deterministic Life-Cycle Model.

A deterministic life-cycle model was developed to compare the differences in equilibrium population levels and in total returns (i.e., spawners and harvest) at maximum sustainable yield (MSY) population levels. Four stocks from various subbasins are considered, including spawning populations of Snake River spring chinook, Upper Columbia spring chinook, Snake River fall chinook, and Hanford Reach or Upriver Bright (URB) fall chinook salmon. The results of comparing existing conditions with John Day drawdown alternatives are

provided in Table 15. These results are not intended to represent estimates of the actual numbers of fish that would be produced if drawdown actions were implemented. There are many variables and uncertainties that complicate deriving such estimates. Rather, the modeling results are intended to illustrate the likely relative contribution, in terms of adult fish production, of alternative drawdown actions. Ranges of responses are given as a means of addressing the substantial range of assumptions that was used in the modeling analysis. The results are highly variable depending upon the assumptions used including high and low 1) smolt survival rates, 2) D levels (i.e., differential mortality between transported and non-transported juvenile migrants related to assumptions regarding the effects of transportation on fish survival), 3) extra mortality factors (i.e., assumptions regarding effects of hatchery-related bacterial kidney disease [BKD], ocean climate regime shifts, or cumulative hydro-system stresses), and 4) adult survival rates.

The results suggest that alternatives for John Day drawdown (without additional benefits from Snake River drawdown) would most likely decrease equilibrium spawning population sizes of Snake River spring chinook and Hanford Reach URB fall chinook, while slightly increasing Upper Columbia spring chinook and perhaps Snake River fall chinook compared to existing conditions. The modeling results range widely depending upon the assumptions of hypotheses tested (refer to Attachment E for a detailed analysis). Maximum benefits for Snake River stocks occur when it is assumed that there are little or no benefits from transportation of juveniles, especially Snake River fall chinook salmon, to below Bonneville Dam.

6.4.1.6 Probability of Improved Juvenile Survival and Adult Return Benefits with Drawdown Alternatives

In summary, the effects of John Day drawdown may change in-river migrant juvenile survival rates for Snake River spring and fall chinook by three to six percent and by one to two percent, respectively, depending upon the drawdown alternative. Comparative modeled changes among drawdown alternatives in numbers of adult spawners (i.e., Equilibrium Population Level) and in numbers of returns at maximum sustainable yield (Total Returns at MSY) are presented in Table 15. Differences amongalternatives with respect to equilibrium spawning population levels for Upper Columbia spring chinook could range from as much as 101 to 184 spawners (a 10 to 17 percent increase) under the various drawdown alternatives (Table 15). Relative changes in natural production of the Hanford Reach URB fall chinook stock located above McNary Dam, however, represent a decrease by 11 to 13 percent (16,876 to 19,315 spawners). Similarly, Snake River spring chinook equilibrium spawning populations could decrease by as much as 7,980 under Alternative 3 (John Day drawdown to natural river without flood control), while Snake river Fall chinook equilibrium spawning populations could increase by as much as 6,179 adults, assuming that transportation effectiveness is low. Under this assumption, however, benefits from termination of transportation alone (i.e., without John Day drawdown) would result in a fall chinook equilibrium spawner population level increase of 5,631 fall chinook adults. The resulting net benefit for snake river fall chinook from John Day drawdown, aside from benefits associated with termination of transportation, is an increase of only 548 or fewer fall chinook spawners. Mortality and injury associated with turbine passage and gas supersaturation at John Day Dam would be eliminated under the natural river level drawdown option (Alternative 3). This improvement is one of several contributing to the results of the analysis presented above.

Table 15. **Comparisons of Estimated Equilibrium Spawning Population Levels under Drawdown with Current Conditions in the Snake and Columbia River Systems (after the Deterministic Life-cycle Model of Anderson et al., 1999).**			
Equilibrium Spawning Population of various Stocks	**Alternative 1, John Day drawdown to spillway crest**	**Alternative 3, John Day drawdown to natural river**	**Alternative 4, Drawdown to natural river with flood control**
Snake River			
Spring Chinook **(Index Pop. = 15,942 spawners)**	-14,022 (-88%) to –9,918 (-62%)	-12,084 (76%) to -7,980 (-50%)	-12,730 (-80%) to –8,626 (-54%)
Fall Chinook **(Index Pop. = 7,259 spawners)**	-2,912 (-40%) to 5,972 (82%)	-2,793 (-38%) to 6,179 (85%)	-2,793 (-38%) to 6,179 (85%)
Columbia River			
Spring Chinook **(Index Pop. = 1,061 spawners)**	101 (9.5%)	184 (17%)	143 (13%)
Fall Chinook **(Index Pop. =132,500 spawners)**	-19,315 (-15%)	-16,876 (-13%)	-16,876 (-13%)

Conversely, under assumptions of high transportation effectiveness, John Day drawdown actions would be detrimental to these spawning populations. Regardless, Bayesian life-cycle assessment and PATH analysis indicated that small levels of benefit would result from John Day drawdown for Snake River spring and fall chinook with respect to the probability of realizing 24-year survival and 48-year recovery standards discussed in the, 1995 NMFS Biological Opinion on Federal Hydropower System Operation (**Table7**).

Table 16.
Fisheries Benefit Summary (Alternatives 1 and 2: John Day Drawdown to Spillway Crest with or without Flood Control).[1, 8]

	Snake River		Columbia River	
Standard	**Sp/Su Chinook**	**Fall Chinook**	**Fall Chinook**	**Spring Chinook**
With Snake River Drawdown				
Decreased Travel Time[2]	4.0	6.0	NA	NA
Juvenile Survival Rate[3]	14%	5%	NA	1%
Probability of Survival[4]	NA	NA	NA	NA
Probability of Recovery[5]	NA	NA	NA	NA
Equilibrium Pop. Level[6,7]	17,860	10,894	-19,315	101
Total Returns at MSY[6,7]	16,454	73,654	-64,178	88
	Snake River		**Columbia River**	
Without Snake River Drawdown				
Decreased Travel Time[2]	1.1	1.0	NA	NA
Juvenile Survival Rate[3]	3%	2%	NA	1%
Probability of Survival[4]	NA	NA	NA	NA
Probability of Recovery[5]	NA	NA	NA	NA
Equilibrium Pop. Level[6,7]	-9,918	5,972	-19,315	101
Total Returns at MSY[6,7]	-5,396	16,877	-64,178	88

1. Based on the most optimistic results regarding potential benefits derived from life-cycle modeling. Actual benefits may be less.
2. Change in days from current conditions for juvenile travel time from above Lower Granite Dam to below Bonneville Dam.
3. Change in percentage points from current survival conditions for in-river (i.e., non-transported) migrants from above Lower Granite Dam to below Bonneville Dam.
4. Change in percentage points of annual spawning escapements projected to be above a predetermined threshold of minimum (Survival Standard) abundance. The PATH process included analysis across both short-term (25-year) and long-term (100-year) time frames. The threshold values used were 150 and 300 spawners, depending on the particular index stock and stream modeled. PATH analyses compared modeling results to an informal survival standard set at 70% of all modeled runs, reflecting a “high likelihood” of spawner escapement being over the threshold. These criteria were used in the development of the, 1995 NMFS Biological Opinion for Columbia River Federal Hydropower System Operation.
5. Change in percentage points of life-cycle simulation model runs for which the average resulting spawner abundance over the last 8 years of a 48-year model run was greater than a predefined Recovery Standard level of abundance. For Snake River fall chinook, the level used was 2,500 spawners. For Snake River spring/summer chinook, the level was set at 60% of the pre-1971 average spawner count for each index stock. These criteria were based upon the, 1995 NMFS Biological Opinion for Columbia River Federal Hydropower System Operation.
6. Change in number of fish from current conditions assuming poorest differential survival (D) for transported juveniles in comparison to in-river migrants (D = 0.65 for Snake River spring/summer chinook, 0.05 for Snake River fall chinook, and 1.00 for Upriver Bright [URB] fall chinook salmon). Equilibrium Population (Pop.) Level (EPL) is at maximum sustainable spawner escapement (i.e., under no-harvest conditions). Total Returns are the combination of spawners and harvest at maximum sustainable yield (MSY) population levels.
7. Under the assumption of poorest D for transported juvenile fish (Item 6), termination of Snake River fall chinook transportation alone (i.e., without drawdown of any dams) would result in benefits of 5,631 EPL spawners and 15,054 total returns.
8. NA = Not analyzed or applicable.

Table 17.
Fisheries Benefit Summary (Alternative 3: John Day Drawdown to Natural River Channel without Flood Control).[1, 8]

	Snake River		Columbia River	
Standard	**Sp/Su Chinook**	**Fall Chinook**	**Fall Chinook**	**Spring Chinook**
With Snake River Drawdown				
Decreased Travel Time[2]	7.4	11.0	**NA**	**NA**
Juvenile Survival Rate[3]	27%	10%	NA	2%
Probability of Survival[4]	4%	1%	NA	NA
Probability of Recovery[5]	28%	33%	NA	NA
Equilibrium Pop. Level[6,7]	19,798	11,024	-16,876	184
Total Returns at MSY[6,7]	19,076	76,406	-57,322	165
Without Snake River Drawdown				
Decreased Travel Time[2]	2.1	2.4	NA	NA
Juvenile Survival Rate[3]	6%	2.0%	NA	2%
Probability of Survival[4]	0.50%	0.00%	NA	NA
Probability of Recovery[5]	3%	4%	NA	NA
Equilibrium Pop. Level[6,7]	-7,980	6,179	-16,876	184
Total Returns at MSY[6,7]	-4,484	18,058	-57,322	165

1. Based on the most optimistic results regarding potential benefits derived from life-cycle modeling. Actual benefits may be less.
2. Change in days from current conditions for juvenile travel time from above Lower Granite Dam to below Bonneville Dam.
3. Change in percentage points from current survival conditions for in-river (i.e., non-transported) migrants from above Lower Granite Dam to below Bonneville Dam.
4. Change in percentage points of annual spawning escapements projected to be above a predetermined threshold of minimum (Survival Standard) abundance. The PATH process included analysis across both short-term (25-year) and long-term (100-year) time frames. The threshold values used were 150 and 300 spawners, depending on the particular index stock and stream modeled. PATH analyses compared modeling results to an informal survival standard set at 70% of all modeled runs, reflecting a “high likelihood” of spawner escapement being over the threshold. These criteria were used in the development of the, 1995 NMFS Biological Opinion for Columbia River Federal Hydropower System Operation.
5. Change in percentage points of life-cycle simulation model runs for which the average resulting spawner abundance over the last 8 years of a 48-year model run was greater than a predefined Recovery Standard level of abundance. For Snake River fall chinook, the level used was 2,500 spawners. For Snake River spring/summer chinook, the level was set at 60% of the pre-1971 average spawner count for each index stock. These criteria were based upon the, 1995 NMFS Biological Opinion for Columbia River Federal Hydropower System Operation.
6. Change in number of fish from current conditions assuming poorest differential survival (D) for transported juveniles in comparison to in-river migrants (D = 0.65 for Snake River spring/summer chinook, 0.05 for Snake River fall chinook, and 1.00 for Upriver Bright [URB] fall chinook salmon). Equilibrium Population (Pop.) Level (EPL) is at maximum sustainable spawner escapement (i.e., under no-harvest conditions). Total Returns are the combination of spawners and harvest at maximum sustainable yield (MSY) population levels.
7. Under the assumption of poorest D for transported juvenile fish (Item 6), termination of Snake River fall chinook transportation alone (i.e., without drawdown of any dams) would result in benefits of 5,631 EPL spawners and 15,054 total returns.
8. NA = Not analyzed or applicable.

Table 18.
Fisheries Benefit Summary (Alternative 4: John Day Drawdown to Natural River channel with Flood Control).[1, 8]

	Snake River		Columbia River	
Standard	**Sp/Su Chinook**	**Fall Chinook**	**Fall Chinook**	**Spring Chinook**
With Snake River Drawdown				
Decreased Travel Time[2]	NA	NA	NA	NA
Juvenile Survival Rate[3]	NA	NA	NA	NA
Probability of Survival[4]	NA	NA	NA	NA
Probability of Recovery[5]	NA	NA	NA	NA
Equilibrium Pop. Level[6,7]	19,190	11,024	-16,876	143
Total Returns at MSY[6,7]	18,202	76,406	-57,322	126
Without Snake River Drawdown				
Decreased Travel Time[2]	NA	NA	NA	NA
Juvenile Survival Rate[3]	NA	NA	NA	NA
Probability of Survival[4]	NA	NA	NA	NA
Probability of Recovery[5]	NA	NA	NA	NA
Equilibrium Pop. Level[6,7]	-8,626	6,179	-16,876	143
Total Returns at MSY[6,7]	-4,750	18,058	-57,322	126

1. Based on the most optimistic results regarding potential benefits derived from life-cycle modeling. Actual benefits may be less.
2. Change in days from current conditions for juvenile travel time from above Lower Granite Dam to below Bonneville Dam.
3. Change in percentage points from current survival conditions for in-river (i.e., non-transported) migrants from above Lower Granite Dam to below Bonneville Dam.
4. Change in percentage points of annual spawning escapements projected to be above a predetermined threshold of minimum (Survival Standard) abundance. The PATH process included analysis across both short-term (25-year) and long-term (100-year) time frames. The threshold values used were 150 and 300 spawners, depending on the particular index stock and stream modeled. PATH analyses compared modeling results to an informal survival standard set at 70% of all modeled runs, reflecting a “high likelihood” of spawner escapement being over the threshold. These criteria were used in the development of the, 1995 NMFS Biological Opinion for Columbia River Federal Hydropower System Operation.
5. Change in percentage points of life-cycle simulation model runs for which the average resulting spawner abundance over the last 8 years of a 48-year model run was greater than a predefined Recovery Standard level of abundance. For Snake River fall chinook, the level used was 2,500 spawners. For Snake River spring/summer chinook, the level was set at 60% of the pre-1971 average spawner count for each index stock. These criteria were based upon the, 1995 NMFS Biological Opinion for Columbia River Federal Hydropower System Operation.
6. Change in number of fish from current conditions assuming poorest differential survival (D) for transported juveniles in comparison to in-river migrants (D = 0.65 for Snake River spring/summer chinook, 0.05 for Snake River fall chinook, and 1.00 for Upriver Bright [URB] fall chinook salmon). Equilibrium Population (Pop.) Level (EPL) is at maximum sustainable spawner escapement (i.e., under no-harvest conditions). Total Returns are the combination of spawners and harvest at maximum sustainable yield (MSY) population levels.
7. Under the assumption of poorest D for transported juvenile fish (Item 6), termination of Snake River fall chinook transportation alone (i.e., without drawdown of any dams) would result in benefits of 5,631 EPL spawners and 15,054 total returns.
8. NA = Not analyzed or applicable.

SECTION 7. Potential Effects on Spawning Adult Salmonids

The objective of this section is to review the effects of various drawdown scenarios on spawning salmonid fishes by evaluating: 1) the species known to historically and currently use the John Day Reach for spawning, 2) the timing and duration of reservoir spawning, 3) estimating the potential change in the quantity and quality of spawning habitat conditions and 4) estimating the increased capacity of the reach to produce salmonid fishes under the four alternative operating scenarios.

Adult fish passage and survival past John Day Dam would improve under most environmental conditions with drawdown to natural channel level. However, under all John Day drawdown alternatives as designed, adult fish migration could be terminated at flows exceeding the 10-year event of 515,000 cfs. Since completion of the Columbia River Basin hydropower system in the 1960's, flows at John Day Dam have exceeded this level only twice, in 1974 and in 1997. Flows above the 10-year level occurred in May and June, and had a duration of from approximately one to three weeks. Adult passage curtailment during this time frame could have impacted late runs of spring and early runs of summer chinook salmon, and perhaps very early runs of sockeye salmon. Earlier or later portions of runs would not have been affected.

Elimination of John Day Dam under Alternative 3 would also eliminate fallback of adult salmon and steelhead that currently occurs at the dam. Bjornn (1999) found that the fallback rate of John Day Dam for adult migrants ranged 12 percent to 15 percent during periods of spill to aid juvenile migrant passage. Approximately three to five percent of those fish that fall back to below John Day Dam after passing it do not reascend ladders and pass the dam.

7.1 Pre-Project Conditions

Prior to impoundment, the John Day Reach of the mainstem Columbia River supported considerable fall chinook salmon spawning (US ACE, 1951; Fulton, 1968; Sheer, 1999). Fulton (1968) estimated an annual spawning population of 34,000 fall chinook adults using the reach between John Day and McNary dams from, 1957 to, 1960. Equally distributed, this population represents 447 fish per mile. It is likely both upriver brights and tules spawned in the area of the John Day Reservoir (Smith, 1966). Dauble and Watson (1997) estimated the adult spawning escapement to the Hanford Reach was 25,000 fish between, 1964 and, 1982. On a macro-habitat scale, these estimates provide comparable number of fish per mile using habitat in both mainstem reaches during the, 1960s.

Based on modeling and criteria for suitable spawning habitat in the Hanford Reach, Sheer (1999) estimated the potential spawning habitat in the John Day Reach, prior to inundation. As much as 53 percent of the area studied may have been suitable for spawning. Extrapolation to the entire John Day Reach with the aid of GIS techniques, resulted in an estimate of approximately 9,700 acres of potential habitat.

There was no historic evidence of other anadromous salmonid species or stocks utilizing mainstem reaches in the lower Columbia River for spawning. Life history characteristics for other salmonid species suggest this reach was historically used as a migration corridor to upstream tributary and river spawning areas (Fulton, 1968;, 1970; Sheer, 1999).

7.2 Project Construction – Spawning Habitat Modification

Construction on John Day Dam began in, 1958. River flow was impounded in, 1967, and the reservoir reached operating pool level in, 1968. Losses in spawning habitat due to construction and operation of John Day Dam are mitigated with annual hatchery production. The Spring Creek Hatchery, located on the Washington shore of the Columbia River in the Bonneville Dam pool operated by the U.S. Fish and Wildlife Service, and Bonneville Hatchery, located on the Oregon shore immediately below Bonneville Dam and operated by the Oregon Department of Fish and Wildlife, made up the bulk of the production. In recent years, the Little White Salmon Hatchery, also in Washington, has been operated as a unit with the Spring Creek Hatchery. Little White Salmon Hatchery produces the upriver bright fall chinook salmon for the John Day modification and Spring Creek Hatchery produces tule (lower river) fall chinook salmon. The agreement for operating the Spring Creek National Fish went into effect in, 1972 and the agreement for operating the Bonneville Salmon Hatchery went into effect in, 1978.

The Corp's original estimate of loss and modification agreement are based on 30,000 fall chinook salmon spawners derived by taking the difference between the fish counts at The Dalles and McNary dams for the years, 1957-1964 and adding an additional 20 percent safety factor. At first, the fisheries agencies reared tule fall chinook salmon for John Day modification. Currently, upriver bright fall chinook are being reared at the Ringgold Hatchery and other facilities for release in the Hanford Reach of the mid-Columbia River. This is done to provide an upriver fishery for the tribes. Releases of juvenile fish from the various facilities managed under John Day Reservoir modification currently represent approximately 11.9 million smolts annually. These releases are approximately four times greater than the anticipated smolt yield from 30,000 adult fall chinook spawning naturally.

7.3 Current Conditions

Researchers have found that fall chinook salmon spawn in certain deepwater habitat in large rivers and reservoirs below dams (Meekin, 1967; Parametric, 1979; Giorgi, 1992; Dauble et al., 1994; Groves, 1994; Garcia et al., 1994). Spawning activity for other anadromous salmonids in deep water areas has not been documented, and in unlikely based on life-history characteristics of the various other species.

Fall chinook salmon spawning generally occurs annually from mid- to late October through November (Watson, 1975, 1976; Parametric, 1979; Chapman et al., 1983; Chapman et al., 1994a; Dauble and Watson, 1997; Garcia et al., 1994). Areas generally less than 10 m (30 feet) in depth, with substantial groundwater upwelling in Columbia and Snake River reservoirs below dams may provide suitable fall chinook spawning conditions. Spawning has been documented primarily in tailrace areas (Meekin, 1967, Giorgi, 1992; Hillman and Miller, 1994, Chapman et al., 1994a; Backfired, 1994; Garcia et al., 1994; Dauble et al., 1994; Stuehrenberg et al., 1995). The presence of redds (nests) downstream of certain Columbia and Snake river dams indicates that those tailrace environments can provide suitable spawning habitat for fall chinook salmon (Dauble et al., 1994). Fall chinook salmon spawning at lower Snake River dams frequently occurs in the vicinity of the juvenile bypass system outfall (Garcia et al., 1994; Dauble et al., 1994).

Spawning is suspected to occur in the upper John Day Reservoir and McNary tailrace, but documentation of the occurrence of redds has not occurred to date. Tailrace spawning surveys are being conducted in some areas by ODFW and WDFW.

7.3.1 Spawning Habitat Quality and Quantity Below McNary Dam

7.3.1.1 Habitat Quality.

Habitat quality typically refers to the capacity of the available spawning substrate to successfully produce alevin and fry. This capacity can be influenced by: 1) the level of fine sediment that may deposit on or within excavated redds, thereby reducing interstitial water flow and oxygen to embryos and the ability to carry away metabolic waste products; 2) the potential for deep scour of the substrate, thereby washing away incubating embryos, 3) the potential for water level fluctuations to expose and dewater redds constructed at shallow reservoir locations and 4) the initial suitability of physical attributes like spawning depths, velocities, substrate sizes, intergravel flow and water temperatures.

No data exist with respect to site-specific substrate conditions at redd locations in the reservoir. Gilbreath et al. (1999) sampled the sediment conditions in the littoral zones around Blalock Islands. Their data indicate most of the sediment in the broad flat nearshore sections of upper reservoir at water depths between 1 and 5 m (3 and 16 ft) would be classified as silty sand (Attachment B). The median grain size of samples varied from 0.1 to 0.2 mm (0.004 to 0.008 in.). The largest median grain size (D_{50}s) sampled was 0.34 mm (0.01 in). These grain sizes are not in the range of preferred fall chinook salmon spawning habitat (dominant substrate, 25 to 150 mm; 1 to 6-in. gravel; after Connor et al., 1993; Raleigh and Miller, 1986). The amount of clay and silt size fractions, that can hinder spawning success, ranged from 11 to 31 percent of the total reported in Gilbreath et al. (1999) samples. Good substrate conditions for spawning are typically reported in the literature to consist of fine sediment levels less than 12 percent [size fractions < 0.85 mm](Chapman, 1988; Tappel and Bjornn, 1987, WFPB, 1997). It can be concluded that the shallow nearshore areas of the Blalock Islands are not presently conducive to spawning fall chinook salmon.

In Attachment C, Bennett (1999) reports gravel and cobble sediment types made up nearly 40 percent of the substrate composition that is submerged under normal reservoir operations in the upper pool between RM 253 and 291 (McNary Dam). Thus, offshore waters, toward the thalweg of the channel may presently support appropriate sized and quality of gravels for spawning. Sheer (1999) and USGS (1999) habitat modeling results indicate most of the suitable spawning habitat presently occurs mid-channel, in the area of the historic riverbed. No data are available to evaluate the existing habitat for level of fines or for estimating the scour potential. It is assumed the habitat quantity reduction described below includes the consideration any habitat quality issues.

7.3.1.2 Habitat Quantity.

There have been no peer-reviewed studies quantifying the amount of spawning habitat potentially available in the John Day Reservoir. However, the USGS, Western Fisheries Research Center, Columbia River Research Laboratory, provided an analysis of potential suitable spawning habitat conditions as part of their digital data analysis (USGS, 1999) for the reservoir. The amount of potential spawning habitat was estimated based solely on

habitat selection criteria for water depths and velocities. Because substrate information was not available for all conditions tested, this parameter was not used in the current model.

Habitat was classified as potentially suitable for spawning where the mean water column velocity was greater than, or equal to, 0.4 m/s (1.3 fps), and less than, or equal to, 2.0 m/s (6.6 fps), and where the water depth was between 0.3 to 9.0 m (1 and 30 feet). Selected criteria were based on habitat descriptions for fall chinook salmon spawning upstream in the Hanford Reach of the mid-Columbia River (Chapman et al., 1983; Swan et al., 1988; and Swan, 1989).

The modeling results indicate approximately 450 ha (1,113 acres) of potential spawning habitat are available under current conditions that would support spawning below McNary Dam during river flows of 100 kcfs (the approximate 90 percent flow exceedence rate during the fall chinook spawning season). This amount represents less than 5 percent of the wetted area between RM 261 and 292 in the John Day Reach. However, most of the currently available potential spawning habitat is located in the uppermost 16 km (10 miles) of the reservoir between Irrigon (RM 282) and the McNary tailrace (RM 292) as shown in Figure 6 Sheer (1999) describes a portion of the reservoir area upstream of RM 284 as the only riffle-run habitat currently occurring in the John Day Reach.

Connor et al. (1994), Garcia et al. (1994) and Geist and Dauble (1998) indicate that only a small fraction of the available habitat usually supports spawning. In one case, after reductions for lateral channel slope, proper substrate conditions and scour potential, Connor et al. (1994) found the useable habitat area (Total Effective Area; TEA) for successful spawning, was only 3.8 percent of the total habitat area modeled under an Instream Flow Incremental Methodology (IFIM) study in the Clearwater River system in Idaho (Arnsberg et al., 1992). IFIM techniques use a similar criteria method to define suitable spawning habitat conditions as the DSS (USGS, 1999). The conservative application of Connor et al.'s (1994) TEA of 3.8 percent to the USGS model output results would provide an estimate of 17 hectares (42 acres) of current potentially effective spawning habitat in the John Day Reservoir.

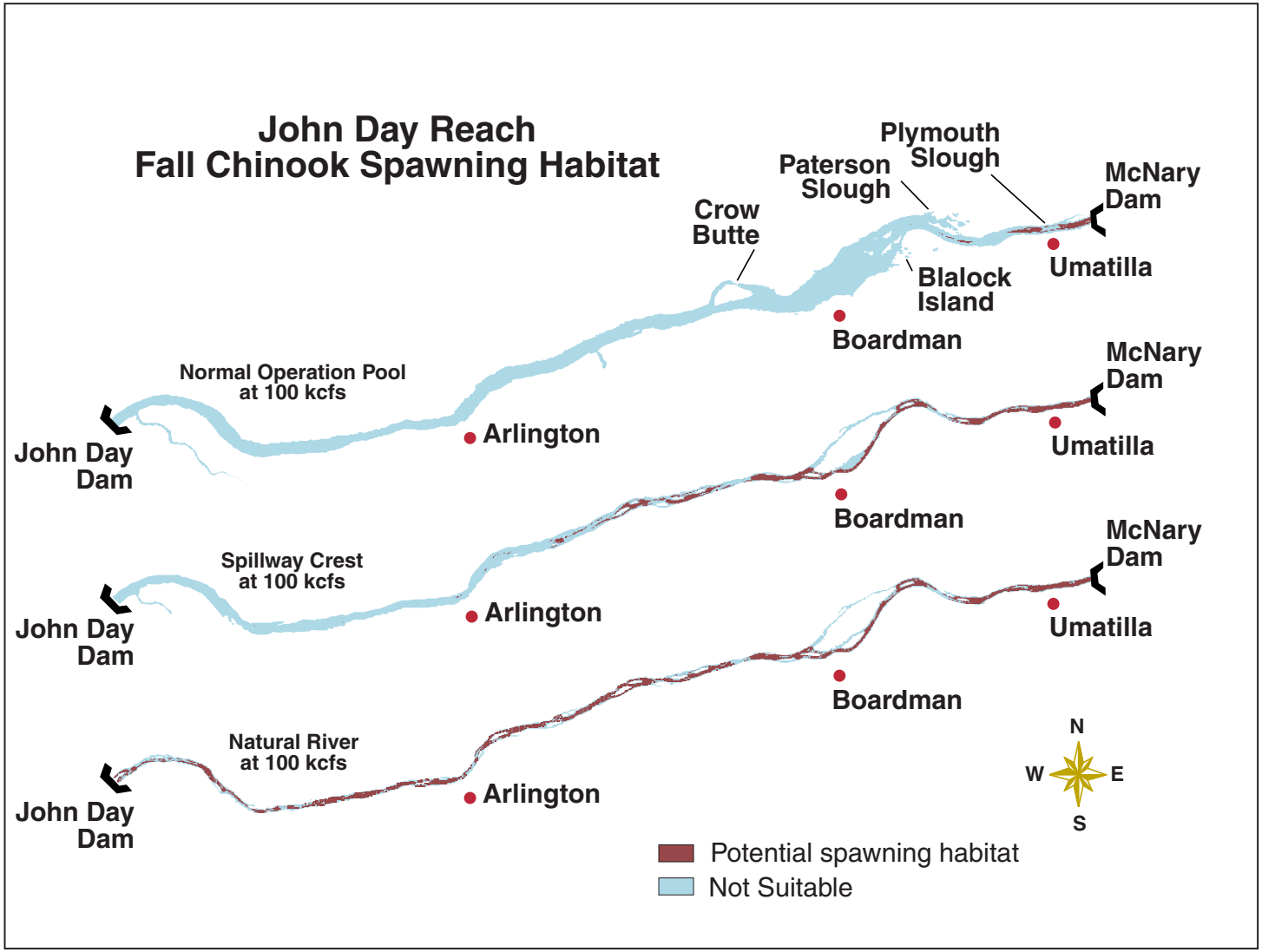


Figure 6. John Day Reach - Fall Chinook Spawning Habitat (from USGS 1999).

7.3.2 Estimated Fish Abundance

Estimating the abundance of spawners based on the level of suitable physical habitat in a river reach is an inherently difficult and questionable task. The accuracy of any such approximation is undoubtedly poor and results reported in the literature are highly variable. A large part of the variability in habitat and extant production estimates may simply lie in the fact that the nuances of habitat conditions influencing adult spawning have not been adequately described. Geist and Dauble (1998) present a credible hypothesis of hyporheic (intergravel) flow and complex habitat structure that offers upwelling and downwelling currents, are critical factors spawners naturally select on a micro-habitat level. Since these parameters are not easy to measure, it remains difficult to predict ultimate habitat use of otherwise suitable channel conditions for spawning.

Nevertheless, for this assessment, estimating actual abundance or production is not critical. The relative differences of the various drawdown scenarios can be assessed, regardless of the inherent weaknesses in the estimates, if all data extrapolations are treated in a consistent manner. It is critical to note, any reference to numbers provided in the following section are presented for a relative comparison between the alternative drawdown scenarios. The reader is advised not to assume the numbers are absolute. Use of these numbers outside the expressly intended purpose stated herein is inappropriate.

Reservoir spawning is only assumed to occur for fall chinook salmon as described below. Similarly, drawdown to spill crest or natural river levels is also assumed to only benefit the fall chinook race that extensively uses large mainstem rivers for spawning.

7.3.2.1 Fall Chinook Salmon.

Welsh (1983) offers a technique of estimating redd densities by dividing the area of available spawning habitat with the average size of a redd. Swan (1989) concluded this approach does not allow for spacing between the redds and modified Welsh's method by using observed redd densities within the boundaries of suitable habitat. Their research from the Hanford Reach indicates redd spacing is typically on the order of one fall chinook salmon redd per 62.3 m^2 (670.2 ft^2) or 65 redds per acre of useable habitat over large areas (Swan, 1989). Extremely intense spawning has been reported at a mean density of 280 redds per acre (Swan, 1989). Swan (1989) also concluded the Hanford Reach fall chinook salmon population supported a male to female ratio of 1:1. Thus, it is assumed each female completes one redd and each redd corresponds to a count of 2.0 adult fish. At full seeding capacity, the 17 ha (42 acres) of effective spawning habitat in the upper John Day Reservoir is anticipated to support 2,750 redds or approximately 5,500 spawners. For the upper 10-mile section of reservoir this level of seeding would equate to 275 redds/mile. This approximation is consistent with other redd density estimates reported in the Columbia and Snake River basins.

7.3.3 Estimated Benefits under Drawdown

The USGS (1999) used modeling to estimate potential fall chinook salmon spawning habitat quantities below McNary Dam for two drawdown scenarios; drawdown to spillway crest and

to natural river level. Both scenarios represent conditions without flood control. The comparative results are shown in Table 19.

The potential benefits estimated in Table 19 are based on three important assumptions. We assumed that spawning, rather than rearing or some other life-history component, was limiting fall chinook production under both current and future, drawdown conditions. However, excess spawning habitat exists in most years in the Hanford Reach above McNary Dam under current conditions. We assumed that all potential spawning habitat would be restored to a usable condition under drawdown conditions. However, compaction of gravel and continuing attenuation of peak flows following drawdown could prevent or limit habitat recovery. Finally, we assumed that the potential spawning habitat (under both current and drawdown conditions) can, and would, become fully seeded with spawners. However, there is no documentation of the use of existing potential spawning habitat currently available in the John Day Reach below McNary Dam even in years of strong adult fish returns.

Table 19.
Potential Spawning Habitat and Estimates of Potential Seeding Capacity in the John Day Reach under Three Drawdown Scenarios.

	Potential Spawning Habitat (Ac)[1]	Total Effective Spawning Area (TEA = acres)[2]	Redd Capacity (#)[3]	Spawner Capacity (#)[4]
Existing Condition	1,113	42	2,750	5,500
Drawdown to Spill Crest	6,296	239	15,550	31,100
Natural River	11,170	424	27,500	55,000

1) After Digital Atlas reservoir model using water depth and velocity criteria only; USGS (1999)

2) Assumed TEA = 3.8% of available area (after Connor et al., 1994)

3) Assumed redd spacing of 65 redds/acre (after Swan, 1989)

4) Assumed one redd per female and a sex ratio of 1:1 males to females [~2.0 fish/redd] (Swan, 1989)

Relative change in potential spawning capacity for the four drawdown alternatives evaluated is shown in Table 20:

Table 20.
Relative Changes in Potential Spawning Capacity

Scenario	Factor
Existing Conditions	1.0
Alternative 1 – Spill Way without Flood Control	5.7x
Alternative 2 – Spill Way with Flood Control	5.7x
Alternative 3 – Full Breach without Flood Control	10.0x
Alternative 4 – Full Breach with Flood Control	10.0x

These values are comparable to the increase in length of river available for spawning under each drawdown scenario. The extent of backwater conditions under all four drawdown scenarios as estimated by the Corps of Engineers (1999) are shown in Table 21.

Table 21. Extent of Backwater in the John Day Reservoir under Various Drawdown Scenarios at a River Discharge of 100 kcfs		
Drawdown Scenario	**Backwater Extent (RM)**	**Length of Available Spawning Habitat (miles)**
Existing Conditions	291.3	10
Spillway without Flood Control	269.0	33
Spillway with Flood Control[1]		
Natural River without Flood Control	216.0	76
Natural River with Flood Control[1]		
1) At 100 kcfs river flow the flood control scenarios operate at the same reservoir pool elevations as the alternatives without flood control.		

Modeling estimates suggested, therefore, that potential fall chinook salmon spawning could increase by an order of magnitude below McNary Dam for the natural river alternative compared to existing conditions (11,170 acres compared to 1,113 acres). Most of this increase is a result of greater lineal distance available for potential spawning. However the amount of potentially suitable habitat along the reach distance increases approximately 32 percent as well (USGS, 1999).

At the estimated capacity level of 55,000 spawners and 27,500 redds, an average equally distributed redd density rate of 362 redds per mile would occur in the John Day Reach. This rate appears reasonable with respect to production levels observed in the Hanford Reach. The 90 km (56 mile) unimpounded Hanford Reach supports an adult escapement of 40 to 80 percent of the fall chinook salmon passing McNary Dam (Rogers and Hilborn, 1988; Swan, 1989), or approximately 12,000 to 80,000 fish. At 2.0 fish/redd the Hanford Reach would produce between 110 and 720 redds per mile. Dauble and Watson (1997) indicated the current adult escapement to the Hanford Reach is averaging around 48,000 fish, which would equate to an estimated 430 redds per mile. Although the accuracy of aerial redd counts is limited, Dauble and Watson (1997) report a peak count of 8,000 redds (or 145 redds/mile) over the long-term record in the Hanford Reach.

It is possible that natural river drawdown will be able to achieve the spawning capacity assumed to occur prior to reservoir inundation (Fulton, 1968; Sheer, 1999). As the upper extent of the possible benefit under the natural river option, Sheer's (1999) estimate of historic spawning habitat, extrapolated to the entire length of the reach, results in approximately 9,700 acres or 87 percent of our estimate based on the USGS (1999) modeling of potential habitat. Treated in the same fashion as our estimate, the 9,700 acres would represent 369 acres of total potentially effective spawning habitat (TEA) at a redd density of 65 redds per acre for an estimated maximum potential spawning population of 48,000 adults. This extrapolation is approximately 40 percent higher than the population level of 34,000 adult fall chinook salmon estimated by Fulton (1968).

The USGS (1999) modeling results estimate a nearly six-fold increase in potential spawning habitat below McNary Dam for fall chinook with a drawdown to spill crest (Alternative 1). This scenario provides approximately 55 percent of the benefit of the natural river alternative

(Alternative 3). It results in a redd distribution rate spread evenly over the 33 miles of available potential spawning habitat of 472 redds per mile and a potential maximum spawning population of nearly 31,100 fall chinook adults.

Flood control operations with either scenario (Alternatives 2 and 4) would not influence the level of potential spawning or incubation. Flood control measures are likely to be implemented during winter and spring high flow events when river discharge may exceed 515 kcfs (USACE, 1999). Flows in exceedence of 200 kcfs are extremely rare during the fall chinook spawning period (USGS gauging records at the Dalles Station; flow exceedence curves for the months of October and November). Peak floods usually occur mid-May to mid-June, annually. Similarly, the primary period of fry emergence occurring in March and April would also generally not be affected by flood events. Thus, successful fry production from redds developed in the fall should not be influenced by Alternatives 2 and 4.

An undefined time period, following altered river hydraulics, will be needed to restore the river bed conditions in the John Day Reach to acceptable spawning habitat quality. Whereas, the river will cut through the soft sediments relatively quickly and create the surface layer of appropriate spawning gravels, it may take a large flood to scour the streambed to sufficient depth (approximately 0.3 to 0.4 m; 12 to 16 inches) to clear embedded fine materials. Alternatively, recruitment of new gravel may bury embedded substrate areas under high flow conditions. In addition, a biological lag of many generations may occur at current commercial fish harvest rates prior to achieving spawning levels approaching full capacity.

7.3.4 Potential Change in Harvest Benefits from Restored Natural Production Below McNary Dam

Assuming a smolt yield for fall chinook salmon of 100 smolts/spawner (after Chapman et al., 1994), the potential increase in spawner levels below McNary Dam in a fully-seeded John Day Reach could produce approximately 2.56 million and 4.95 million more naturally produced smolts under Alternatives 1 and 3, respectively, than under the current habitat conditions. However, as a result of improved egg-to-smolt survivals from artificial propagation facilities compared to natural production, the smolt production under current mitigation hatchery programs exceeds the natural river alternative by more than two-fold (Table 22).

Adult returns to the mouth of the Columbia River could reach up to 3.4 percent of the John Day smolt yield arriving below Bonneville dam, under good estuarine/ocean survival conditions and in the absence of fisheries (after Chapman et al., 1994). The total equilibrium run size annually entering the Columbia River from restored natural production in the John Day Reach under Alternatives 1 and 3 could equal up to 66,600 and 130,000 adult fall chinook, respectively. Subtracting the estimated number of spawners required to sustain this production level results in an estimated 35,500 and 74,000 adults available for harvest, as shown in Table 22.

In comparison, existing hatchery production for mitigation of lost natural production in the John Day Reach could produce under optimum ocean survival conditions up to 144,000 fall chinook salmon for harvest. These represent current high harvest rates that can not be sustained under natural production conditions. Presumably, hatchery production in mitigation for lost natural production in the John Day Reach would be continued at a

decreasing level as natural production was restored, and would ultimately be replaced by restored natural production.

Based on hatchery returns, the existing smolt-to-adult (SAR) return rates including variable ocean conditions and commercial, tribal and sport fish harvests in Alaska, British Columbia, Washington and Oregon is only 0.22 percent. This return rate compares well with estimates made by Anderson et al. (1999) of current SARs for Snake River and upper Columbia River fall chinook salmon. Under poor ocean conditions and high harvest rates, the adult returns under any scenario are insufficient to replace the spawning population.

Table 22.
Available Adults to Fisheries from Increased Habitat Capacity in the John Day Reservoir under Various Drawdown Scenarios.

Natural Production

Drawdown to Spill Crest	Spawner Capacity (#)	Smolt Yield (100 smolts/ spawner)[1]	Hydro System Survival[2]	Potential Adult Returns[3]	Fish Available to Fishery[4]
Existing Conditions	5,500	550,000	314,500	16,100	10,600
Drawdown to Spillway Crest	31,100	3,110,000	1,960,000	100,600	69,500
Natural River	55,000	5,500,000	3,790,000	194,500	139,500

Artificial Production

	Spawner Equivalent	Smolts Released[5]	Hydro System Survival[2]	Potential Adult Returns[3]	Fish Available to Fishery[4]
Hatcheries	30,000	11,900,000	5,110,000	168,000	138,000

1) Assumes 4,300 eggs per female spawner and an egg- to-smolt survival of 4.65% in the wild (Chapman et al., 1994).

2) Assumes a project survival rate of 83%, and estimate varies by the number of hydro projects passed.

3) Assumes a smolt to adult return of 3.29% in good ocean conditions and in lieu of a fishery (Peters et al., 1999), and a survival level for naturally produced smolts that is 1.56 times better than that of hatchery smolts (Chapman et al., 1994).

4) Adult returns less spawner replacements under periods of good ocean conditions.

5) JDD sponsored mitigation releases only (1999 goals).

In summary, drawdown to natural river (Alternative 3) may result in an estimated 8- to 10-fold increase in fall chinook salmon spawning capacity in the John Day Reach compared to current potential levels of spawning. Drawdown to spillway crest (Alternative 1) is estimated to result in 50 to 75 percent of the benefit achieved under natural river drawdown conditions. None of the flood control scenarios (Alternatives 2 and 4) would influence the level of spawning habitat improvements estimated in Alternatives 1 & 3.

If a restored population of fall chinook in the John Day Reach is capable of sustaining itself, increased natural production in the reach could potentially provide harvest benefits equivalent to approximately two to two-and-a-half times the number of spawners using the reach, under good estuarine and ocean rearing conditions. The estimated potential contribution of natural spawners in the John Day Reach to commercial, tribal and sport fisheries (139,500 fish) is only slightly less than the potential number of adults available for harvest under the Corp's existing hatchery mitigation program in combination with current natural production potential (148,600 fish). The primary functional differences between artificial and natural production alternatives are improved egg-to-smolt survival rates and higher sustainable harvest rates under hatchery production conditions. However, continued harvesting of mixed-stock fisheries at hatchery harvest rates would continue to be detrimental to natural production that cannot sustain as high a harvest rate as artificial production.

The potential improvements to spawning habitat under drawdown conditions would benefit the Upriver Bright stock of fall chinook salmon, which is not at risk of extinction nor is it a targeted stock for the intended benefits from John Day drawdown. The Snake River fall

chinook stock, listed as threatened, and the other species that were evaluated will not benefit from changes in available spawning habitat in the John Day Reach. They could, however, be affected by modifications in the amount of rearing habitat, by improvements in reservoir travel time, or by alterations in predator interactions as described in Sections 6, 8 and in Attachment E (Anderson et al., 1999), and by alterations made to access local tributary streams (WEST, 1999).

SECTION 8. Combining Estimates of Change in Fall Chinook Production from Above and Below McNary Dam Resulting from John Day Drawdown

8.1 Purpose of Analysis Integration

The purpose of this analysis is to combine the estimates of change in production of Columbia River Upriver Bright (URB) fall chinook from three sources and to estimate the total potential change in URB production under each of three John Day drawdown alternatives: existing conditions, natural river drawdown and drawdown to spillway crest. The three production sources of URB fall chinook considered are naturally produced fall chinook from the Hanford Reach located above McNary Dam; potentially restored natural production of fall chinook from the John Day Reach, located below McNary Dam; and John Day mitigation hatchery releases, which currently mitigate for lost natural production in the John Day Reach below McNary Dam. The potential contribution of these production sources to the fall chinook fishery and to recruitment of total adult returns were estimated by two independent methods in previous section of this report. In this section, the differences in the estimation methods are considered, a common framework is identified for integration of the estimated production changes among the three sources, and the total change in URB fall chinook production under each drawdown alternative is estimated.

8.2 Sources of Fall Chinook Production

8.2.1 Upper Columbia River Fall Run ESU Naturally Produced Above McNary Dam

The Upper Columbia River Fall Run ESU is comprised of fall-run “ocean-type” chinook salmon in the Columbia River. The chinook salmon in this ESU mature at an older age than the ocean-type chinook salmon in the Lower Columbia and Snake River ESUs. The Upper Columbia ESU includes the fish that managers have termed Upriver Bright (URB) fall chinook that reproduce naturally in the Hanford Reach of the Columbia River located above McNary Dam. The Hanford Reach of the Columbia River is the only unimpounded section of the Columbia River remaining in this ESU. Productivity estimates of Hanford Reach fall chinook were obtained from the PATH analysis (Peters et al 1999).

8.2.2 John Day Reach Natural Production Below McNary Dam

The productivity of naturally reproducing fall chinook in the John Day Reach below McNary Dam was estimated based on hydraulic modeling estimates of potential spawning habitat assuming full seeding. Research from the Hanford Reach indicates redd spacing is typically on the order of one fall chinook salmon redd per 62.3 m^2 (670.2 ft^2) or 65 redds per acre of potentially useable habitat over large areas (Swan 1989). Extremely intense spawning has been reported at a mean density of 280 redds per acre (Swan 1989). Swan (1989) also concluded the Hanford Reach fall chinook salmon population supported a male to female ratio of 1:1. Thus, in this analysis we assume that each female completes one redd and that each redd corresponds to a count of 2.0 adult fish. At full seeding capacity, the estimated 17 ha (42 acres) of potential spawning habitat under current conditions in the upper John Day Reservoir is estimated to support approximately 2,750 redds, or approximately 5,500 spawners. For the upper 10-mile section of the reservoir, this level of seeding would result in approximately 275 redds/mile. This approximation is consistent with other redd density estimates reported in the Columbia and Snake River basins. For estimating redd densities associated with other drawdown alternatives the area of potentially usable spawning habitat was determined as described in Section 7.3.3.

8.2.3 John Day Mitigation Hatchery Production

The John Day hatchery mitigation program produces fall chinook from five hatcheries located at Tanner Creek, Ringold, Umatilla River, Little White Salmon River, and Yakima River. The total return to the mouth of the Columbia River from these production facilities was estimated assuming a smolt-to-adult survival rate (SAR) of 3.29% to represent potential total fish returns under good ocean conditions (Peterson et al., 1999).

8.3 Comparison and Integration of Analysis Methods

Two different methods were used to assess the changes in production of fish from the three production sources. For the naturally reproducing Hanford Reach fish, Anderson et al. (1999) (Attachment E) assumed the production could be described by a Ricker production curve (Figure 7). For the naturally reproducing fish from the John Day Reach, R2 Resources Consultants estimated the maximum potential production in terms of the product of available potential spawning habitat, redds per unit area of potential spawning habitat, average fecundity of a spawning pair and an estimated egg-to-smolt survival rate for juveniles. Hatchery production was estimated from hatchery spawner capacity and average egg-to-smolt survival rates for juveniles produced in hatcheries. Production estimates made by R2 Resources Consultants used what has been referred to as a “hockey-stick” production curve (Figure 7). In this type of curve, next generation spawner recruits increase as a linear function of parent spawners at relatively low spawner levels, but the number of spawner recruits remains constant above some habitat or hatchery facility capacity threshold.

The Ricker curve and the hockey-stick curve treat density dependent mortality differently, which can make comparison of coefficients and results from the two different production analysis methods somewhat problematic. The Ricker curve has a smooth density dependent relationship between spawners and recruits. Thus, offspring mortality rate increases smoothly

as the number of parent spawners increases. In comparison, density dependent mortality with the hockey-stick function is absent below the system carrying capacity and it fixes the recruit number to a constant above the carrying capacity. The two types of production curves are illustrated in Figure 7. The 45-degree line in Figure 7 demarks the one-to-one replacement relationship between spawner recruits and parent spawners. That is, the "equilibrium population level" where production of spawner recruits just replaces the parent spawner population occurs where this line crosses the production curves.

The "sustainable yield" is the excess of next generation recruits produced after spawner replacement and is denoted as the distance between the one-to-one replacement line and the production curve. Theoretically, there is one average spawning population size that produces the maximum sustainable yield, designated by Cs in Figure 7. This maximum sustainable yield, and the associated parent spawning population level, Ss, required to achieve it can be defined for each production curve.

Since Anderson et al (1999) estimated natural production levels at MSY for fall chinook salmon above McNary Dam under various John Day drawdown scenarios using the Ricker function, and R2 Resources Consultants estimated maximum potential natural production levels for fall chinook in the John Day Reach below McNary Dam using the hockey-stick function, both types of production estimates were linked to the maximum potential harvest production in their respective areas. Thus, the two methods represent assessments of maximum sustainable yield (MSY) for URB fall chinook. The two curves intersect at Cs and at Ss as illustrated in Figure 7. Because the two types of production curves estimate production potential around a common MSY value we can directly compare the results from the two analytical methods as long as we refer to the production results obtained at the common MSY value. The two approaches do not give comparable equilibrium population size levels, nor are the Ricker *b* and *a* parameters simply related to the hockey-stick carrying capacity and to the rate of increase in recruits per parent spawner at spawner levels below the carrying capacity, respectively.

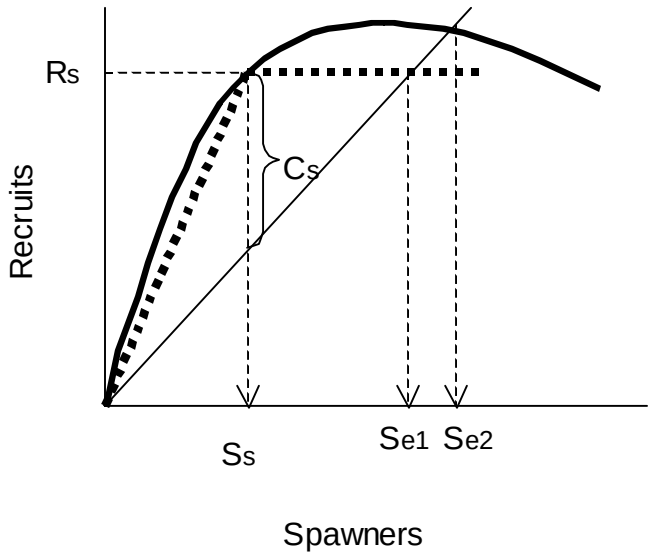


Figure 7 Ricker (solid line) and hockey-stick (dashed line) curves showing relationship between spawners and recruits. The maximum sustainable yield (MSY) is shown by Cs and the associated average spawner population size is shown by Ss. The equilibrium spawner population size levels (without harvest) are denoted by the intersections of the one-to-one spawner-recruit line with the production curves. Se1 is the equilibrium spawner population level for the hockey-stick production function and Se2 is the equilibrium spawner population level for the Ricker production function.

The mathematical equivalence between the Ricker curve and the hockey-stick curve at MSY is developed as follows: The Ricker spawner recruit function takes the form

$$R = S\exp(a - bS)$$

To calculate Ricker *a* and *b* using the parent spawner (*Ss*) and total recruit (*Rs*) values at MSY for the hockey-stick function, the coefficient *b* is solved using the implicit equation for a Ricker curve at MSY of $(1 - bSs)\exp(a - bSs) = 1$ and noting that $\exp(a - bSs) = Rs / Ss$. Once *b* is known, *a* can be determined and the Ricker coefficients that have the same *Ss* and *Rs* values at MSY as the hockey-stick function are given by

$$a = \ln(Rs / Ss) + 1 - Ss / Rs$$
$$b = 1 / Ss - 1 / Rs$$

To integrate and compare the total production of fall chinook from below McNary Dam (Table 22) with production from above McNary Dam under the three John Day drawdown alternatives, the current mitigation hatchery program production must be included with estimates of current Hanford Reach and potential John Day Reach natural production. The production contribution from the mitigation hatcheries under good ocean conditions is given in Table 22.

The production of fall chinook from the Hanford Reach is based on the analysis of Anderson et al. (1999) (Attachment E). In Anderson et al. (1999), total returns at maximum sustainable yield were referenced to the spawning grounds in the Hanford Reach. To integrate these results with estimates of potential natural production in the John Day Reach and with mitigation hatchery production, the Hanford Reach estimates were expanded to reflect total returns to the mouth of the Columbia River using an expansion factor of 1.36 for drawdown to natural river channel and 1.39 for drawdown to spillway crest. The differences in expansion factors relate to the differences in upstream survival under the different alternatives.

8.4 Integration of Results

The total effect of the drawdown alternatives on URB fall chinook production is given in Table 23 where adult productions are assumed to occur under conditions of maximum sustainable yield. Total recruitment to the mouth of the Columbia River is designated by R, the catch at maximum sustainable yield (or the maximum potential catch) is C, and the spawners needed to sustain production (referenced to the mouth of the Columbia River) is designated by S. The change in adult numbers under drawdown compared to current conditions is illustrated by Δ R and Δ S. Note that under natural river and spillway crest drawdowns the mitigation hatchery program is phased out. The total Upriver Bright fall chinook available to the fishery decreases under natural river and spillway crest drawdowns when compared to the existing condition if both mitigation hatchery and potential natural production are considered. However, harvestable surplus from natural production alone would likely increase under drawdown to natural river channel.

Table 23. Comparison of total Up-River Bright (URB) production under three alternatives: existing conditions (A1), natural river drawdown (B1), and spill crest drawdown (B2). R is the number of potential adult returns, C is the number of fish available to fisheries, and S is the number of spawner equivalents referenced to the mouth of the Columbia River. Δ terms are the total difference between alternative and existing conditions for each drawdown scenario. John Day Reach and hatchery productivities are based on a smolt-to-adult survival rate (SAR) of 3.4%

	Existing Conditions			Natural River Drawdown			Spill Way Crest Drawdown		
	R	**C**	**S**	**R**	**C**	**S**	**R**	**C**	**S**
Hanford	256,483	214,640	41,843	179,780	140,659	39,121	169,135	130,544	38,591
John Day	16,100	10,600	5,500	194,500	139,500	55,000	100,600	69,500	31,100
Hatcheries	168,000	138,000	30,000	0	0	0	0	0	0
Total Production	440,583	363,240	77,343	374,280	280,159	94,121	269,735	200,044	69,691
Natural Production	272,583	225,240	47,343						
			Change In	**Δ R**	**Δ C**	**Δ S**	**Δ R**	**Δ C**	**Δ S**
			Total Production	-66,303	-83,081	16,778	-170,848	-163,196	-7,652
			Natural Production	101,697	54,919	46,778	-2,848	-25,196	22,348

SECTION 9. Potential Impacts on Resident Fish and Habitat

The objective of this section is to review the effects of various drawdown scenarios on the resident fish community in the reservoir by evaluating: 1) existing physical habitat conditions and aquatic productivity, 2) potential changes in habitat conditions, 3) the species historically and currently residing in the John Day Reach, and 4) potential changes in resident fish communities based on habitat preferences and changes. Special attention is given to the potential change in resident fish populations of major predators on juvenile salmonid fishes and the associated potential change in predator-related mortalities under the four alternative drawdown scenarios.

9.1 Existing Reservoir Habitat Conditions

Li et al. (1981) identified three habitat types in John Day Reservoir based on water velocities; 1) the forebay (from John Day Dam to RM 247), characterized as slow water velocity due to the pooling effect of John Day Dam; 2) the transitional zone (from RM 247 to RM 282) of intermediate water velocity; and 3) the tailrace (from RM 282 to McNary Dam) defined as fast water velocity due to the influence of turbine outflow and spill from McNary Dam.

Twenty-five percent of John Day Reservoir is less than five m (16 ft) deep (Parsley et al., 1993). The majority of this shallow water habitat is in the upper third of the reservoir, where littoral zone (nearshore, shallow water) habitat is provided in main channel and backwater areas, particularly in Plymouth and Paterson Sloughs (Beamesderfer and Nigro, 1989;

Barfoot et al., 1999 [Attachment A]; Gilbreath et al., 1999 [Attachment B]). Littoral zones can be one of the most productive habitats in aquatic ecosystems, owing to the variety of aquatic macrophytes (large rooted plants) and attached algae that can grow in the shallow waters (Wetzel, 1983) and the residency of many fish species for at least a part of their life history. Water depths gradually increase in a downstream direction, as the riverbanks become more steep-sided and littoral habitat is limited. Depths reach 50 m (164 ft) near the John Day forebay (Giorgi et al., 1985).

Growths of aquatic macrophytes, primarily water milfoil (*Myriophyllum* spp.), are seasonally abundant in the shallow waters of the upper reservoir. Aquatic plants provide habitat for fish and their prey. The abundance of aquatic macrophytes in John Day Reservoir may have increased relative to levels observed by earlier investigators in, 1984 and, 1985 (Barfoot et al., 1999). The reason for this apparent increase in macrophyte biomass is unknown, but is consistent with other reservoirs as they age.

Embayments provide slow water velocities and protection from wind and wave action and can be important habitat for resident fishes. Embayments are present in the Irrigon area and the John Day forebay (Beamesderfer and Rieman, 1991). Substrate in the reservoir is mud, sand, gravel, and cobble (Parsley et al., 1993). Sediment collected from 1, 3, and 5 meter depths at upper reservoir sampling stations was typically classified as silty sand by ASTM D 2487 criteria. The median grain size ranged from 0.1 mm to 0.2 mm, volatile solids content from 0.8 to 2.7 percent, and percentage silt/clay from 11 to 31 percent (Gilbreath et al., 1999).

The different habitats in John Day Reservoir provide spawning and rearing areas for a number of resident and anadromous fish species. Poe et al. (1991) sampled 29 species of resident and anadromous fish in John Day Reservoir from, 1983-1986. Although fish have been collected throughout the reservoir, densities are higher in the upper reservoir, where some resident species show a preference for littoral habitat.

The relative abundance of Northern pikeminnow, largescale suckers, sand rollers, crappies, yellow perch, and walleye was greatest in the upper reach of John Day Reservoir (Palmer et al., 1986; Beamesderfer and Nigro, 1989; Beamesderfer and Rieman, 1991; Barfoot et al., 1999). Most of these fish may favor the upper reservoir due to the slow water velocities and protected conditions provided in backwaters (Palmer et al., 1986; Barfoot et al., 1999) that offer preferred habitat conditions during at least a portion of their life history (Scott and Crossman, 1973; Wydoski and Whitney, 1979). Yellow perch are probably most common in the upper reservoir because of the abundance of aquatic macrophytes. Yellow perch spawn on or near aquatic macrophytes (Scott and Crossman, 1973; Wydoski and Whitney, 1979). The recent appearance of dense growths of milfoil in John Day Reservoir may be favoring the production of this species.

Northern pikeminnow are found reservoir-wide, but the largest numbers are found in the McNary tailrace boat restricted zone (MTBRZ) (Beamesderfer and Rieman, 1991). These fish utilize low-velocity microhabitats of 100 cm/sec (3.3 ft/sec) (Beamesderfer, 1983; Faler et al., 1988; Shiverly et al., 1996). They likely concentrate in the MTBRZ because of the high prey densities available below the dam and they can locate refuge from the high velocity water in backeddies of the tailrace. Walleye also concentrate in the upper reservoir (upstream of Arlington at RM 242) and are frequently found in the McNary tailrace

(Beamesderfer and Rieman, 1991; Beamesderfer and Nigro, 1989). The upper reservoir provides the favorable water depths [shallow water <15m (50 ft) deep] walleye prefer (Bennett, 1999; Attachment C). Walleye are frequently found in nearshore areas (within 165 ft of shore) from April through June and move offshore in late summer. This offshore movement was believed to be the result of late summer flow reduction and a concomitant decrease in mid-channel water velocities (Beamesderfer and Nigro, 1989).

In contrast to observations of northern pikeminnow and walleye, smallmouth bass are more prevalent in the lower reservoir (Beamesderfer and Rieman, 1991). This species is most common in embayments near the John Day forebay. Smallmouth bass are also frequently sampled in embayments in the Irrigon area (RM 282). These embayments provide slow water velocities [less than 15 cm/sec (0.5 ft/sec); Edwards et al., 1983] and protected areas (Hjort et al., 1981; Palmer, 1982) smallmouth bass use.

9.1.1 Planktonic Communities

As reported in Attachment B, Gilbreath et al. (1999) performed limnological surveys of John Day Reservoir in, 1994 and, 1995. These surveys allow for a characterization of the planktonic and benthic communities in the reservoir.

Seasonal phytoplankton abundance (as indexed by chlorophyll a) in John Day Reservoir is similar to other temperate waters. The spring maximum in April and May is followed by decreased abundance in summer with a secondary, small peak in autumn (Gilbreath et al., 1999). Based on summer chlorophyll *a* levels measured in, 1994 and, 1995, John Day Reservoir is classified as mesotrophic, or moderately productive (Chapra and Tarapchak, 1976).

Similar to other Columbia River reservoirs, the densities of zooplankton in the John Day Reach are generally low. Zooplankton abundance in free flowing rivers is often low because these organisms are destroyed by turbulence and contact with suspended material (Gilbreath et al., 1999). Zooplankton densities, like phytoplankton, are positively correlated with water retention time. Consequently, zooplankters may not have time to complete their life cycles in John Day Reservoir (Gilbreath et al., 1999). It is interesting to note that zooplankton concentrations were considerably higher at the protected Crow Butte backwater than at stations off the main river channel.

Rotifers are the most common taxon of zooplankton in John Day Reservoir, however, they are not important prey items for fish (Gilbreath et al., 1999). Important prey species like, Copepod and cladoceran zooplankton, are also present. The maximum densities of copepods are more prevalent than cladocerans. Copepods and cladocerans are more abundant in the deep waters of the lower and middle reservoir than in the shallow waters of the upper reservoir (Gilbreath et al., 1999). This pattern of horizontal distribution is not surprising since cladocerans and copepods favor deep pelagic zones of lakes (Wetzel, 1983).

The planktonic community supplies food for juvenile salmonids during their outmigration through the reservoir. Subyearling chinook migrating through the mid-reservoir section of the McNary Reservoir were found to feed primarily on a large type of Cladoceran (*Daphnia* spp.) and terrestrial insects (Rondorf et al., 1990). These species were selected because these were the two most abundant prey items in the reservoir during the outmigration period of the

juvenile chinook. *Daphnia* spp. densities during the outmigration period ranged from 1.2 to 18.5 organisms per liter.

The peak occurrence of subyearling chinook in the nearshore habitats in the John Day Reservoir occurs in May and June (Poe et al., 1991; Barfoot et al., 1999; Gilbreath et al., 1999). *Daphnia thorata* are present in John Day Reservoir during May and June. However, their densities do not average greater than 0.21 organisms per liter (Gilbreath et al., 1999), so they are probably not abundant enough to be an important food item for outmigrating juvenile chinook. The most common cladoceran and copepod during May and June are the small-bodied, *Bosmina longirostris* and nauplii (immature copepod). The densities at all stations of *B. longirostris* ranged between 0.07 and 1.17 organisms per liter and the densities for nauplii ranged between 1.24 and 5. 06 organisms per liter during the, 1994 and, 1995 subyearling outmigrations (Gilbreath et al., 1999). The small size and low abundance of these zooplankton reduces their importance as prey items for juvenile salmonids during their outmigration through John Day Reservoir.

9.1.2 Benthic Communities

Macroinvertebrate density in the upper section of John Day Reservoir is highest in June and September, annually (Gilbreath et al., 1999). No data for macroinvertebrate densities in the lower section of the reservoir are available.

Although the diversity of macroinvertebrates is high, the community is numerically dominated by burrowing oligochaetes and chironomids (Gilbreath et al., 1999). It is common for the benthos of reservoirs to be characterized by a few taxa, particularly chironomids (Merritt and Cummins, 1983). The deep water ensures a simple environment, where water velocity is minimal, temperature varies little, and substrate is homogenous.

The benthic community can provide food for species of resident and anadromous fishes. Subyearling chinook in the Hanford Reach of the Columbia River were found to consume mostly insects (95 percent), of which the majority were chironomids (Chapman et al., 1994). Caddisflies were the predominate food of subyearling chinook in the riverine section of McNary Reservoir (Rondorf et al., 1990).

9.1.3 Resident Fish Species Abundance and Community Structure

Of the 35 fish species sampled in John Day Reservoir by Poe et al. (1991), thirty are residents. Fourteen of these resident species (56 percent) have been introduced to the Columbia River Basin from other areas.

Palmer et al. (1986) sampled John Day Reservoir at several locales with a beach seine to determine the relative abundance of prey fishes (fish smaller than 250 mm) in the spring and summer of, 1985. They captured over 16,000 fish representing 27 species. They found largescale suckers were the most abundant prey fish (30 percent of catch), followed by sand roller (17 percent), northern pike minnow (12 percent), and crappies (11 percent).

Barfoot et al. (1999) used a beach seine in, 1995 to determine resident fish use of nearshore habitats in six different regions of John Day Reservoir. They sampled main channel and backwater areas in the reservoir. Yellow perch, sculpins, and sand rollers dominated the fish community at main channel stations in, 1995. Introduced species comprised 34 percent of

the main-channel catch. Northern pikeminnow, yellow perch, and peamouth were the most common species found in backwater areas. Introduced species were similarly a third of the catch in backwater areas.

Beamesderfer and Rieman (1991) found northern pikeminnow were the most abundant predator in John Day Reservoir, followed in order of decreasing abundance by smallmouth bass and walleye. The largest concentrations of northern pikeminnow and walleye were found in the McNary tailrace and the boat restricted zone. Smallmouth bass were distributed in the lower reservoir and were most common in the John Day forebay (Beamesderfer and Rieman, 1991).

Gilbreath et al. (1999) performed 116 seine hauls in the upper section of John Day Reservoir from May, 1994 to September, 1995, catching a total of 5,842 fish. Fourteen species were collected, the majority of which were juveniles. American shad were the predominate species, followed in abundance by chinook salmon, northern pike minnow, smallmouth bass, and other centrachids. Young-of-the-year (0-age) fall chinook (42 to 98 mm in size) were collected along the nearshore habitats of the upper reservoir from May to mid-July during, 1994 and, 1995. Abundance peaked annually in May. Less than 1 percent of the chinook fry catch occurred in early July and juvenile chinook were absent in the catch from late July through November, annually. Reservoir water temperatures reach 20°C by late July and remain near this level into October. Water temperatures above, 19°C are suboptimal for juvenile chinook growth and metabolism (Raleigh and Miller, 1986; Bell, 1990; Reiser and Bjornn, 1991) and may account for the lack of nearshore catches after mid-July. Juvenile chinook fry are known to move offshore as temperatures warm in the summer months. They may continue to rear in the reservoir in deep waters as evidenced by purse seine catches in the John Day forebay (Giorgi et al., 1995) or they may initiate downstream movements. Smoltification and the speed of outmigration have been shown to increase with warmer temperatures late in the migration season (Giorgi, 1993; Skalski and Townsend, 1999; Attachment D).

Conversely, the activity and abundance of common resident species known to prey on salmonid fry (northern pikeminnow, smallmouth bass, and other centrachids) increases with water temperature. The overlap with fall chinook salmon juveniles and major salmonid predators in the shallow water areas of the upper John Day Reservoir may be limited in duration.

9.1.4 Predation Estimates

In studies of the John Day Reservoir, four resident fish species were found to consume the greatest amount of juvenile salmonids including: northern pikeminnow; walleye, smallmouth bass and channel catfish (Poe et al., 1991, Vigg et al., 1991; Beamesderfer and Rieman, 1991). The northern pikeminnow accounted for approximately 78 percent of the salmonids lost to predation. Walleye and smallmouth bass were less significant predators consuming an estimated 13 and 9 percent of the total. Channel catfish also consumed juvenile salmonids, but their consumption rates were less than the other predators (Vigg et al., 1991). The tailrace downstream of McNary Dam is an are of intense predation (Poe et al., 1991; Ward et al., 1995). However, given the large area and abundance of predators, the mid-reservoir area may account for the greatest volume of predatory-related losses (Petersen, 1994). Of the four

predator species studied, only northern pikeminnow included juvenile salmonids as the dominant food item during the spring and summer emigration periods through John Day Reservoir.

The most recent estimate of smolt loss in the reservoir is approximately 1.4 million smolts annually (Petersen, 1994). The predation-related mortality of salmonid fishes is a function of many factors including: 1) the predator feeding behavior and metabolism, 2) prey condition and behavior, and 3) the spatial and temporal distributions of both smolts and predators in the reservoir. Three items generally influence the amount of loss annually due to predation; 1) the abundance of predator populations, 2) the consumption rate, that varies seasonally with biological activity and water temperatures and 3) the overlap of predator and prey distributions spatially and seasonally.

On average, the number of northern pikeminnow greater than 250 mm in size found in the reservoir from 1984 to 1986 was estimated at 85,316 fish with an overall density of 4.4 fish per hectare or 1.8 fish per acre (Beamesderfer and Rieman, 1991). The density of northern pikeminnow in the boat restricted zone downstream of McNary dam was 12 to 18 times higher than elsewhere in the reservoir. Recent data suggest juvenile pikeminnow relative abundance (ranks) and catch-per-unit-effort (CPUE) in nearshore waters of the reservoir may have declined in the last decade compared to other non-native resident species (Table 24; after Barfoot et al., 1999). The impact of reduced juvenile abundance may lead to lower recruitment of adults. These findings may reflect poor juvenile production resulting from changing environmental conditions in the reservoir or from reduced spawner abundance as a result of targeted fisheries on northern pikeminnow.

Table 24.
Catch-Per-Unit-Effort (CPUE)[1] of Three Dominant Predators of Juvenile Salmonid Fishes in the John Day Reservoir; after Barfoot et al. (1999).

Species	1984	1985	1995
Northern Pikeminnow	24.9	20.9	8.7
Smallmouth Bass	1.3	2.8	2.1
Walleye	0.01	0.03	0.17

1) CPUE = Number of resident fishes collected at six sampling locations/number of seine hauls.

Beamesderfer and Rieman (1991) estimated the annual adult population of smallmouth bass ranged between 31,948 and 37,959 fish at an average density of about 1.8 fish/ha. (0.7 fish/acre). Year class strength was highly variable between the years indicating a large influence of physical or biological factors on recruitment of adult smallmouth bass. Beamesderfer and Ward (1995) reported a five-fold difference in recruitment between, 1976 and, 1982. Barfoot et al. (1999) show relatively little change in CPUE of juvenile smallmouth bass in the reservoirs between the mid-1980's and, 1995 (Table 24).

Beamesderfer and Nigro (1989) reported the walleye population in John Day Reservoir consisted of about 15,000 fish (0.7 fish/ha.; 0.3 fish/acre) > 250 mm in length. Most of the walleye are found in the upper half of the reservoir because of the extensive littoral habitat. Beamesderfer and Rieman (1991) indicated a general decrease in abundance of walleye from upstream to downstream. Barfoot et al. (1999) show relatively strong year classes of juvenile

walleye in the, 1995 sampling effort in comparison to juvenile CPUEs in the mid-1980s (Table 24).

Based on model predictions, Beamesderfer et al. (1990) expect predation in the reservoir to have a significant effect on reservoir survival during most years. Rieman (1990) believes that predation-related mortalities can equal or exceed dam-passage related mortalities.

Conversely, many researchers believe that adverse effects of reservoirs on outmigrating juvenile salmonids are much less of an influence than passage issues at dams for those with a strea-type (i.e., yearling migrant) life history strategy. Iwamoto et al. (1994) and Muir et al. (1995) indicated that virtually all of the mortality measured in the mainstem Snake River for yearling outmigrants was attributable to fish passing through the hydroelectric structures and the reservoir losses, including predation, were quite low.

Fall chinook subyearling juveniles, on the other hand, suffer considerable mortalities as a result of predation. Summer PIT-tag data for impounded reaches in the Snake River indicate reach survivals range between 62 and 95 percent, and average 75 percent (NMFS, 1999), implying greater reservoir and dam passage mortalities for this race compared to yearling outmigrants.

Survival of PIT tagged fall chinook salmon juveniles passing the John Day Reservoir was found to be on the order of 83 percent during the, 1998 outmigration period (Skalski and Townsend, 1999). However, this estimate includes both reservoir and dam passage survival. Since, only hatchery fish were PIT-tagged, this survival estimate may not represent survival of naturally produced fall chinook from either the Snake River or from the mid- and upper Columbia River that migrate through, or rear in, the reservoir. The smaller naturally produced fish may have greater exposure to predators than larger hatchery fish.

The literature concerning the magnitude of predator-related mortality in the mainstem reservoirs is mixed and the role predation plays with respect to determining population run sizes is uncertain. It is clear that predacious fishes in the John Day Reservoir are consuming salmonids, but the extent of loss remains uncertain.

9.2 Estimated Habitat Conditions and Altered Community Structures after Drawdown

The following four alternative drawdown scenarios are under consideration for this Phase I evaluation; Alternative 1, drawdown to spillway crest without flood control; Alternative 2, drawdown to spillway crest with flood control; Alternative 3, drawdown to natural river without flood control; Alternative 4, drawdown to spillway crest with flood control. The anticipated reservoir habitat conditions and subsequent changes in the aquatic biological communities are assessed for each alternative below. The greatest expected physical, chemical and biological changes would occur under drawdown to natural river at an elevation of 165 feet MSL (Alternative 3). Habitat conditions and existing biota would be altered less with a drawdown to spillway crest (Alternative 1) compared to drawdown to natural river, since reservoir-like conditions would prevail in the lower reservoir between John Day Dam and RM 246 under drawdown to an elevation of 220 feet MSL (Bennett, 1999). As a consequence, habitat for existing biota would be available and the fish, plankton, and benthic community structure would not change appreciably in the lower

portion of the reservoir. Changes in biological communities are more difficult to predict for alternatives with flood control options. These measures are not initiated until the river exceeds a discharge of 500 kcfs. The variable frequency and duration of these events makes estimating biological community changes obscure.

9.2.1 Drawdown to Spillway Crest (Alternative 1)

Lacustrine (slow-flowing, lake-like) habitat in the lower 15 miles of John Day Reservoir, between the dam and RM 231, would be similar to present conditions, under the drawdown alternative to spillway crest (Bennett, 1999). Between RM 231 and RM 246, habitat conditions would be transitional between lacustrine and riverine. However, they would change noticeably compared to existing habitat features as more riverine conditions would predominate. Upstream of RM 246 to McNary dam, habitat conditions would be considered primarily riverine under this alternative.

The drop in water elevation in the upper reservoir would concentrate all water in the main channel of the reservoir, where water velocities would increase, depths would decrease, and substrate composition would change relative to existing conditions. The drop in water elevation would effectively eliminate the littoral zone and backwater habitat in the upper reservoir. Although shallow water habitat would exist along the margins of the main channel, the fast water velocities would preclude the use of this habitat by species preferring shallow, but slow water. The loss of slow, shallow-water areas would decrease the availability of habitat currently used by some resident and anadromous fish in the upper reservoir. Drawdown would also reduce the area that could be colonized by submerged macrophytes. The fast water would transport small-sized material downstream, exposing large-sized gravel, cobbles, and boulders (Bennett, 1999). It is uncertain how long a change from fine to coarse-bedded substrate would take.

Drawdown would eliminate most submerged macrophytes in the upper reservoir due to exposure of the littoral zone and subsequent macrophyte desiccation. The change to a riverine environment containing gravel and cobble substrate would prevent the re-establishment of macrophytes in shallow water because they do not tolerate fast water or coarse substrate. An adequate supply of fine, nutrient-rich sediment must exist for the rooting of macrophytes (Welch, 1992).

The decreased water depths and short water retention time following drawdown would also change the structure of primary producers in the upper reservoir. The production of phytoplankton in the upper reservoir would decline because decreased water depths and shorter water retention times would be disadvantageous to their growth. Pelagic phytoplankton are intolerant of the short residence time and turbulence that characterize running water (Welch, 1992). Shallow water depths would allow more light to reach the substrate and would promote the growth of attached algae (Bennett, 1999). Attached algae are known to strongly influence the productivity in lotic or running waters (Welch, 1992). Attached algae have a higher net production rate per unit area relative to submerged macrophytes (Wetzel, 1964; Westlake, 1975). It is assumed the attached algae community could substitute for the lost primary production if the submerged macrophyte population was eliminated following drawdown. However, given the lack of studies quantifying net primary productivity of phytoplankton in running waters, it is uncertain whether attached algae can

make up for the loss of submerged macrophyte and phytoplankton productivity in the upper reservoir.

The benthic macroinvertebrate community (invertebrate species living on or within the bed of the river) in the upper reservoir would change as a result of the drawdown. The shallow, swift waters and variable substrate conditions resulting from drawdown would create a greater diversity of benthic microhabitats relative to existing conditions (Bennett, 1999). This diversity of microhabitats favor the colonization of substrates by a variety of rheophilic (current-loving) macroinvertebrates, primarily caddisflies, stoneflies, mayflies, and true flies. These species currently do not reside in the upper reservoir in great numbers (Gilbreath et al., 1999). These macroinvertebrates often feed on attached algae or material provided by the current (the drift) and often passively or actively drift with the current. This change in the benthic macroinvertebrate community could increase the food supply in the upper reservoir because a greater percentage of the benthic community would be likely drift into the water column and be available as prey for juvenile salmonids that outmigrate through the reservoir (Bennett, 1999).

Although the biomass (unit weight) of benthic macroinvertebrates in John Day Reservoir is high, the most common species (oligochaetes and chironomids) are burrowers (Gilbreath et al., 1999). Chironomids are reported to be an important prey item of juvenile salmonids under riverine conditions. Subyearling chinook in the Hanford Reach of the Columbia River were found to consume mostly adult and subadult insects (95 percent), of which the majority were chironomids. Most of the insects consumed were drifting or swimming in the current or floating at the surface and not attached to the substrate (Becker, 1970). Chironomids in John Day Reservoir burrow into the substrate and are not available in the water column until they emerge from their cases as subadults. Although chironomids are undoubtedly present in the substrate of the mid-reservoir sections, they were not found in the stomachs of outmigrating juvenile salmonids (Rondorf et al., 1990). It is possible that lower water temperatures at the bottom of the reservoir alter the emergence timing of the chironomids so their presence in the water column does not overlap that of outmigrating salmonids. For whatever reason, chironomids are not preyed upon by outmigrating chinook in slow moving reservoir habitats and they are not believed to be an important food source for juvenile chinook.

The drop in water elevation in the lower reservoir would not have a substantial influence on the benthic or planktonic communities (Bennett, 1999). Water depth would decrease, but temperatures, substrate, and water velocities would not change noticeably from existing conditions. Due to the lack of gross physical changes, the biological communities residing in the lower reservoir would not change materially.

The production of most centrachid fishes, such as crappie, in the upper reservoir would be limited by the lack of littoral habitat. Due to their compressed body type, centrachids can not reside in constantly flowing water. They need slow water to efficiently maintain position in the water column and capture food.

Walleye and yellow perch require similar protected habitat during at least a phase of their life history. Yellow perch in particular could be adversely affected by the drawdown. This species can use submerged macrophytes to spawn and the elimination of these plants could severely reduce their production. Walleye juveniles require protected areas with long water retention times for rearing and the lack of such habitat in the upper reservoir will decrease

adult recruitment. Littoral habitat is also used by outmigrating anadromous salmonids, but they will be able to utilize the riverine habitat in the upper reservoir, which would provide a superior rearing environment compared to littoral habitat (Bennett, 1999).

Habitat for smallmouth bass would still be available in embayments near the forebay and their production is not expected to be altered. Slow water habitat for other centrachids would also be present in the forebay. Phytoplankton would remain the dominant primary producer in the lower reservoir due to prevailing long residence times and deep water following drawdown to 220-ft MSL (Bennett, 1999). Zooplankton communities would also be similar due to the deep, slow water in the lower reservoir. Because the bottom sediments of the lower reservoir would still experience low water velocities, minimal temperature variations, and no change in dominant substrate classes, the benthic macroinvertebrate community is not expected to be different from existing conditions (Bennett, 1999).

9.2.2 Drawdown to Spillway Crest with Flood Control (Alternative 2)

Options to control flood waters create variable water depths in the lower reservoir of Alternative 1 that are a function of flood stage and duration. This changing variability makes it difficult to assess the habitat conditions available to resident fishes. In general, flood control alternatives are initiated at 500 kcfs. The time period for such flows is typically in May and June annually. This season is an active biological period and flood control could modify the short-term rearing potential for a number of species as identified in Alternative 1. It is not possible to quantify the changes in a meaningful way with the available information. However, in a general way the biological alterations discussed above under Alternative 1 will apply to this alternative as well.

9.2.3 Drawdown to Natural River (Alternative 3)

Drawdown to an elevation of 165 feet MSL would create riverine habitat throughout the entire reservoir (Bennett, 1999). The normative river concept predicts the resulting habitat would be similar to conditions present in the free-flowing Hanford Reach. Changes that would result throughout the reservoir following drawdown to 165 feet MSL would be similar to those described previously for the upper reservoir under drawdown to spillway crest.

The drop in water elevation would concentrate all water in the main channel of the reservoir. Water depths would decrease and water velocities would increase reservoir-wide. The fast water velocities would move fine substrates through the system, leaving behind mainly gravel, cobble, and boulders (Bennett, 1999).

The flowing water and large substrate would prevent the colonization of shallow water habitats by submerged macrophytes. Instead, these areas would serve as habitat for attached algae. The attached algae would subsequently provide food for macroinvertebrates as well as herbivorous (plant-eating) and omnivorous (both plant and animal-eating) fish that could take advantage of the new riverine habitat. Per unit area, attached algae are more productive than submerged macrophytes in both river and lake environments (Wetzel, 1964; Westlake, 1975). Assuming the area colonized by attached algae under drawdown to 165 feet MSL is greater than the littoral area currently colonized by submerged macrophytes, primary production supplied by the attached algae would exceed the production of submerged macrophytes.

The short water residence time in John Day Reservoir resulting from the increased water velocities would eliminate current planktonic communities. Phytoplankton growth would be limited by the short exposure to nutrients in the fast flowing water. Zooplankton production would also be reduced because of the flowing water and the lack of phytoplankton. It is uncertain whether the primary production formerly provided by phytoplankton would be replaced by the primary production of attached algae expected to colonize the reservoir under riverine conditions.

The benthic community would be altered substantially by drawdown to the natural river elevation. The community would change from one dominated by quiescent-water organisms that burrow into the substrate to a population comprised mainly of riverine organisms attached to the surface of the substrate (Bennett, 1999). Due to the variety of substrates, water velocities, and temperatures that would occur under riverine conditions, the benthic community would be more diverse relative to existing conditions. Because the attached community is more likely to be dislodged by the current and enter the water column, the riverine benthic invertebrates would be more available as forage for fish species that use the reservoir and should benefit salmonid production (Bennett, 1999).

The shift to a more riverine environment would expose littoral zones, backwater habitat and the main embayments within the reservoir. This exposure would eliminate habitat where northern pike minnow, largescale suckers, yellow perch, walleye, and smallmouth bass and other centrachids are found in great abundance.

The fish community would shift to one dominated by riverine species that are native to the Columbia River (Bennett, 1999). Chiselmouth, northern pike minnow, peamouth, and redside shiners production would likely increase and these species would dominate the fish community in terms of abundance and biomass. The production of white sturgeon would probably also benefit from a return to more riverine conditions due to the more favorable water velocities throughout the reservoir and length of flowing water habitat that could offer successful embryo attachment (Bennett, 1999). The amount of potential spawning habitat estimated for white sturgeon under this alternative is approximately ten times the current level (Figure 8) as modeled by the USGS (1999). Rearing conditions for white sturgeon may not improve with drawdown. This life stage may subsequently limit the population size of sturgeon in the John Day Reach following drawdown.

Introduced species that flourish in John Day Reservoir would decline in abundance under the turbulent, riverine conditions. The production of yellow perch, walleye, and crappie and other centrachids would decrease due to the loss of macrophyte cover and the loss of preferred quiescent habitat.

9.2.4 Drawdown to Natural River with Flood Control (Alternative 4)

Options to control flood waters create variable water depths in the lower John Day Reach that are a function of flood stage and duration. This changing variability makes it difficult to assess the habitat conditions available to resident fishes. In general, flood control alternatives are initiated at 500 kcfs. The time period for such flows is typically in May and June annually. This season is an active biological period and flood control could modify the short-term rearing potential for a number of species as identified in Alternative 1. It is not possible to quantify the changes in a meaningful way with the available information.

However, in a general way the biological alterations discussed above under Alternative 3 will apply to this alternative as well.

9.3 Estimated Change in Predation-Related Mortality after Drawdown

The change in predation losses to juvenile salmonid fishes as a result of altered resident fish communities is uncertain. With drawdown, it is safe to predict that predators and prey will redistribute according to the characteristics of the newly encountered habitat (i.e., water depths, water velocity, substrates and food resources). With this new distribution, predator and prey encounters will likely change and consumption of salmonids will change accordingly.

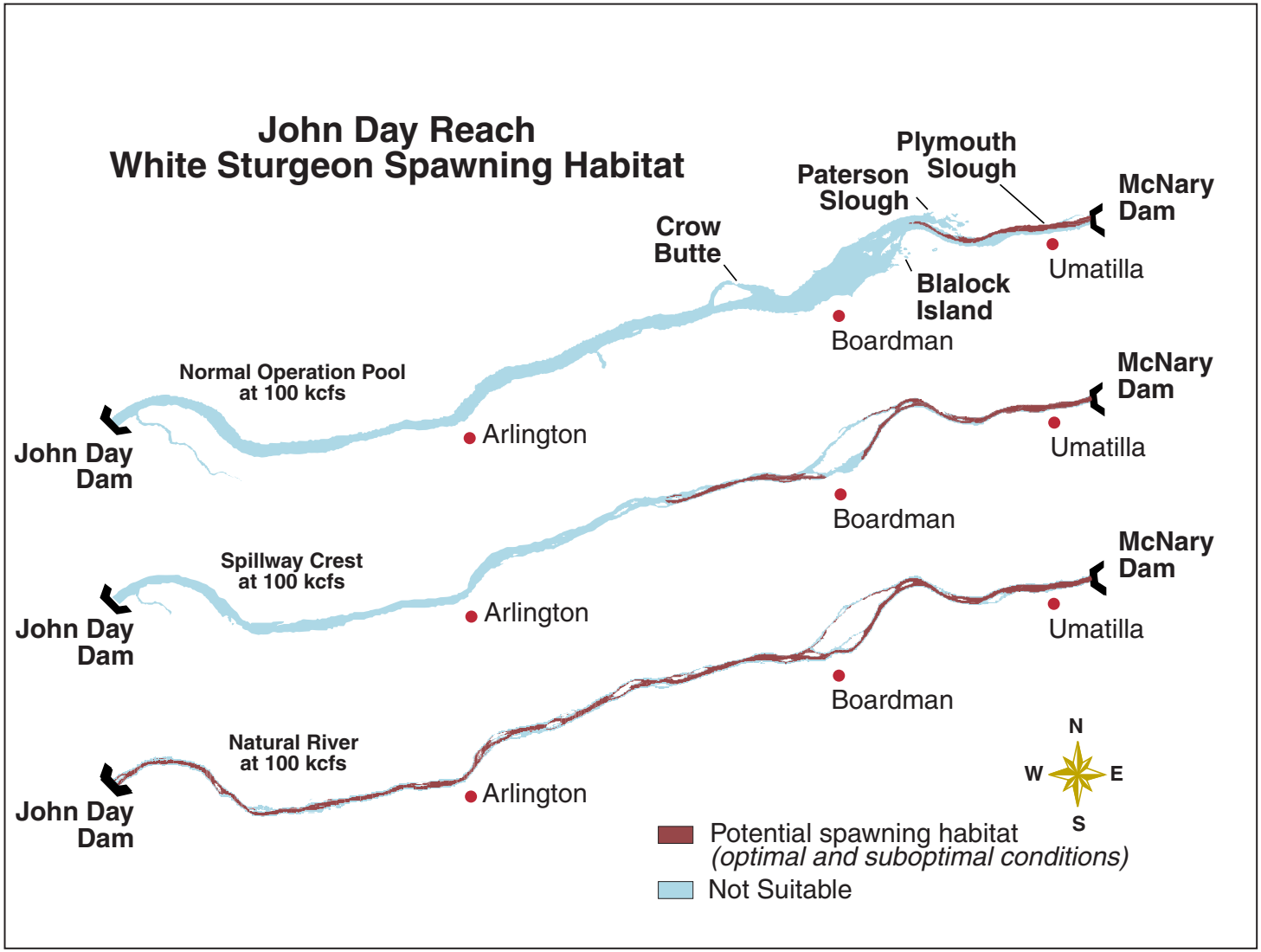


Figure 8. John Day Reach - Potential White Sturgeon Spawning Habitat (after USGS Digital Atlas Reservoir Model 1999).

Populations of three of the most influential predators, the northern pikeminnow, channel catfish and smallmouth bass, are expected to either stay the same or increase slightly under drawdown conditions (Attachment C, Bennett, 1999). Based on studies in free-flowing sections of the Snake River, Petersen et al. (1999) suggest that northern pikeminnow populations may double, but that the populations of smallmouth bass may decrease in half with a change from an impounded reach to a natural Snake River Reach.

Of the four major predators, drawdown may substantially influence only the walleye and perhaps the smallmouth bass populations. Although predator populations may not decrease, predator efficiency in locating and consuming salmonids may change with drawdown. Pikeminnow and smallmouth bass would likely be restricted to slow moving waters, backeddies, embayments and littoral zones in the John Day Reach and they may have less access to juvenile salmonids than under existing reservoir conditions. Thus, habitat overlaps between predators and prey may decline under drawdown.

Although predation on salmonid fishes has been studied extensively in the Columbia and Snake Rivers, relatively few data are available from free-flowing reaches that can be used for assessing the effects of drawdown. However, Petersen et al. (1999) specifically sampled northern pikeminnow (NPM) and smallmouth bass (SMB) in the free-flowing portion of the lower Snake River (RM 147 to RM 172) to assess possible changes in predation of salmonids related to the evaluation of breaching mainstem Snake River dams. Their sampling suggested that NPM densities were higher and SMB densities were lower in the free-flowing reach compared to mid-reservoir densities. They found juvenile salmonids were relatively uncommon in the diets of the predators collected in the riverine reach of the lower Snake River. During the April to May time period, juvenile salmonids represented 7.8 percent of the NPM diet and < 1.0 percent of the SMB diets. During June through August, juvenile salmonids were 1.0 percent and 4.2 percent of the diets of NPM and SMB, respectively (Petersen et al., 1999).

Bioenergetic modeling for the Snake River reservoirs imply predation-related mortalities from NPM and SMB are currently minimal on spring migrants (~1 percent), but could be considerable on summer migrants, mostly fall chinook (~ 59 percent). However, Petersen's et al. (1999) analysis indicates after dam breaching, the total consumption of juvenile salmonids would decrease 74 percent in spring and 83 percent in summer. Most of the change was attributable to a shift in the diets of NPM and SMB, and a reduction in the SMB population.

We conclude that potential benefits with respect to reduced predator-related mortalities on the various stocks would be minimal for spring migrants (spring chinook, sockeye, steelhead), but may be considerable for fall chinook salmon that outmigrate during summer under drawdown conditions. Quantification of the anticipated reduction is not possible with the information available to date for any of the drawdown alternatives or for the various stocks.

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SECTION 11. Attachments

Attachment A: Composition of resident fish communities in near-shore habitats of John Day Reservoir, and potential for changes in predator-prey relationships Barfoot et al. (1999). Draft progress report, ongoing research studies, John Day Dam Phase I Drawdown. U.S. Army Corps of Engineers, Portland, OR.

Attachment B: Limnological investigations in John Day Reservoir including selected upper reservoir habitats, April, 1994 to September, 1995. Gilbreath et al. (1999). Draft report. U.S. Army Corps of Engineers, Portland, OR.

Attachment C: Potential effects of drawdown on habitat and associated biota in John Day Reservoir, Columbia River. Bennett, D. H. (1999). U. S. Army Corps of Engineers, Portland, OR.

Attachment D: Analysis of smolt travel time and survival through the John Day Project using available PIT-tag data. Skalski and Townsend. (1999). U. S. Army Corps of Engineers, Portland, OR.

Attachment E: Modeling the impacts of John Day drawdown on the survival of salmonid stocks. Anderson et al. (1999). Draft report to the U.S. Army Corps of Engineers, Portland, Oregon. October 6, 1999. 77pp.

Attachment A: *Near-shore Fishes*
John Day Reservoir Phase I Drawdown

FINAL REPORT

Near-shore Fishes
John Day Reservoir
Phase I Drawdown

U.S. Geological Survey
Western Fishery Research Center
Columbia River Research Laboratory
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Report prepared for:

U.S. Army Corps of Engineers
Portland District
Portland, OR

Introduction

Drawdown of the John Day Reservoir is part of a strategy proposed in the Northwest Power Planning Council's fish and wildlife program to increase survival of juvenile salmonids (*Oncorhynchus* spp.) migrating through the reservoir (NPPC 1994). Hydroelectric development within the Columbia River Basin has modified the timing and magnitude of water flows during juvenile salmonid outmigration. The creation of impoundments greatly reduced water surface gradients and increased cross sectional areas in impounded segments of the basin resulting in substantially lower water velocities throughout these areas. Reservoir drawdown is hypothesized to increase average cross sectional water velocities and therefore decrease juvenile salmonid travel times. Decreased juvenile salmonid travel times will presumably result in less exposure to predators and other mortality agents such as disease organisms.

In addition to decreasing travel times, reservoir drawdowns may provide other benefits to anadromous salmonids. Recent discussions concerning changes in hydrosystem operations have focused on restoration of salmonid populations through ecosystem-based approaches such as the "Normative River" concept put forth by the Independent Science Group (ISG). This approach calls for re-establishment of a riverine character in key stream reaches of the Columbia River Basin. In impounded reaches, substrates, velocities, depths, and other important habitat components may be unsuitable for establishing or maintaining healthy salmonid populations and other associated coevolved biota. Drawdowns may result in riverine conditions similar to those in the unimpounded Hanford Reach of the Columbia River, for example, where

main-stem spawning populations of fall chinook salmon (*Oncorhynchus tshawytscha*) are healthy and productive, and fish species composition presumably more closely resembles pre-impoundment assemblages. Historically, fish communities in the Columbia River Basin were dominated by stenothermic coldwater salmonids and cottids (Li et al. 1987). However, development for hydropower and other uses tremendously modified aquatic habitats and altered native biotic communities in rivers of the basin.

These impacts greatly restructured fish assemblages which were then further modified by a proliferation of introduced warmwater and coolwater fishes, such as centrarchids and percids, in littoral and sub-littoral habitats of reservoirs (Li et al. 1987; Poe et al. 1994).

The consequences of drawdowns, however, are largely unknown and dependent on a variety of drawdown scenarios (i.e., magnitude, timing, duration). Of specific importance are questions concerning both the short-term and long-term effects of drawdowns on the physical environment, trophic-level interactions (particularly predator-prey relationships), main-stem fall chinook rearing habitats, and on fish assemblage composition in the currently impounded areas. Drawdown will greatly impact shoreline areas which are important habitats for all life stages of resident fishes and also critical rearing habitats for fall chinook salmon. Little is known, however, about the composition of resident fish assemblages in shoreline habitats of the John Day Reservoir. Thus, biotic communities in shallow littoral habitats of the reservoir should be identified prior to any changes in water-level management.

To provide more information on fish communities in the John Day Reservoir, in 1995 we replicated an earlier (1984-85) study (Gray et al. 1984; Palmer et al. 1986) of fishes in near-shore habitats to better understand spatial and temporal patterns of assemblage structure in this impoundment. Identifying patterns of assemblage structure is important for evaluating the consequences of management actions such as drawdowns. However, a number of factors may contribute to substantial variability in reservoir assemblage structure over time, which may make assessment of management actions difficult. Reservoir fish assemblages may vary greatly in time

because of the artificial nature of impoundments and their biotic makeup. Habitat conditions in impoundments, for example, are typically much more physically variable (e.g., short-term water-level fluctuation, low water retention rates) than in natural lakes. In addition, reservoir fish communities often consist of a non-coevolved mixture of both native and introduced species (Gelwick and Matthews 1990), which may contribute to assemblage instability over time. In this report we examine trends in shoreline fish assemblages in a lower Columbia River reservoir to determine if existing information is adequate for evaluating possible future management actions. We analyzed assemblage similarity at three temporal scales: seasonal (spring compared to summer), annual (1984 compared to 1985), and over a long time interval (1984-85 compared to 1995). We additionally summarize catch information for juvenile salmonids in shallow habitats of the reservoir. This information will complement baseline limnological and invertebrate studies begun in 1994 in the John Day Reservoir by the National Marine Fisheries Service and a literature review conducted by the University of Idaho (Dr. David Bennett).

Methods

Study area and field methods.--During 1995 we replicated an earlier (1984-1985) study (Gray et al. 1984; Palmer et al. 1986) of resident fish assemblages in shoreline areas of the John Day Reservoir (Figure 1). The 123-km long reservoir, the largest impoundment on the lower Columbia River, was created when the John Day Dam was completed in 1968. The upper third of the reservoir contains extensive shallow water habitat in main-channel and backwater areas. Further downriver banks become steep-sided and shallow littoral habitat is limited. Substrate is mud, sand, gravel, and cobble (Parsley et al. 1993). Growths of aquatic macrophytes, primarily water milfoil *Myriophyllum* spp., are seasonally abundant in shallow littoral areas, particularly in shallow embayments and backwater habitats.

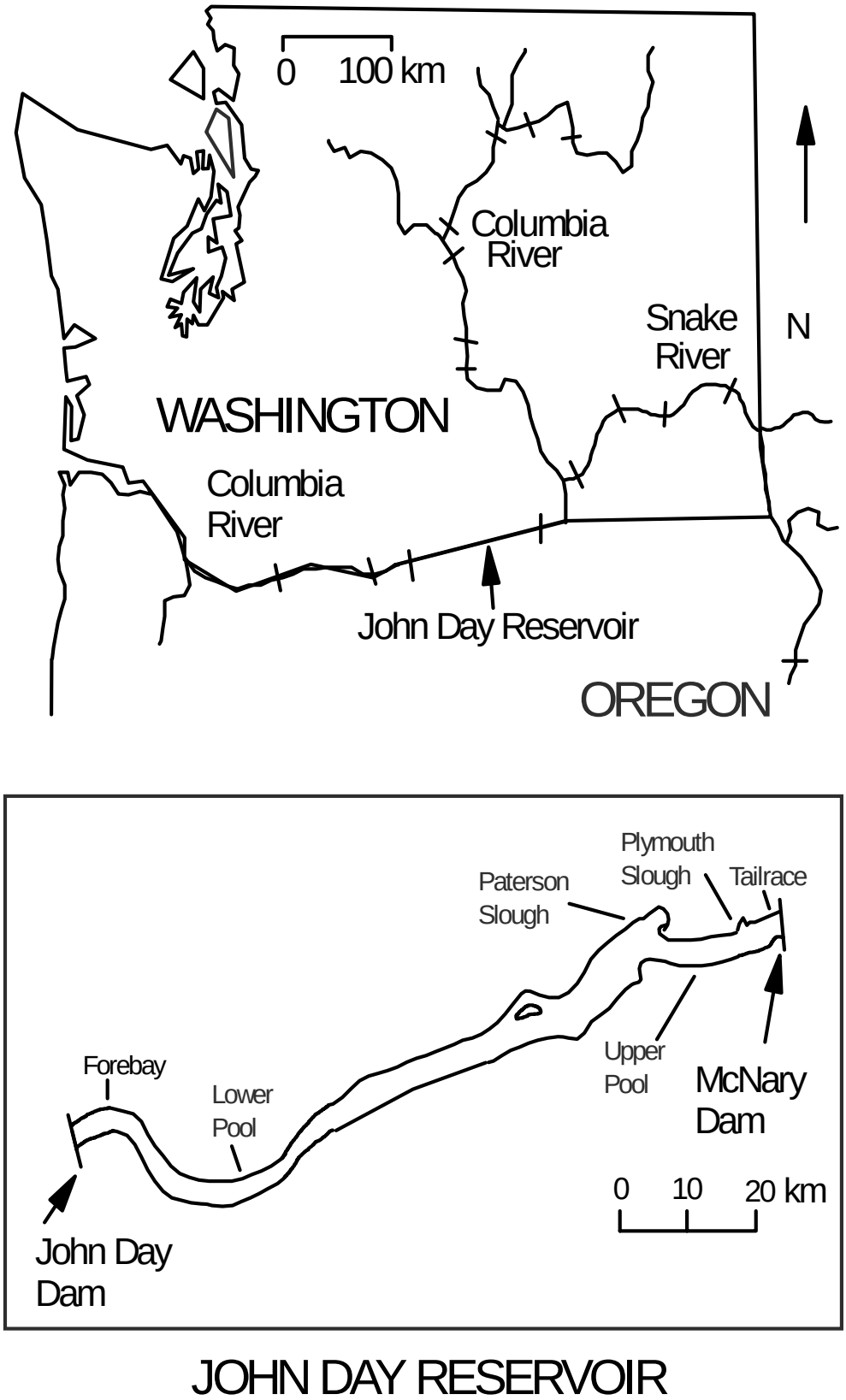


Figure 1. John Day Reservoir study area.

We sampled fishes at four shoreline sites at each of four main-channel and two backwater locations within the John Day Reservoir (Figure 1). Main-channel habitats were sampled at tailrace, forebay, and two mid-reservoir locations. One mid-reservoir location was in the upper pool and the second mid-reservoir location was in the lower pool (pool is defined as the area of the reservoir excluding the tailrace and forebay). Backwater habitats were sampled at locations in the Plymouth and Paterson sloughs (Figure 1). In 1984 and 1985, fish samples were collected on a monthly basis from April through August (except July) at each of the six major locations. During 1995 we sampled on a biweekly schedule from May through August. In 1995, we generally sampled at or nearby the same sites as in the earlier (1984-85) sample period except at some sites in the Plymouth and Paterson slough backwater locations. Some sites within these locations were no longer accessible by boat, or sampling was prevented by thick mats of aquatic macrophytes (*Myriophyllum* spp.). After mid- to late June in 1995, we were unable to sample at any site in the Paterson Slough backwater location because of water milfoil abundance. Dominant substrates at the 24 individual sample sites ranged from fine sediment to cobble.

Fish samples were collected during nighttime using 30.5 x 2.4 m beach seines with a 2.4 m^3 central bag and 6.4-mm knotless nylon mesh. Leads (15.2 m) and weighted brails were attached to both ends of the seine. To conduct a seine haul, one end of the seine was anchored to the shoreline by attaching the lead to streamside vegetation or to a metal stake driven into the ground, and the opposite lead was attached to a cleat on the bow of a boat. The seine was deployed by backing the boat

in a semicircle until water depth was insufficient for boat operation; the seine was then manually pulled to shore using the lead ropes. This method of sampling is not without biases, i.e., capture efficiency can be influenced by the habitat types being sampled, and by size- and species-specific behavioral differences. We attempted to minimize or maintain the consistency of these problems by sampling only at night, by restricting the size of the fish included in the analyses, and by sampling at the same sites across years with generally equal effort among sites within a year.

For comparison with the earlier (1984-85) data set (Gray et al. 1984; Palmer et al. 1986), only fish 250 mm or less in fork length (FL) were retained, these were measured to the nearest mm and identified to the lowest possible taxon. Most fishes were released after sampling; however, large catches of small fish were preserved in 10% formalin and returned to the laboratory for processing. Some suckers *Catostomus* spp., bullheads *Ameiurus* spp., sunfishes *Lepomis* spp., crappies *Pomoxis* spp., and sculpins *Cottus* spp. were inconsistently identified to species between the two study periods. Because of this, these fishes were grouped into their respective genera for analysis (Table 1). To describe major environmental conditions during each season we obtained daily discharge and temperature information from Columbia River Data Access in Real Time (DART; at http://www.cqs.washington.edu/dart/dart.html).

Data analysis.—We examined the similarity of resident fish assemblages at various temporal scales by calculating Spearman's rank correlation coefficients on the ordered abundances of the most commonly collected fish taxa. Anadromous fishes (American shad *Alosa sapidissima* and juvenile salmonids *Oncorhynchus* spp.) were not included

in these analyses because of large temporal variations in abundance. Rank correlation analyses are commonly used to assess temporal concordance of fish assemblages. However, the use of such analyses has been questioned because the significance of the test can vary with the number of species included in the analysis (Grossman et al. 1985). This is because the strength of the association and the significance of test can be influenced by including large numbers of rare species in the analysis; i.e., small hierarchical changes in the ranks of several uncommon species between sample periods could outweigh relatively large changes in the rank orders of more abundant species, since all species are equally important in the analysis (Grossman et al. 1982). Because of this, we restricted our analyses to the 10 overall most abundant taxa for the sample periods being compared. This confined the analyses to those taxa that were probably most important in assemblage structure (Matthews et al. 1988). Moreover, in our collections, many taxa were represented by only a few individuals and the bulk of the catch was generally dominated by a small core group of taxa. Thus, limiting the comparisons to the 10 most abundant taxa still always included an average abundance of at least 90.0% of the fishes collected for the periods being compared.

We examined seasonal, annual, and long-term continuity of assemblage structure. Seasonal concordance was assessed by comparing spring (April-May in 1984 and 1985; May in 1995) with summer (June and August in 1984 and 1985; June-August in 1995) catches to determine if temporal changes in fish habitat use or an increased abundance of young-of-the-year fishes in the summer caused large shifts in the rank orders of the most common taxa. Annual assemblage consistency was examined by comparing 1984 taxa ranks to 1985 ranks, and long-term assemblage

similarity was assessed by comparing the ranks of the pooled 1984-85 catches to the 1995 taxa ranks. Assemblages were compared at each of the six sample locations of the reservoir, except long-term changes were only examined at main-channel locations because of some backwater sample site differences in 1995. Seasonal, annual, and long-term changes in assemblage structure were also examined at a larger-scale by comparing catches pooled across all main-channel locations of the reservoir and by comparing catches pooled across both backwater locations.

We additionally examined assemblage similarities using Schoener's (1968) proportional similarity index (PSI) (Meffe and Sheldon 1990). The index varies from zero to one, with a value of zero indicating no similarity. Indices are not statistical tests, but they do provide a relative assessment of assemblage similarity (Matthews et al. 1988; Meffe and Sheldon 1990). In addition, PSI values, unlike rank order analyses, may not be as sensitive to the inclusion of rare taxa in the calculations. Thus, we computed PSI values on the entire taxa list for each of the periods being compared.

We used principal components analysis (PCA) to identify fish taxa associations and to assess assemblage composition in time and space (Rose and Echelle 1981; Moyle et al. 1986; Finger and Stewart 1987; Ross et al. 1987; Meng et al. 1994; Mahon et al. 1998; Hoff and Bronte 1999). The PCA was conducted on a correlation matrix of the 10 overall most abundant taxa; these made up an average of ~ 97% of the reservoir-wide annual catch and excluded those taxa that generally made up less than 1% of the total catch. We first averaged the individual seine haul data by month for each of the six locations. Input data were the arcsine-square root transformed proportions of each taxon for each year-month-location combination (3 years x 4

months x 6 locations). Samples were not available for the forebay and lower pool locations during April, 1984, and for the Paterson Slough location during July and August, 1995; thus, the analysis was conducted on 68 monthly samples. Fish taxa associations were identified by plotting taxa loadings on the first and second principal components (PC) (Carpenter et al. 1981). We then graphically examined temporal and spatial relations of the sample locations by plotting the scores of each year-month-location combination on the first two principal components (Finger and Stewart 1987; Ross et al. 1987).

In addition to the above analyses of temporal trends in community composition, we also examined the use of shallow littoral reservoir habitats by juvenile chinook salmon. To describe the importance of near-shore habitats to juvenile chinook salmon, we summarized monthly catch information by year for each reservoir location.

Results

Overview of resident fish catch.—During the three years of sampling, 37,400 resident fishes representing nine families and 24 taxa were collected in 359 seine hauls. Length frequency information for the ten overall (pooled across years and locations) most abundant taxa indicated that most species were typically represented by two or more year classes, except smallmouth bass, which was overwhelmingly represented by age-0 individuals in all years (Figure 2). Of the 24 resident fish taxa collected, approximately one-half (13 species) were introduced (Table 1). The overall presence and absence of individual taxa remained similar across years except speckled dace (an uncommon species in all years) was not collected in 1995, and goldfish was not collected in 1985 (Table 2). Native fishes dominated the combined main-channel catches in all three years, but to a much lesser extent in 1995. In the earlier sampling period (1984-1985) four native taxa (chiselmouth, northern pikeminnow, suckers, and sand rollers) constituted more than 90% of the combined main-channel catch, with introduced taxa comprising only 1.3%.

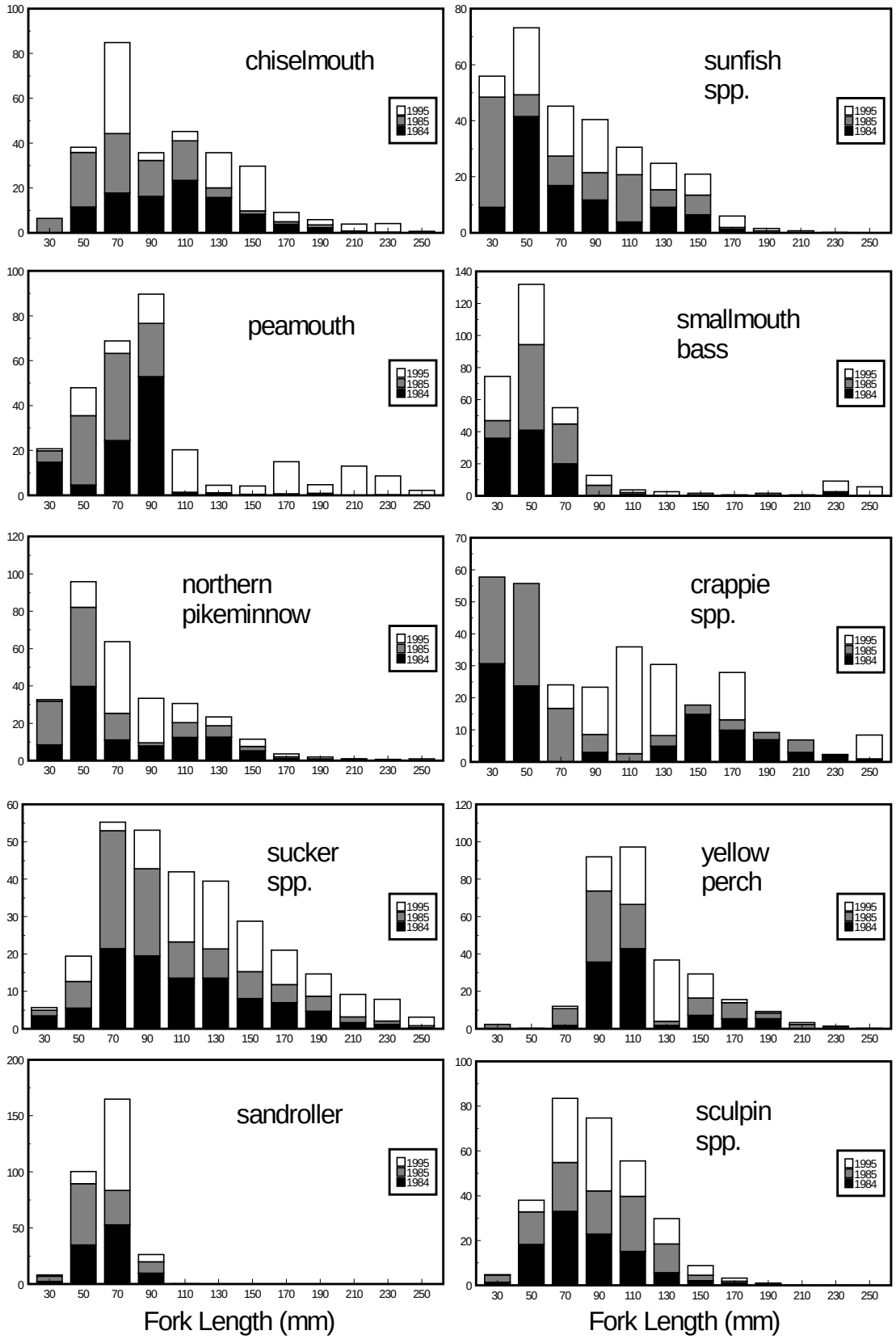


Figure 2. Length frequency distributions of the ten overall most abundant resident fishes collected by beach seining in John Day Reservoir during 1984, 1985, and 1995.

Table 1.-Scientific and common names of resident fishes collected by beach seining in the John Day Reservoir, Columbia River in 1984, 1985, and 1995.

Scientific name	Common name
Salmonidae	Trouts
Prosopium williamsoni	Mountain whitefish
Cyprinidae	Carps and minnows
Acrocheilus alutaceus	Chiselmouth
Carassius auratus	Goldfish[a]
Cyprinus carpio	Common carp[a]
Mylocheilus caurinus	Peamouth
Ptychocheilus oregonensis	Northern squawfish
Rhinichthys osculus	Speckled dace
Richardsonius balteatus	Redside shiner
Catostomidae	Suckers
Catostomus columbianus	Bridgelip sucker
Catostomus macrocheilus	Largescale sucker
Ictaluridae	Bullhead catfishes
Ameiurus natalis	Yellow bullhead[a]
Ameiurus nebulosus	Brown bullhead[a]
Ictalurus punctatus	Channel catfish[a]
Percopsidae	Trout-perches
Percopsis transmontana	Sand roller
Gasterosteidae	Sticklebacks
Gasterosteus aculeatus	Threespine stickleback
Centrarchidae	Sunfishes
Lepomis gibbosus	Pumpkinseed[a]
Lepomis macrochirus	Bluegill[a]
Micropterus dolomieu	Smallmouth bass[a]
Micropterus salmoides	Largemouth bass[a]
Pomoxis annularis	White crappie[a]
Pomoxis nigromaculatis	Black crappie[a]
Percidae	Perches
Perca flavescens	Yellow perch[a]
Stizostedion vitreum vitreum	Walleye[a]
Cottidae	Sculpins
Cottus spp.	Sculpin spp.

[a] Introduced species

Table 2.--Number of resident fishes collected at six sample locations in the John Day Reservoir, Columbia River in 1984, 1985, and 1995. N = number of seine hauls.

Common name	Forebay	Lower pool	Upper pool	Tailrace	Paterson Slough	Plymouth Slough
		1984				
	(N = 14)	(N = 14)	(N = 14)	(N = 18)	(N = 14)	(N = 18)
Mountain whitefish	5	0	2	9	0	0
Chiselmouth	1,729	24	26	26	4	82
Goldfish	0	0	8	1	0	0
Common carp	0	2	0	0	2	1
Peamouth	1	0	56	22	6	1,152
Northern pikeminnow	124	22	337	99	88	1,619
Speckled dace	2	6	0	0	0	0
Redside shiner	8	2	3	67	1	89
Suckers	1,282	279	773	320	126	1,744
Bullheads	0	0	0	0	1	9
Channel catfish	0	0	0	0	1	0
Sand roller	5	67	801	568	23	80
Threespine stickleback	0	0	0	6	0	25
Sunfishes	0	5	1	0	34	41
Smallmouth bass	3	6	3	3	42	66
Largemouth bass	1	0	3	0	18	61
Crappies	0	1	16	0	18	61
Yellow Perch	0	2	2	0	32	20
Walleye	0	1	0	0	0	0
Sculpins	153	27	21	40	36	36
		1985				
	(N = 16)	(N = 16)	(N = 15)	(N = 15)	(N = 16)	(N = 16)
Mountain whitefish	2	1	2	0	0	0
Chiselmouth	241	76	16	14	3	134
Goldfish	0	0	0	0	0	0
Common carp	0	3	3	9	4	13
Peamouth	2	6	79	155	4	346
Northern pikeminnow	5	34	469	789	75	591
Speckled dace	0	0	3	4	0	2
Redside shiner	1	5	1	29	0	132
Suckers	567	481	978	512	161	2,680
Bullheads	0	3	0	0	5	124
Channel catfish	0	2	0	0	1	0
Sand roller	27	198	1,763	566	152	166
Threespine stickleback	0	0	0	1	0	3
Sunfishes	0	6	15	2	390	176
Smallmouth bass	5	7	16	8	65	65
Largemouth bass	1	0	1	8	76	149
Crappies	0	0	9	0	971	850
Yellow Perch	0	1	13	6	246	19
Walleye	2	1	0	0	0	0
Sculpins	93	12	60	24	56	22

Table 2. continued--Number of resident fishes collected at six sample locations in the John Day Reservoir, Columbia River in 1984, 1985, and 1995. N = number of seine hauls.

Common name	Forebay	Lower pool	Upper pool	Tailrace	Paterson Slough	Plymouth Slough
		1995				
	(N = 34)	(N = 34)	(N = 34)	(N = 34)	(N = 9)	(N = 28)
Mountain whitefish	0	2	0	1	0	0
Chiselmouth	48	22	10	58	2	121
Goldfish	0	15	2	1	0	1
Common carp	0	3	1	0	0	1
Peamouth	123	33	122	186	7	460
Northern pikeminnow	6	19	207	151	61	1,058
Speckled dace	0	0	0	0	0	0
Redside shiner	0	1	0	8	0	2
Suckers	52	265	110	261	99	219
Bullheads	0	89	1	1	1	3
Channel catfish	0	0	3	0	0	0
Sand roller	13	17	655	310	0	335
Threespine stickleback	0	0	0	0	1	1
Sunfishes	21	321	63	14	135	28
Smallmouth bass	33	48	33	12	5	227
Largemouth bass	0	2	0	4	0	16
Crappies	1	17	2	2	4	1
Yellow Perch	3	332	733	211	428	406
Walleye	2	9	1	6	0	12
Sculpins	464	169	257	297	20	123

In contrast, during 1995, chiselmouth, northern pikeminnow, suckers, and sand rollers comprised only 37.7% of the main-channel catch, while introduced taxa made up 33.9%. The greater percentage of introduced fish in the 1995 catch was primarily due to an increase in the relative abundance of yellow perch and sunfishes (Table 2). In backwaters (catches pooled across both backwaters) the percentage of introduced fishes remained relatively similar between the earlier sampling period (28.4%) and 1995 (33.6%). The dominant introduced taxa, however, differed between periods, with crappies dominating backwater catches in the earlier period, and yellow perch comprising the greatest proportion in 1995.

General patterns of assemblage composition also varied among individual sample locations. In all years, fish abundance and species richness were highest at Plymouth slough, and lowest at the forebay location, which was characterized by catches of only a few numerically dominant taxa (Table 2). Although patterns of relative abundance varied somewhat between locations, during 1984 and 1985, a core group of native taxa (cyprinids, suckers and sand rollers) dominated the catch at all individual locations except Paterson slough, where introduced sunfishes, smallmouth bass, crappies, and yellow perch made up a large percentage of the catch. Conversely, during 1995, introduced taxa, particularly yellow perch, comprised a larger percentage of the catch than in the earlier (1984-85) sample period at nearly all main-channel sample locations. This shift in catch composition was most dramatic at the lower pool location where introduced taxa constituted 61% of the 1995 catch (Table 2).

Comparisons of assemblage structure.—Catch composition did not vary greatly

between seasons in main-channel habitats. Agreement between spring and summer resident fish taxa ranks of main-channel catches (pooled across locations) was strong (range of Spearman's r = 0.76-0.97) and significant ($P < 0.05$) during all three years of the study (Table 3). Likewise, taxa ranks were concordant among seasonal catches at individual main-channel locations except at the lower pool in 1984 and the forebay in 1995. In contrast to the general pattern at main-channel locations, correlations between seasonal taxa ranks in backwaters (catches pooled across both backwater locations) were low and nonsignificant except during 1995 (Table 3). Similarly, assemblages changed markedly (Spearman's r, range = -0.18 - 0.38; $P > 0.05$) at individual backwater locations between spring and summer in both 1984 and 1985, but not in 1995 (Table 3). In backwaters during 1984 and 1985 there were consistent negative changes in the rank positions of native taxa (e.g., northern pikeminnow, peamouth, suckers, and sand roller) and positive changes in the rank positions of introduced taxa (e.g., sunfishes, smallmouth bass, and crappies) from spring to summer.

Proportional similarity indices (PSI) exhibited moderate agreement with rank correlation measures of assemblage similarity at the seasonal level of comparison (Table 3). Six of 15 significant rank correlations had PSI values of < 0.70, although only two of these were < 0.62, indicating at least moderate similarity for nearly all seasonal comparisons. An additional three comparisons lacked significant concordance among ranks, but had relatively high (> 0.70) PSI values. These differences are possibly a function of ranking the catch data in the statistical measure of concordance.

Differences between PSI values and measures of rank association may have resulted from disparities between the hierarchical changes in individual taxa ranks and their proportional contributions to the assemblage, or differences may be due to the inclusion of numerically uncommon taxa in the rank analyses. Annual comparisons of main-channel fish faunal composition revealed few differences between 1984 and 1985. The correlation between rank orders (catches pooled across main-channel locations) was 0.85 ($P < 0.05$), indicating strong agreement between years (Table 4).

Annual comparisons of assemblage structure at individual main-channel locations also demonstrated strong agreement between the two years (1984-1985; Table 4). Backwater faunal composition, in contrast, was somewhat dissimilar between the two years at individual locations (Table 4). Ranks of annual catches pooled across both backwaters had a low ($r = 0.56$) but significant correlation. In both years, suckers dominated backwater catches—differences in catch composition (catches pooled across both backwaters) were largely due to hierarchical changes in the rank abundances of introduced sunfishes, smallmouth bass, and yellow perch.

Table 3.-Seasonal comparisons (spring and summer) of fish assemblages in the John Day Reservoir, Columbia River in 1984, 1985, and 1995. Agreement between spring and summer assemblages was calculated with Spearman's rank analysis on the rank ordered abundances of the 10 overall most abundant taxa for each comparison. Proportional similarity indices (PSI) were calculated on the catch of all taxa. * = P < 0.05

Year and location	Spearman's correlation coefficient	PSI
1984		
Forebay	0.86*	0.83
Lower pool	0.25	0.45
Upper pool	0.82*	0.77
Tailrace	0.80*	0.62
Paterson slough	0.22	0.29
Plymouth slough	0.33	0.76
(Pooled data)		
All main-channel locations	0.97*	0.82
All backwater locations	0.37	0.71
1985		
Forebay	0.80*	0.95
Lower pool	0.71*	0.73
Upper pool	0.87*	0.68
Tailrace	0.81*	0.36
Paterson slough	-0.18	0.22
Plymouth slough	-0.07	0.28
(Pooled data)		
All main-channel locations	0.85*	0.66
All backwater locations	-0.21	0.22
1995		
Forebay	0.54	0.70
Lower pool	0.91*	0.76
Upper pool	0.92*	0.56
Tailrace	0.77*	0.69
Paterson slough	0.78*	0.77
Plymouth slough	0.72*	0.71
(Pooled data)		
All main-channel locations	0.76*	0.73
All backwater locations	0.73*	0.79

Long-term comparisons of fish taxa ranks indicated that main-channel shoreline fish faunal structure changed dramatically between the earlier period (pooled 1984-85 catch) and 1995. Large changes in catch composition were indicated by low ($r \leq 0.47$) and nonsignificant correlations ($P > 0.05$) at all levels of main-channel comparison (Table 4). Major shifts in faunal composition between the two periods (catches pooled across all main-channel locations) resulted from declines in the relative (rank) abundance of native suckers and cyprinids (chiselmouth and northern pikeminnow) and increases in ranks of sculpins, and introduced sunfishes and yellow perch. Similar long-term changes in taxon contributions to assemblage structure occurred at individual main-channel locations.

Annual and long-term PSI values were generally very consistent with rank concordance measures of assemblage similarity (Table 4). Annual comparisons of assemblage structure with significant Spearman's rank correlations always had PSI values of ≥ 0.65, whereas most long-term comparisons of main-channel assemblage structure (all correlations were nonsignificant) had PSI ≤ 0.5. Proportional similarity index values, however, indicated higher similarities at individual backwater locations between 1984 and 1985 than did rank concordance measures of annual similarity, which were low and nonsignificant.

Table 4.-Annual (1984 compared to 1985) and long-term (pooled 1984-85 compared to 1995) comparisons of fish assemblages in the John Day Reservoir, Columbia River. Agreement between assemblages was calculated with Spearman's rank correlation analysis. Analyses were conducted on the rank ordered abundances of the 10 overall most abundant taxa for each comparison. Proportional similarity indices (PSI) were calculated on the catch of all taxa.

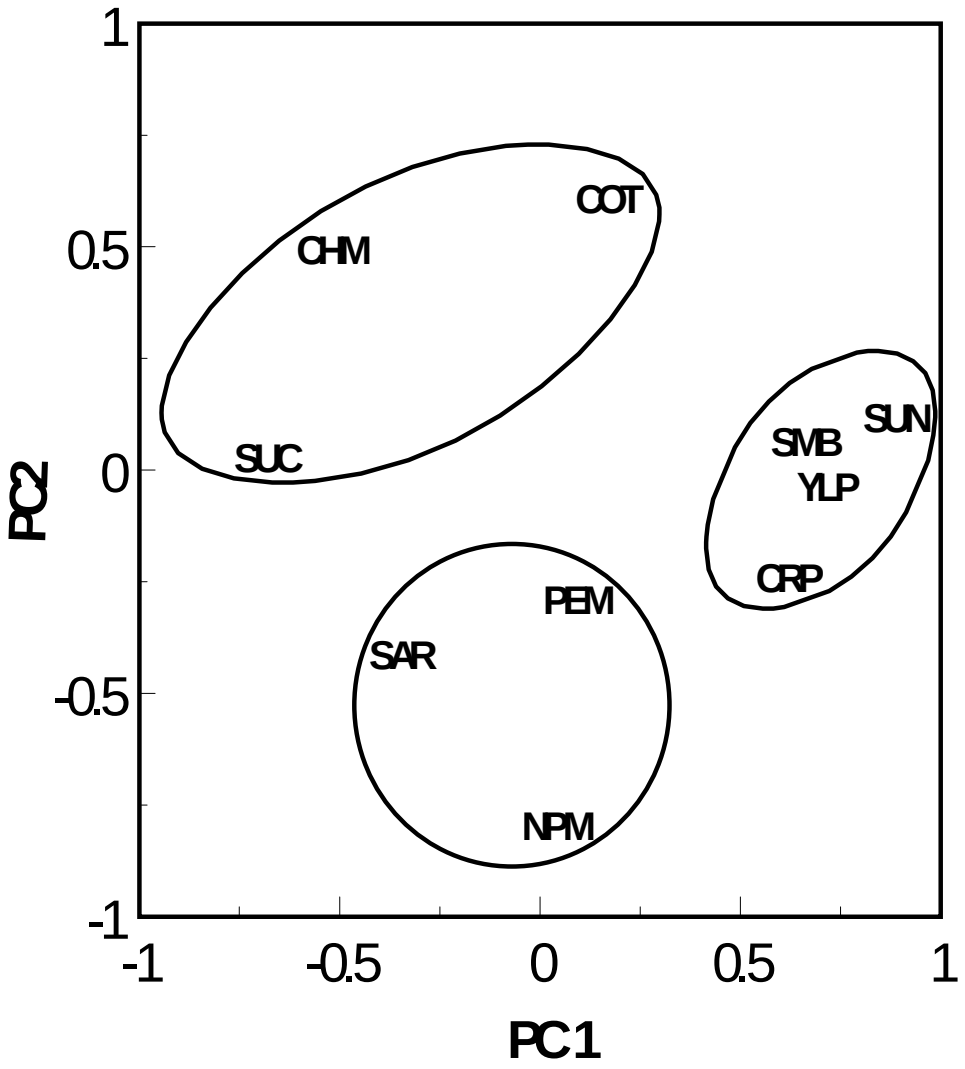
Time period and location being compared	Spearman's correlation coefficient	PSI
Annual		
Forebay	0.74*	0.70
Lower pool	0.85*	0.86
Upper pool	0.89*	0.86
Tailrace	0.84*	0.65
Paterson slough	0.38	0.74
Plymouth slough	0.43	0.61
(Pooled data)		
All main-channel locations	0.85*	0.73
All backwater locations	0.56*	0.63
Long-term		
Forebay	0.29	0.21
Lower pool	-0.44	0.30
Upper pool	0.24	0.57
Tailrace	0.47	0.50
(Pooled data)		
All main-channel locations	0.07	0.44

* $P < 0.05$

Ordinations of taxon associations and sample location relationships.—A plot of fish taxa loadings on the first two principal component axes revealed three relatively well-defined, ecologically interpretable fish taxa associations (Figure 3). The first component explained 30% of the variance in the relative abundance data and separated native (negative or low loadings on PC 1) and introduced taxa (loadings $\geq$ 0.60 on PC 1). The second component accounted for 21% of the variation in the data set and contrasted taxa (high positive loadings on PC 2) which were generally

associated with hard substrate (gravel-cobble) habitats from taxa associated with soft (fines-sand) sediments. The three major groups of taxa were a native chiselmouth-sucker-cottid association (at the upper left of the figure), an association of the native northern pikeminnow, peamouth, and sand roller, and a third group of introduced sunfishes (pumpkinseed and bluegill), smallmouth bass, crappies, and yellow perch.

An ordination of monthly (by year) location scores on the first two principal components exhibited both spatial and temporal (seasonal and annual) trends in clusters of sampling units. Temporal patterns in groupings of monthly samples concur closely with the seasonal, annual, and long-term measures of concordance (Figure 4). There was generally very little separation of monthly location scores from within a year at individual main-channel locations. Monthly scores at individual main-channel locations also did not differ greatly between 1984 and 1985; however, 1984 and 1985 location scores typically differed substantially from 1995 scores. Monthly backwater scores showed an overall greater amount of variability than main-channel scores, both within a year and between years, which is consistent with the rank measures of assemblage similarity (Tables 3 and 4).



CHM = chiselmouth	SUN = sunfish spp.
PEM = peamouth	SMB = smallmouth bass
NPM = northern pikeminnow	CRP = crappies spp.
SUC = sucker spp.	YLP = yellow perch
SAR = sand roller	COT = sculpin spp.

Figure 3. A plot of fish taxa loadings on the first two axes of a principal components analysis of fish relative abundance data from the John Day Reservoir during 1984, 1985, and 1995.

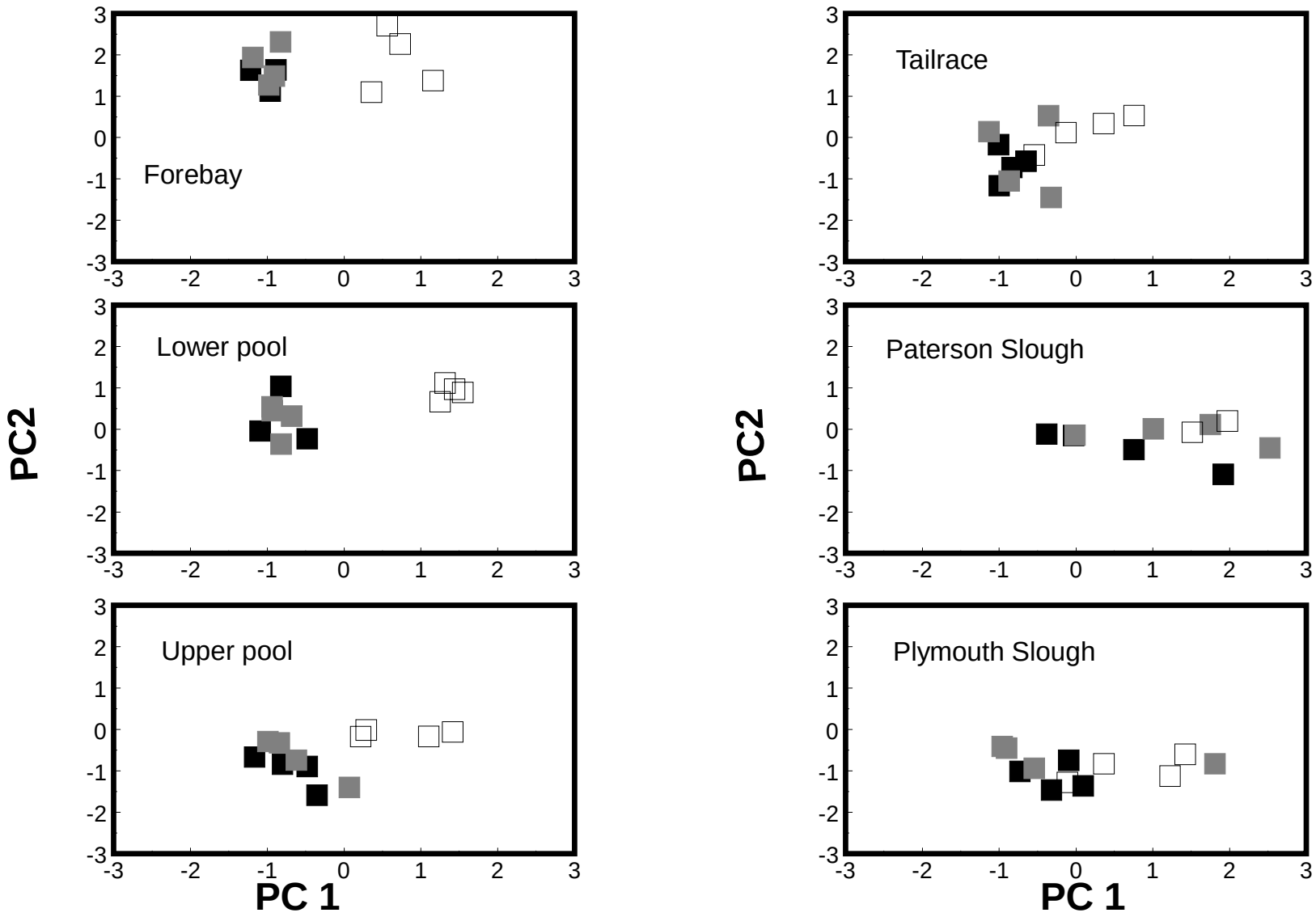


Figure 4. An ordination of monthly location scores based on a principal components analysis of fish relative abundance data for John Day Reservoir during 1984 (black squares), 1985 (gray squares), and 1995.

The fish assemblages present at a location influenced the groupings of location scores (Figures 3 and 4). For example, introduced taxa had comparatively high positive loadings on PC 1; thus, monthly samples with high relative abundances of introduced taxa have higher scores on PC 1. Samples from Paterson Slough generally had the highest positive scores on the first component, particularly during August in 1984 and 1985, and May and June 1995 (we were unable to sample in this area after June, 1995), indicating the importance of this habitat to introduced taxa (e.g., centrarchids and yellow perch). In Plymouth Slough, high relative abundances of young-age classes of northern pikeminnow and peamouth contributed to relatively large negative monthly scores on PC 2, regardless of year (Figure 4). An increased relative abundance of introduced taxa in Plymouth Slough during August 1985, and during July and August 1995, shifted location scores to the right of the ordination. Positive location scores on PC 1 in 1995 indicate the increasing importance of introduced taxa to assemblage structure at nearly all main-channel locations. The main-channel area with the highest average monthly PC 1 scores in 1995 was the lower pool location, where more than half of all fishes collected were introduced. The main-channel area with the lowest average scores on PC 1 in 1995 was the tailrace location. Monthly tailrace location ordination scores also exhibited the least amount of separation from 1984-1985 ordination scores; this area additionally had the highest, but still low (r = 0.47), measure of concordance over the long term sampling period (Table 4).

Juvenile chinook salmon.—During 1984, 1985, and 1995, a total of 1029, 674, and 1007 juvenile chinook salmon, respectively, were collected at the six major John Day

Reservoir sample locations. Highest numbers of juvenile chinook salmon were generally collected in shallow littoral habitats of the upper main-channel reservoir and in near-shore backwater habitats (Table 5). The timing of peak catches varied between locations and years, but highest average catches were always between April and June (Table 5). In August, juvenile chinook salmon were either absent (1995), or present in extremely low numbers (mean catch < 2.0 fish per haul) in shoreline habitats. The size of juvenile subyearling chinook salmon in our catch ranged from 26 mm FL in early May to 141 mm FL in late August.

Water temperature and discharge patterns.—Temperature and discharge conditions in 1995 were generally intermediate between 1984 and 1985 (Figure 5). Discharge was lowest in 1984, and highest in 1985, with peaks in the hydrograph occurring in all years during late May or early June. Water temperatures and discharge were inversely related. Temperatures were highest in 1985, lowest in 1984, and intermediate in 1995; however, water temperatures early in 1995 were slightly warmer (typically $\leq$ 1.5 C) than temperatures in 1985 until early June. Within a year, main-channel temperatures were highest in late July or early August, with peaks of 22.2 to 25.6 C depending on the year.

Table 5. Mean (SD) catch per seine haul of juvenile chinook salmon in the John Day Reservoir 1985-85 and 1995. In 1984 and 1985 effort (number of seine hauls) was normally four hauls per month at each of the six sample locations. In 1995 effort was typically eight hauls per month at each location.

Year and month	Forebay	Lower pool	Upper pool	Tailrace	Paterson Slough	Plymouth Slough
1984						
April			28.5	10.8	0.0	10.3
			(2.1)	(12.2)		(9.0)
May	0.5	9.5	23.8	16.0	9.0	28.0
	(1.0)	(7.1)	(21.0)	(11.5)	(8.3)	(22.0)
June	9.2	17.0	14.8	43.3	7.8	21.3
	(3.7)	(16.8)	(16.9)	(56.0)	(10.2)	(13.4)
August	0.0	0.3	0.0	1.8	0.0	0.0
		(0.5)		(2.2)		

1985						
April	5.0	2.5	3.5	13.3	1.8	4.5
	(3.8)	(2.5)	(1.7)	(13.0)	(1.3)	(2.4)
May	0.8	4.5	8.0	7.8	8.9	6.5
	(1.5)	(3.4)	(2.6)	(7.8)	(14.9)	(6.8)
June	1.5	7.0	70.0	13.0	2.5	9.8
	(1.3)	(7.7)	(91.1)	(6.0)	(3.0)	(6.3)
August	1.0	0.3	0.0	0.0	0.0	0.0
	(2.0)	(0.5)				
1995						
May	12.4	14.0	7.4	18.4	7.0	27.3
	(4.1)	(20.7)	(4.4)	(21.7)	(3.6)	(10.2)
June	5.9	7.4	9.9	9.1	3.2	19.6
	(5.2)	(9.4)	(6.6)	(13.5)	(1.6)	(18.0)
July	0.1	0.0	0.5	1.4		0.0
	(0.4)		(0.5)	(1.4)		
August	0.0	0.0	0.0	0.0		0.0

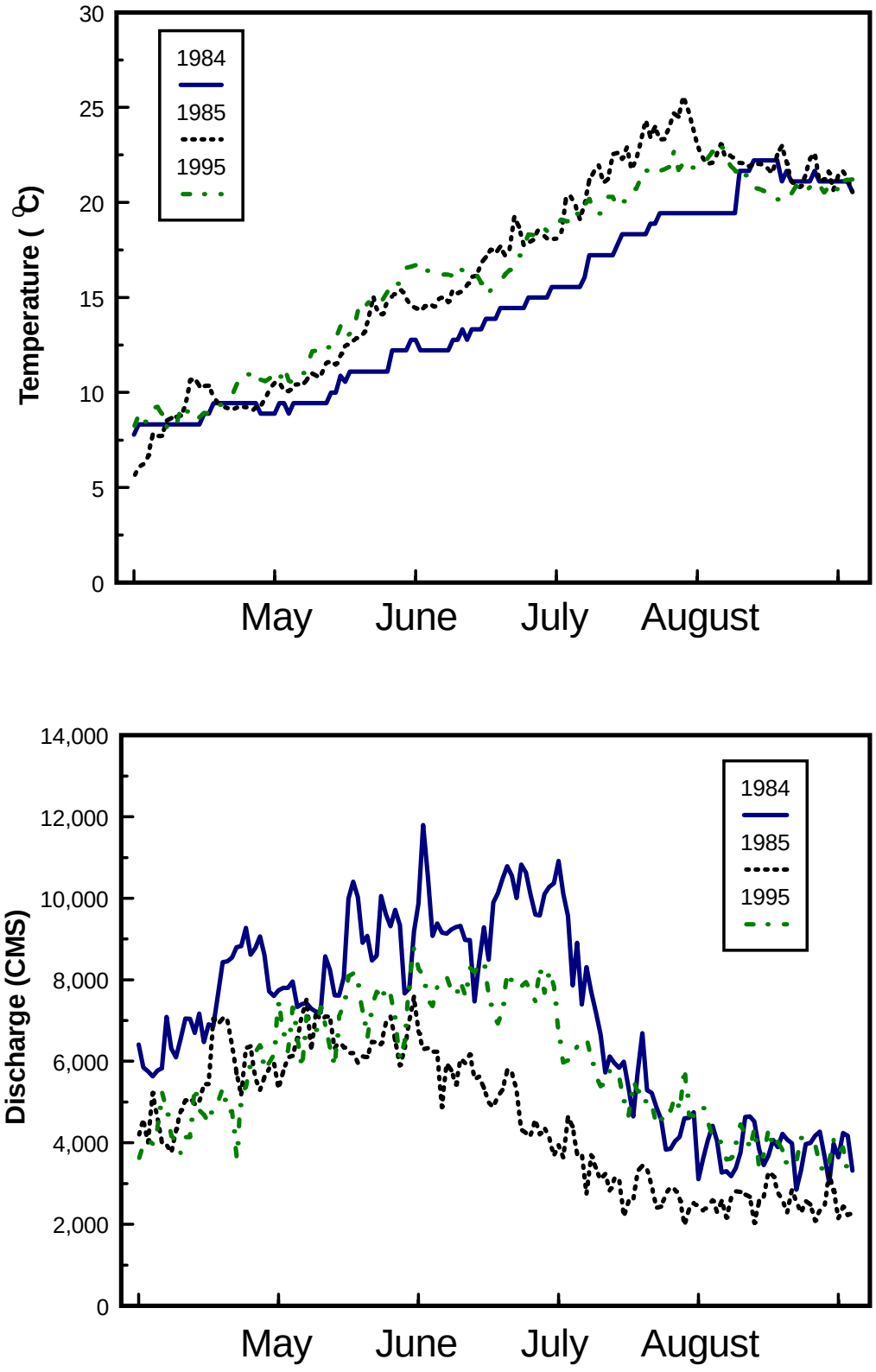


Figure 5. Temperature and discharge patterns during April through August, 1984, 1985, and 1995 in John Day Reservoir.

Discussion

Our primary objectives were to describe resident fish community structure in shallow near-shore habitats of John Day Reservoir and to determine if existing data from different time periods could be used collectively as a baseline for comparison with possible future Phase II studies. Our analyses suggest that the use of existing resident fish data as a baseline may be limited because of long-term differences in assemblage structure. Fish catch composition between 1984 and 1985 was remarkably similar, but was very different from 1995 catches, possibly indicating a shift in the composition of the near-shore reservoir fish community.

We are uncertain, however, about inherent annual variability in John Day Reservoir near-shore resident fish communities because we were restricted to only one season of sampling for comparison with the earlier study. Similar studies in the basin are unavailable for comparison with ours, and there is a general paucity of information on both the temporal variability and structure of shallow-water fish assemblages in impoundments. Differences in the John Day Reservoir fish community could be due to either short-term variation or long-term changes. For example, differences in catch composition between the earlier sample period and 1995 might be relatively recent, due to variations in the reproductive success of some taxa because of abiotic conditions (e.g., temperature and discharge regimes) present in the years immediately prior to 1995. Alternatively, the decadal changes observed in shoreline fish assemblages at our study sites may indicate a more persistent shift in the structure of the near-shore fish community in response to long-term habitat changes (e.g., due to impoundment

aging).

Although we are uncertain about the inherent annual variability of John Day Reservoir shallow-water fish communities, the trends we observed in 1995 (i.e., an increasing abundance of introduced species more typically associated with lake-like habitats) are consistent with long-term post-impoundment patterns of community change described by Li et al. (1987) for the Snake and Columbia river basins, suggesting that persistent changes may have occurred in the reservoir near-shore fish community. Similar shifts in post-impoundment fish community composition have also been described elsewhere; for example, Martinez et al. (1994) studied the fish community in a Colorado Reservoir and determined that species composition changed substantially over a six-year period from a community dominated by native riverine species to one primarily comprised of introduced taxa adapted to lake-like conditions.

In this study we identified three assemblages or covarying groups of fish taxa in the John Day Reservoir, with varying contributions to the overall catch at individual locations and between different sampling periods. Two fish taxa associations were comprised of native fishes, and a third group consisted of introduced taxa (Figure 3). The two native taxa assemblages (on the left side of Figure 3) dominated the catch in the earlier 1984-1985 sampling period and appeared to be separated along a gradient of hard versus soft substrate types. The hard substrate assemblage was comprised of chiselmouth, suckers, and sculpins (positive scores on PC 2; Figure 3), with chiselmouth and sculpins having the highest positive scores on PC 2. The chiselmouth is a native herbivorous cyprinid that is strongly associated with hard substrates from

which it scrapes plant material (Moodie and Lindsey 1972), and sculpins as a group are lithophilic. Chiselmouth and sculpins were typically most abundant in cobble habitats of the forebay sample location (Table 2; Figure 4).

The second native species association (negative loadings on PC 2) was most abundant over soft substrates (silt and sand) and contained two cyprinids (northern pikeminnow, peamouth) and the sand roller. Young age classes of northern pikeminnow and peamouth rear in sandy low-gradient habitats (Beamesderfer 1992), while sand rollers are particularly abundant over sandy bottoms at nighttime (Gray and Dauble 1976). Seine sites in the Plymouth Slough and some upper pool sample locations were characteristic of the habitat type supporting the highest relative abundance of species in the soft substrate native species group (Table 2; Figures 3 and 4).

The third group of covarying species (on the right side of Figure 3) was comprised of introduced sunfishes (bluegill, pumpkinseed), smallmouth bass, crappies (white and black), and yellow perch, and was separated from the native species along the first PC. These cool and warmwater taxa were generally not abundant in the earlier sampling period, except in the Paterson Slough backwater (Table 2; Figure 4), but became increasingly more important to fish community composition at nearly all main-channel locations in 1995 (Figure 4). All of these taxa are typical of lake-like habitat conditions, with the exception of smallmouth bass which occurs in a variety of both still and moving water habitats.

Differences in habitat conditions may help explain both seasonal and long-term

variation in assemblage composition in John Day Reservoir. Although main-channel assemblage structure was generally similar across seasons within a year, rank correlation analyses indicated that assemblages changed in backwaters between spring and summer in both 1984 and 1985 (Table 3). During 1984 and 1985 there were consistent negative changes in the rank positions of native taxa (e.g., northern pikeminnow, peamouth, suckers, and sand roller) and positive changes in the rank positions of introduced taxa (e.g., sunfishes, smallmouth bass, and crappies) from spring to summer. This pattern of change agrees closely with the seasonal differences in slough assemblages shown by the multivariate analysis (Figure 4). Monthly samples from Paterson Slough had much higher scores on PC 1 during August in 1984 and 1985, indicating a shift towards a greater proportion of introduced taxa in late summer. An increased relative abundance of introduced taxa in Plymouth Slough during August 1985, and during July and August 1995, also shifted monthly scores to the right of the ordination at this location.

The seasonal increase in the proportion of introduced taxa in backwaters during summer months may be related to temperature regimes in these areas. Both slough locations are shallow, vegetated, and have low water exchange rates relative to main-channel areas. Because of these conditions, summer water temperatures in backwater locations can be several degrees higher than nearby main-channel habitats (Gadomski and Barfoot 1998). Temperature has been previously shown to influence the distribution of fishes in reservoir habitats (Hall and Werner 1977; Tufescu 1994; Garret and Bennett 1995). Higher late summer temperatures may have caused native taxa to

shift habitat use to offshore areas or out of backwaters, while introduced warmwater taxa may have moved into these areas for reproductive purposes or because sloughs had temperatures nearer to optimal for those taxa. Main-channel temperatures in August ranged between 19 and 24 C, but slough habitats can be 2-5 C higher in this area (Gadomski and Barfoot 1998), which are closer to optimal temperatures reported for some species (e.g., bluegill, smallmouth bass) in the introduced taxa assemblage (Horning and Pearson 1973; Lemke 1977).

Long-term differences in the shallow-water fish community of John Day Reservoir between the two sample periods (1984-85 and 1995) may be due to simple year-to-year variation, decadal environmental differences, or to persistent habitat changes related to reservoir aging. At our scale of examination, fish catch composition between 1984 and 1985 was remarkably similar, but was very different from 1995 catches. The three years of sampling do not allow us to assess year-to-year variation in any detail, however. Additionally, similar studies in the basin are unavailable for comparison with ours, and there is a general paucity of information on both the temporal variability and structure of shallow-water fish assemblages in impoundments.

General environmental or climatic differences between the 1980's and 1990's could be responsible for the changes that we observed in the John Day Reservoir fish fauna. Discharge and temperature regimes, in particular, can influence annual variation in reservoir fish communities by differentially influencing year class strengths of individual species or groups of species with similar life histories. For example, Kallemeyn (1987) found a positive relationship between percid year-class strengths and

thermal regimes, with the strongest year classes being produced in years with comparatively high, stable water temperatures. Elevated turbidities (Mitzner 1991) and high reservoir flushing rates (Walburg 1971, Beam 1983, Maceina & Stimpert 1998), both correlates of high reservoir inflows, can be negatively related to crappie recruitment success.

Spring and summer water temperatures (Quinn et al. 1997), turbidities, and reservoir water-retention rates all covary with discharge in the Columbia River and could thus directly or indirectly affect reproductive success of many fishes in main-stem reservoirs. Discharge regimes associated with the two sample periods were very dissimilar in the Columbia River (Figure 5). Conditions leading up to and encompassing the earlier 1984-1985 sampling period were primarily characterized by above average annual discharge, but in contrast, average annual discharge for the later interval was generally below the long-term average (Figure 5). Thus, several successive years of low precipitation and runoff in the early 1990's might have favored the reproductive success of many introduced taxa in the John Day Reservoir, while the high discharge regime during the early 1980's may have had the opposite influence.

Alternatively, the long-term changes that we observed in fish assemblages may be more persistent and less related to short-term climatically influenced variations in reproductive success. The abundance of aquatic macrophytes in John Day Reservoir, a habitat feature likely associated with reservoir aging, is one plausible mechanism that may have influenced fish assemblage composition over the long term. An increase in the abundance of aquatic macrophytes in impoundments over time is a predictable

outcome of siltation due to both anthropogenic induced and natural reservoir aging processes (Kimmel & Groeger 1986). We observed an abundance of aquatic macrophytes in many shallow areas of John Day Reservoir during sampling in 1995, whereas investigators who conducted the earlier sampling (1984-1985) noted few macrophytes. In Paterson Slough, for example, macrophyte abundance (primarily Myriophyllum spp.) prevented us from sampling with a beach seine after mid-summer in 1995. Rooted aquatic vegetation can play a key role in altering the ecology of shallow habitats by changing water circulation patterns, DO, pH, and the temperature of localized areas (Carter et al. 1991). In Columbia River impoundments, these changes may favor non-native lentic taxa over native riverine species. Macrophyte beds can create structurally complex spawning, rearing, and foraging habitats (Kilgore et al. 1989, Weaver et al. 1997) for many of the introduced fish species (e.g., bluegill and pumpkinseed, black crappie, and yellow perch) present in John Day Reservoir. This may in part provide an explanation for the greater influence of introduced taxa in reservoir community composition in 1995 compared to 1984-85.

It is uncertain how these potential long-term changes in near-shore habitats and fish communities might affect juvenile chinook salmon rearing in the John Day Reservoir. Shallow littoral habits provide important rearing areas for fall chinook salmon. Drawing down John Day Reservoir would greatly disrupt shallow-water habitats for a period until substrate stability and a more riverine food web developed. A drawdown would also likely result in the loss or modification of a reservoir-based food web; in particular, changes from a drawdown would presumably alter zooplankton

diversity and abundance, especially in sloughs where water exchange rates are lower than main-channel areas. Food availability in Columbia River reservoirs is especially critical to fall chinook salmon, since they spend a protracted amount of time in reservoirs compared to other chinook salmon races (Rondorf et al. 1990). A food habits study conducted by Rondorf et al. (1990) found that in impounded reaches of the Columbia River, zooplankton (*Daphnia* spp.) and terrestrial insects comprised a large percentage of juvenile fall chinook salmon diets; conversely, in riverine reaches subyearling chinook salmon diets consisted largely of caddisflies. Rondorf et al. (1990) postulated that the observed diet differences were primarily due to differences in food availability between the habitats. In addition to altering reservoir zooplankton food webs, a drawdown might reduce terrestrial insect input until reestablishment of riparian vegetation. However, potential harmful affects to juvenile salmonids associated with short-term changes in food availability resulting from drawdown may be offset by other survival benefits.

Further studies are necessary to determine to what extent food webs in John Day Reservoir differ from those available to juvenile chinook salmon in riverine areas (e.g., the Hanford Reach). The presence of small (26-60 mm FL) subyearling chinook salmon in our samples indicates that shallow areas of John Day Reservoir are used for early rearing habitat by wild fish. However, more information is needed to determine the relative growth and survival of juveniles reared from a small size in John Day Reservoir compared to juveniles that rear for a greater amount of time in more riverine areas (e.g., Hanford Reach). Do juvenile chinook salmon rearing in the reservoir grow

similarly to those feeding on a riverine food web, and are juveniles that spend comparatively greater amounts of time in the reservoir environment more vulnerable to introduced predators?

Drawing down John Day Reservoir may partially restore riverine conditions and natural fish communities in the reach, and this may ultimately benefit juvenile salmonids. Introduced taxa represent only a minor component of the fish community in the Hanford Reach (Li et al. 1987), an area with highly productive fall chinook salmon spawning and rearing habits. It is possible that long-term habitat and fish community changes in the John Day Reservoir since impoundment could be affecting the suitability of reservoir rearing habitats for juvenile chinook salmon. Many exotic fish species now abundant in shoreline and backwater areas of the John Day Reservoir (e.g. cyprinids, ictalurids, centrarchids, and percids) could directly prey upon or compete with juvenile salmonids, although these relationships have not been studied. Earlier predation studies in John Day Reservoir did not identify the apparently increasingly abundant sunfishes, crappies, or yellow perch as important predators of juvenile salmonids. However, these relationships may differ with changing habitat quality and fish community composition. For example, juvenile salmonids comprised 20-25% of yellow perch and crappie diets in some locations of the impounded lower Snake River, an area with large numbers of introduced species (Bennett 1999). Further research is necessary to better understand the complex relationships between reservoir food webs and habitat conditions, introduced taxa, and juvenile salmonids to more fully understand the possible effects of a drawdown.

Additional Phase I and Phase II study needs:

- Establishment of a long-term monitoring program for the John Day Reservoir to determine baseline information on variability in near-shore fish communities, abundance and distribution of benthic macroinvertebrates, and other reservoir biotic parameters (e.g., abundance and distribution of aquatic vegetation, juvenile salmonid predation rates by resident fish predators, growth and abundance of age-0 predators) that might be most influenced by a future reservoir drawdown.

- Development of predictive models for predator-prey dynamics (see Petersen et al. 1999) and salmonid rearing, spawning, and predator habitat availability under different water-level management scenarios. This would largely be a modeling exercise (using existing data), possibly with some additional data needs, such as information from telemetry studies to describe behavior and habitat use by juvenile salmonids and predators in mid-reservoir areas. These data, along with resource selection models developed using information collected in riverine habitats, would be used to model habitat use and habitat overlap between predators and juvenile salmonids under different dam operational scenarios.

- Improved resolution in substrate data for the impounded reach.

- Continued sampling of John Day Reservoir fish communities to allow quantification of annual and long-term variability. We recommend a multi-year approach using a stratified (by habitat features) random sampling design, possibly with several gear types.

- Study the importance of riparian habitats to food production in reservoirs versus free-flowing reaches.

- Investigate the importance of submerged macrophytes as substrates for juvenile salmonid food production, and determine if invertebrates associated with macrophytes supplement John Day Reservoir food webs.

- Examine trophic linkages in shallow water habitats of reservoirs compared to those in free-flowing reaches.

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Attachment B: *Limnological Investigations in John Day Reservoir Including Selected Upper Reservoir Habitats, April, 1994 to September, 1995.*

LIMNOLOGICAL INVESTIGATIONS IN JOHN DAY RESERVOIR

INCLUDING SELECTED UPPER RESERVOIR HABITATS,

APRIL 1994 TO SEPTEMBER 1995

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EXECUTIVE SUMMARY

The National Marine Fisheries Service conducted limnological sampling of John Day Reservoir from April 1994 to September 1995. The study was initiated to acquire baseline data prior to reservoir drawdown, which was under consideration at that time. Five years of sampling were scheduled, centered on the drawdown year.

The study was designed in anticipation of drawdown to minimum operating pool. Under this scenario, emergent lands would total an estimated 8,400 acres, mostly in the upper reservoir. Accordingly, sampling effort was concentrated at upper-reservoir locations, although we also obtained samples from mid- and lower-reservoir and to a lesser extent at a protected backwater at Crow Butte. Sampling ended in October 1995, following the decision to suspend study of John Day Reservoir drawdown.

We conducted sampling for a wide range of physical, chemical, and biological attributes. Data ranges given in the following summary represent average values for samples collected at lower-, mid-, and upper-reservoir sampling stations. Unless stated otherwise, data do not include results from sampling at the Crow Butte backwater.

Physical variables and corresponding value ranges determined during the study were 1) temperature, 3.6 to 21.8 C; 2) Secchi disk visibility, 0.5 to 3.3 m; 3) turbidity, 2 to 23 NTU; 4) vertical light absorption coefficients, 0.77 to 2.85; and 5) sediment volatile solids (0.7 to 2.7%), sediment median grain size (0.1 to 0.2 mm), and sediment silt/clay (11.4 to 31.4%).

Chemical properties and ranges were 1) dissolved oxygen, 8.4 to 12.8 mg/L; 2) pH, 7.7 to 8.6; 3) specific conductance, 112 to 191 µS/cm; 4) alkalinity, 46 to 75 mg/L $CaCO_3$; 5) silica, 4.9 to 17.7 mg/L; 6) ammonia nitrogen, 0.01 to 0.06 mg/L; 7) nitrite + nitrate nitrogen, 0.01 to 0.55 mg/L; 8) total nitrogen, 0.20 to 0.88 mg/L; 9) orthophosphate phosphorus, less than 1 to 27 µg/L; 10) total phosphorus, 18 to 83 µg/L; 11) calcium, 9.4 to 23.0 mg/L; 12) magnesium, 3.4 to 7.0 mg/L; 13) sodium, 3.5 to 11.2 mg/L; 14) potassium, 0.8 to 2.3 mg/L; 15) sulfate, 8.4 to 18.1 mg/L; and 16) chloride, 1.4 to 4.0 mg/L.

Biological attributes included chlorophyll *a*, zooplankton, benthic macroinvertebrates, and fish. Chlorophyll *a* content of samples collected in John Day Reservoir ranged from 1.3 to 10.6 mg/m^3. Cladoceran and copepod densities ranged from less than 1 to 25 and from less than 1 to 42 organisms/L, respectively. Densities of benthic macroinvertebrates (by month of sample collection; including Crow Butte samples) varied from 3,100 to 10,700 organisms/m^2. Beach seining at upper reservoir locations resulted in the catch of 5,830 fishes, including 11 common resident taxa and 3 migratory species: American shad, chinook salmon, and coho salmon. Except for a few adult prickly sculpins, suckers, and common carp, fishes captured were juveniles.

The variation we observed in John Day Reservoir attributes was largely temporal. Spatial variation was negligible between samples collected from the lower-, mid-, and upper-reservoir. Vertical profiles and samples collected from different depths within the water column generally indicated uniformity from surface to bottom.

The homogenous nature of reservoir waters with regard to physical, chemical, and biological attributes is largely a result of low water-retention times within the reservoir. Along the

mainstem Columbia and Snake Rivers, thermal and chemical stratification are known to develop only in reservoirs with retention times exceeding those of John Day Reservoir. Biological characteristics of the reservoir are also affected by retention time. In reservoirs with low retention times, as in free-flowing rivers, phytoplankton populations are swept downstream before they can fully develop. Many zooplankton species cannot complete their life cycles during the relatively short time of passage through the reservoir

In contrast, samples collected at the Crow Butte backwater differed markedly from those collected at near-channel sampling stations. Although sampling at Crow Butte was limited to the final 7 months of the study, results were sufficient to indicate higher phytoplankton and zooplankton abundance than at near-channel stations. The contrast between habitat types was striking.

For the weeks during which we sampled at both habitat types, chlorophyll *a* at Crow Butte averaged 11.2 mg/m^3 compared with 6.8 mg/m^3 at nearby upper-reservoir stations. Cladoceran densities at Crow Butte and at upper-reservoir stations averaged 64 and less than 1 organisms/L, respectively. Copepod densities averaged 32 organisms/L at Crow Butte, but only 2 organisms/L at upper-reservoir stations. Results suggest relatively high productivity of backwater habitats in the present reservoir.

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INTRODUCTION

Drawdown of John Day Reservoir was proposed in 1991, principally as a means of reducing travel times of outmigrant juvenile salmonids passing through the reservoir. The drawdown under consideration in 1991 was from the normal reservoir elevation of 263.5 to 266.5 ft mean sea level (MSL) to minimum operating pool (MOP) level of 257 ft MSL. Because of the low gradient of the original streambed in the upper reservoir, drawdown to MOP would result in loss of an estimated 8,400 acres of shallow-water habitat, mostly in the Blalock Islands area of the upper reservoir. To evaluate changes in the reservoir environment during and after drawdown to MOP, the National Marine Fisheries Service (NMFS) began collection of baseline physical, chemical, and biological data in April 1994. Data collection continued until September 1995, when funding for the study of reservoir drawdown was withdrawn. Although drawdown to MOP did not occur, NMFS limnological sampling produced an extensive set of baseline data which is detailed and summarized in this report.

In response to direction provided in the Energy and Water Development Appropriation Bill, 1998, the Corps of Engineers (COE) is again studying drawdown of John Day Reservoir. Options under consideration include operation at spillway crest level (about 220 ft MSL) and restoration to natural river elevation (about 170 ft MSL). During Phase I of the present COE study, existing information, including the NMFS limnological data set, will be used to evaluate potential benefits and costs from the proposed drawdown scenarios.

Study Area

John Day Dam is located at river kilometer (RKm) 347 on the Columbia River (Fig. 1). The pool, Lake Umatilla, filled in 1968, has a surface area of about 20,000 hectares (49,000 acres) and extends 123 km upstream to McNary Dam. The maximum pool elevation is 81.7 m (268 ft) above MSL, and MOP is 78.4 m (257 ft) above MSL. Normal operating levels are usually in the 80.8-81.7 m range for the July to October period and in the 79.3-80.8 m range for the November to June period. Operation below 80.8 m (265 ft) is generally for flood control purposes. Median Columbia River flows are about 7,930 m^3/second in May and June, 5,660 m^3/second in July, and 3,680 m^3/second in August. Water particle travel time through the reservoir has been estimated to range from 3.8 days at flows of 8,495 m^3/second to 11.2 days at river flow of 2,830 m^3/second (COE 1992).

COE data indicate a relatively flat reservoir from John Day Dam upstream to RKm 451. From RKm 451 to McNary Dam, a distance of 19 km, the pool rises more sharply, creating conditions somewhat resembling those of a free-flowing river. In a previous study (Li et al. 1981) velocity data were used to characterize the reservoir into forebay (John Day Dam to RKm 398), transitional (RKm 398 to 454), and tailrace zones (RKm 454 to McNary Dam).

Sampling Stations

A drawdown of John Day reservoir from elevation 80.8 m to elevation 78.4 m (MOP) would result in the loss of approximately 8,400 acres of upper reservoir habitat (COE 1992). Much of this habitat loss would occur in the Blalock Islands area of the upper reservoir, where topography has resulted in broad littoral zones. Accordingly, with the drawdown-to-MOP scenario under consideration at the time of our study, sampling effort was concentrated at five upper-reservoir

stations within the Blalock Islands area. These stations were characterized by water depths of 5 to 7 m and were located along the main river channel. Less intensive sampling was conducted at lower- and mid-reservoir stations where water depths ranged from 30 to 40 m. In 1995, the second year of the study, we also collected data from a backwater location at Crow Butte State Park. Sampling areas and stations are shown in Figure 1.

METHODS

At each sampling station, field measurements were made of physical variables including temperature, turbidity, and Secchi disk visibility. Water samples were taken and preserved for laboratory determination of chemical variables including alkalinity, nutrients, and major ions. Other chemical variables such as pH, dissolved oxygen, and specific conductance were monitored directly using a multi-function meter. Biological variables included chlorophyll *a*, zooplankton, benthic invertebrates, and resident and migrant fish, which were collected using a 30-m beach seine.

Schedule

The sampling schedule followed during the study is provided in Table 1. Upper-reservoir stations were generally sampled biweekly from April through August and monthly from September through March for water quality variables, chlorophyll *a*, and zooplankton. Fish were sampled biweekly from May through August and monthly from September through March. Benthic invertebrates were sampled monthly, and sediment samples were collected quarterly. An exception to the above schedule was December 1994, when data collection was limited to the water quality variables. Sampling at the Crow Butte backwater began in March 1995 and generally followed the schedule for upper-reservoir stations. Lower- and mid-reservoir stations were sampled at monthly intervals, and no sampling of benthic invertebrates, sediment, or fish occurred at these stations.

Physical Variables

Physical attributes measured or recorded at all sites included turbidity, ambient and underwater irradiance, Secchi disk visibility, and surface-to-bottom profiles of water temperature. Physical attributes were logged at the 5-m depth contour at upper-reservoir stations and in depths of 30 to 40 m at lower- and mid-reservoir stations. Sediment samples were taken at the 1-, 3-, and 5-m depth contours of upper-reservoir stations for analyses of organic content, particle size, and soil classification.

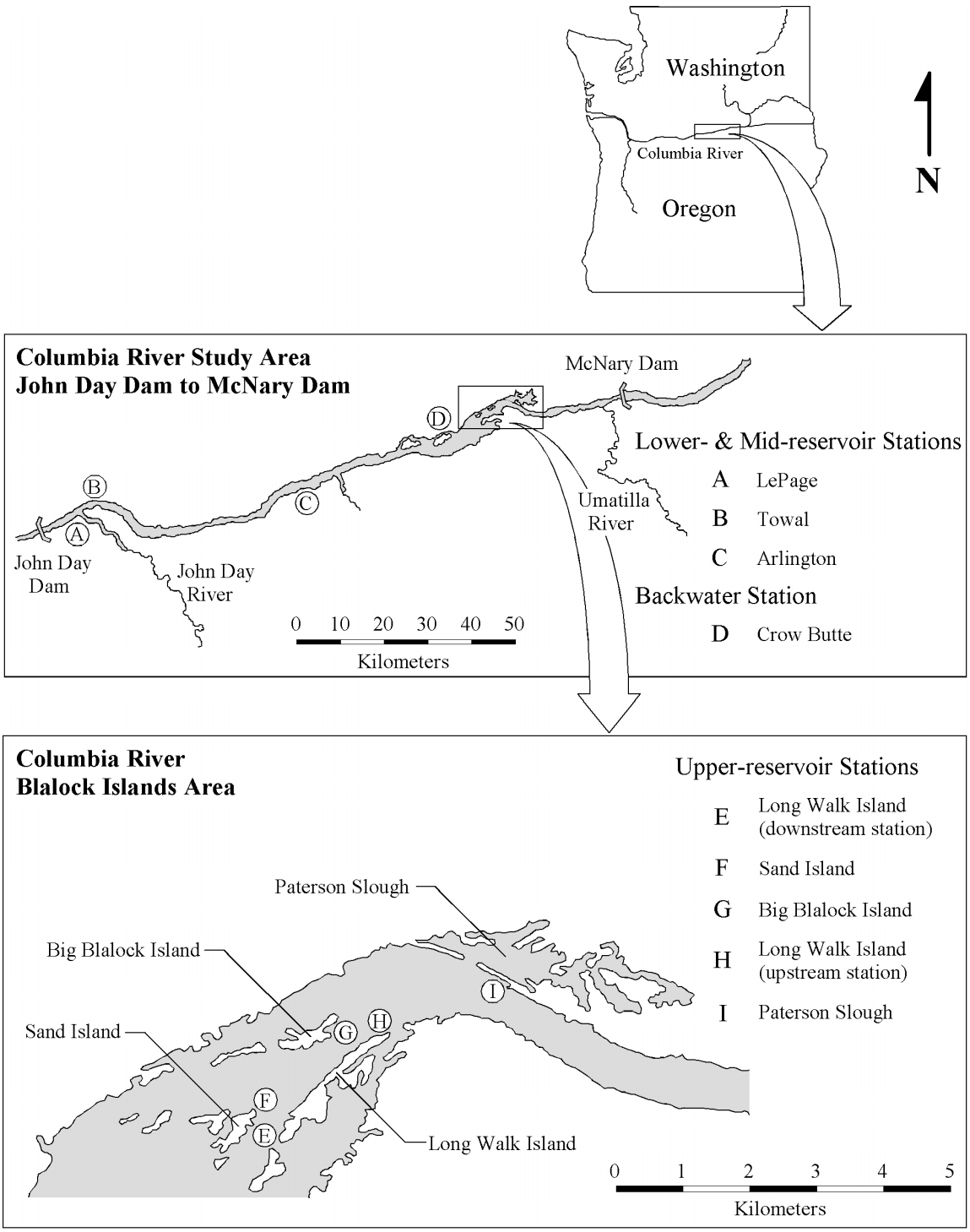

Figure 1. Sampling areas and stations on the Columbia River. John Day Reservoir limnology study, 1994-95.

Table 1. Sampling schedule and reservoir properties for which data were collected during limnological sampling of John Day Reservoir, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/11/94	C[b]W[c]Z[d]	CWZ	CWZ						
4/12/94					B[e]			B	B
4/13/94					CWZ	BCWZ	BCWZ	CWZ	WZ
4/26/94					CWZ	CWZ	CWZ	CWZ	CWZ
5/10/94	CWZ	CWZ	CWZ						
5/11/94					CWZ	CWZ	CWZ	CWZ	CWZ
5/12/94					BS[f]	BS	BS	BS	BS
5/13/94					F[g]	F	F	F	F
5/24/94					CFWZ	CFWZ	CFWZ	CFWZ	CFWZ
6/7/94	CWZ	CWZ							
6/8/94					BCFWZ	BCFWZ	CFWZ	CFWZ	CFWZ
6/9/94			CWZ				B	B	B
6/21/94					CFWZ	CWZ	CWZ	CWZ	CFWZ
7/6/94	CWZ	CWZ	CWZ						
7/7/94					BCFWZ	BCWZ	BCWZ	BCWZ	BCWZ
7/20/94					CFWZ	CWZ	CWZ	CWZ	CFWZ
8/2/94	CWZ	CWZ	CWZ						
8/3/94					BFS	BS	BS	BS	BFS
8/4/94					CWZ	CWZ	CWZ	CWZ	CWZ
8/18/94					CFWZ	CWZ	CWZ	CWZ	CFWZ
9/13/94					BF	B	B	B	BF
9/14/94					CWZ	CWZ	CWZ	CWZ	CWZ
9/15/94	CWZ	CWZ	CWZ						
10/4/94	CWZ	CWZ	CWZ						
10/5/94					BF	B	B	B	BF
10/6/94					CWZ	CWZ	CWZ	CWZ	CWZ
11/15/94	CWZ	CWZ	CWZ						
11/17/94					BCFSWZ	BCSWZ	CWZ	CWZ	CFWZ
11/18/94							BS	BS	BS
12/13/94					W	W	W	W	W
12/14/94	W	W	W						
1/10/95	CWZ	CWZ	CWZ						
1/24/95					BFS	BS	BS	BS	BFS
1/25/95					CWZ	CWZ	CWZ	CWZ	CWZS
2/21/95	W	W	W						
2/22/95					W	W	W	W	W
3/13/95	CWZ	CWZ	CWZ						
3/14/95					B	B	B	B	B
3/15/95				WZ	CWZ	CWZ	CWZ	CWZ	CWZ
4/4/95				CWZ	CWZ	CWZ	CWZ	CWZ	CWZ
4/18/95				BCSWZ					
4/19/95					BFS	B	B	B	BFS
4/20/95					CWZ	CWZ	CWZ	CWZ	CWZ

Table 1. Continued.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/21/95	CWZ	CWZ	CWZ						
5/1/95				CWZ					
5/2/95									CFWZ
5/3/95					CWZ	CWZ	CWZ	CWZ	
5/15/95	CWZ	CWZ	CWZ						
5/16/95					B	B	B	B	B
5/17/95				CWZ	CWZ	CWZ	CWZ	CWZ	CWZ
5/30/95				BCWZ					
5/31/95					CFWZ	CWZ	CWZ	CWZ	CFWZ
6/12/95	CWZ	CWZ	CWZ						
6/13/95					B	B	B	B	BF
6/14/95					CWZ	CWZ	CWZ	CWZ	CFWZ
6/15/95				BCWZ					
6/28/95				CWZ					
6/29/95					CWZ	CWZ	CWZ	CWZ	CFWZ
7/10/95	CWZ	CWZ	CWZ						
7/11/95					BFS	BS	BS	BS	BFS
7/12/95					CWZ	CWZ	CWZ	CWZ	CWZ
7/13/95				BCSWZ					
7/24/95				CWZ					
7/25/95					CFWZ	CWZ	CWZ	CWZ	CFWZ
8/8/95				BCWZ	CWZ	CWZ	CWZ	CWZ	CWZ
8/9/95					BF	B	B	B	BF
8/10/95	CWZ	CWZ	CWZ						
8/21/95				CWZ					
8/22/95					CFWZ	CWZ	CWZ	CWZ	CFWZ
9/5/95				BCWZ					
9/6/95					BF	B	B	B	BF
9/7/95					CWZ	CWZ	CWZ	CWZ	CWZ
9/8/95	CWZ	CWZ	CWZ						
9/19/95				CWZ	CFWZ	CWZ	CWZ	CWZ	CFWZ

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.
[b] C = Depth-integrated samples for analysis of monochromatic chlorophyll *a*.
[c] W = Samples and field measurements for analysis of water quality.
[d] Z = Zooplankton samples.
[e] B = Benthic invertebrate samples.
[f] S = Sediment samples.
[g] F = Beach seine sampling of resident and anadromous fish.

Turbidity was measured in Nephelometric Turbidity Units (NTU) using either an HF Instruments Model DRT-15 or Hach Model 2100P portable turbidimeter.[1] Irradiance readings were recorded from a Kahlsico Model 268WD305 digital underwater irradiameter to calculate light extinction coefficients and to delineate the euphotic zone. Secchi disk visibilities were obtained with a standard 20-cm-diameter Secchi disk. Surface-to-bottom profiles of water temperature were obtained using a Hydrolab Corporation H20 transmitter and Surveyor 3 logging system. At upper-reservoir stations, we recorded data in 1-m increments from surface to bottom (5-7 m). At lower- and mid-reservoir stations we recorded data in 1-m intervals from the surface to 10-m depth, then at 5-m intervals to the bottom (30 to 40-m depth).

Chemical Variables

Surface-to-bottom water profiles of dissolved oxygen, pH, and specific conductance were obtained using a Hydrolab Corporation H20 transmitter and Surveyor 3 logging system. Water samples were analyzed for nitrogen (NH_3-N, NO_2+NO_3-N, and total), phosphorus (ortho and total), anions (sulfate and chloride), cations (calcium, magnesium, sodium, and potassium), alkalinity, turbidity, and silica. Analyses were conducted by the Water Research Center (WRC) at Washington State University, Pullman, Washington. Alkalinity was also determined at the time of sample collection with a LaMotte alkalinity kit. Water samples for laboratory analyses and for on-board determination of alkalinity were obtained using a 4-L van Dorn bottle lowered to the desired depth. At upper-reservoir stations and at Crow Butte, we collected samples from the surface and just above the bottom. At lower- and mid-reservoir stations we collected samples from just below the surface, at mid-depth, and just above the bottom.

Biological Variables

At all stations, we collected water samples to determine chlorophyll *a* concentrations and sampled zooplankton to determine species composition and abundance. At upper-reservoir stations, benthic macroinvertebrates were sampled, and we also conducted limited beach seining to determine use of these habitats by resident and migrant fish.

Replicate composite samples for chlorophyll *a* analysis were collected from the surface to a depth of 5 m at upper-reservoir stations, from the Crow Butte backwater, and from the surface to a depth of 15 m at lower- and mid-reservoir stations. Composite samples (integrated over depth) were obtained by lowering a weighted 5-cm-diameter hose (both ends open) vertically in the water column. With the lower, weighted end of the hose at the appropriate depth, the upper end was sealed with a rubber stopper. The lower end of the hose was then retrieved by means of an attached rope. The hose was uncapped and the composite water sample collected in a 19-L bucket as the upper end of the hose was retrieved into the boat. A 1-L water sample was taken from the stirred contents of the bucket. The remaining composite sample was filtered through a 25-µm net to obtain the fraction of the composite sample containing phytoplankton less than 25 µm in size. Phytoplankters of this size often constitute a large portion of the overall phytoplankton, and may contain over 90% of the chlorophyll *a* in a unit volume of water. A 1-L sample of the filtrate was obtained from the bucket. Samples were stored on ice in dark

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1 Reference to trade names does not imply endorsement by the National Marine Fisheries Service, NOAA.

polyethylene bottles and transported to the NMFS Point Adams Biological Field Station for filtration and spectrophotometric analysis.

Zooplankton samples were obtained from vertical tows using a Wisconsin-style plankton net with a nonporous truncated upper cone (11.4-cm mouth diameter, 17.8-cm second ring diameter, 87.6 cm long, 80-µm mesh). Three replicate vertical tows were made from 5-m depths to the surface at the 5-m contour of upper-reservoir stations and at the Crow Butte backwater. At lower- and mid-reservoir stations, three replicate tows were made from the 15-m depth to the surface. The plankton net was retrieved at about 0.5 m/second with the vessel either anchored or drifting with the current, as necessary to maintain a vertical tow. Zooplankton were preserved with Lugol's iodine, added at about 5 mL of solution to 100 mL of sample. Samples were stored and shipped in 250-mL polyethylene bottles. Analysis of samples was performed by Beaver Schaberg Associates, Inc. In the contractor's laboratory, at least three 1-mL subsamples were removed from each sample concentrate and placed in Utermohl chambers. A minimum of 200 organisms were identified to species and enumerated using an inverted microscope at 100x magnification. If the species tally was less than 200 organisms, additional subsamples were examined.

Benthic invertebrates were sampled only at upper-reservoir stations and at the Crow Butte backwater. We considered the two to four benthic invertebrate samples collected from each date/station/depth strata as replicates. Replicate sample data were used to calculate mean density (number/m^2) and standard deviation for total benthic invertebrates collected from each strata. Data combined by depth and site were used to summarize monthly densities of dominant benthic invertebrate taxa during the sampling period.

We sampled bottom sediments for benthic invertebrates using a standard 0.052 m^2 Ponar grab. Collected sediments were washed through a U.S. Standard No. 35 sieve bucket. Samples were preserved in 10% buffered formalin solution. Replicate samples were taken at the 1-, 3-, and 5-m contours at each upper-reservoir station.

We sampled resident and migratory fish from the Long Walk Island downstream site and from Patterson Slough using a 30- by 2.4-m beach seine constructed of 6.4-mm (stretch measure) knotless nylon mesh. At the midpoint of the net, a 2.4- by 2.4- by 2.4-m bunt was attached to retain the catch. The net was set parallel to the beach in about 1.2 m of water and the ends retrieved to the shoreline. Fish captured in the net were anesthetized with tricaine methanesulfonate (MS-222) and counted by species. A subsample of each species was measured to the nearest mm (fork length for salmonids, total length for resident species). Following recovery from the anesthetic, all fish were released into the area from which they were captured.

RESULTS

Physical Variables

Temperature

Comparison of water temperatures (overall means for measurements at lower-, mid-, and upper-reservoir sampling stations) during the April to September periods in 1994 and 1995 showed little difference between the two years (Fig. 2). Minimum and maximum temperatures

observed at individual sampling stations (lower-, mid-, and upper-reservoir) during the April 1994 to September 1995 study period were 3.4□C in January and February 1995 and 21.8□C in August 1994, respectively (Appendix Table A-1). Temperatures at the Crow Butte backwater were typically 1 to 2□C higher than at upper-, mid-, and lower-reservoir stations visited during the same weeks from March through July 1995 (Appendix Fig. B-1).

Average surface-to-bottom temperatures were slightly lower (by 0.1 to 0.3□C) at upper-reservoir stations than at lower- and mid-reservoir stations sampled during the same weeks in 1994 (Appendix Table A-1). Average surface-to-bottom temperature at lower- and mid-reservoir stations in 1994 varied by as much as 1.4□C between stations sampled on the same date. Similar variations (up to 1.0□C) were observed between upper reservoir sampling stations.

Vertical temperature profiles at lower- and mid-reservoir stations in 1994 show gradually declining temperatures from the surface to depths of 10 to 15 m and then relatively stable temperatures from 15 m to the bottom (Appendix Fig. B-2). In contrast, at upper-reservoir stations during 1994, temperature decreases with depth were noted in only the top meter of water (Appendix Fig. B-3). Maximum temperature changes with depth in 1994 were observed in May at lower- and mid-reservoir stations (by 3.4□C) and from May to August at upper-reservoir stations (by 1.6□C).

Water Clarity

Water clarity, as indicated by Secchi disk visibility, generally increased from April to September in both 1994 and 1995 (Fig. 3). Data from individual sampling stations (lower-, mid-, and upper-reservoir) show highest visibility (4.4 m) in December 1994 (Appendix Table A-2). Lowest visibility (0.2 m) coincided with a runoff event during February 1995. Secchi disk visibility was usually greater at lower-, mid-, and upper-reservoir stations than at the Crow Butte backwater (Appendix Fig. B-4).

Comparisons of turbidity measurements between years show that river conditions were slightly less turbid in 1994 than in 1995 (Fig. 2). Although turbidity measured at lower-, mid-, and upper-reservoir stations during the study period varied from 1 to 42 NTU, readings on most sampling dates were within the range of 2 to 8 NTU (Appendix Table A-3). Turbidity readings at lower- and mid-reservoir stations generally increased with depth, whereas surface-to-bottom differences were minimal at upper-reservoir stations (Appendix Table A-3). Turbidity readings varied between reservoir locations sampled during the same weeks, with upper-reservoir stations usually showing slightly higher turbidities than lower reservoir stations (Appendix Fig. B-1).

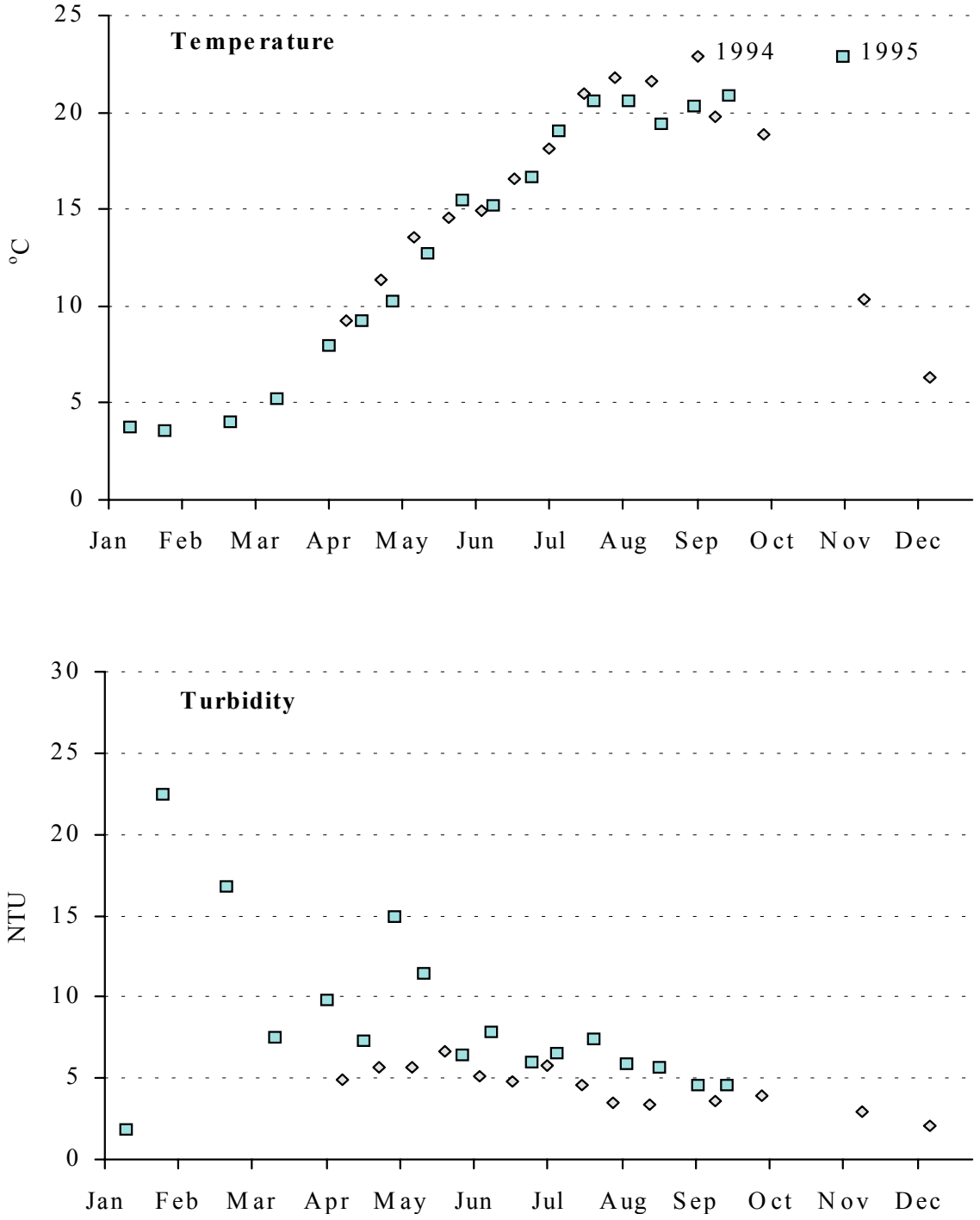


Figure 2. John Day Reservoir water temperature and turbidity in 1994 and 1995. Data points represent overall means for samples collected at lower-, mid-, and upper-reservoir sampling stations.

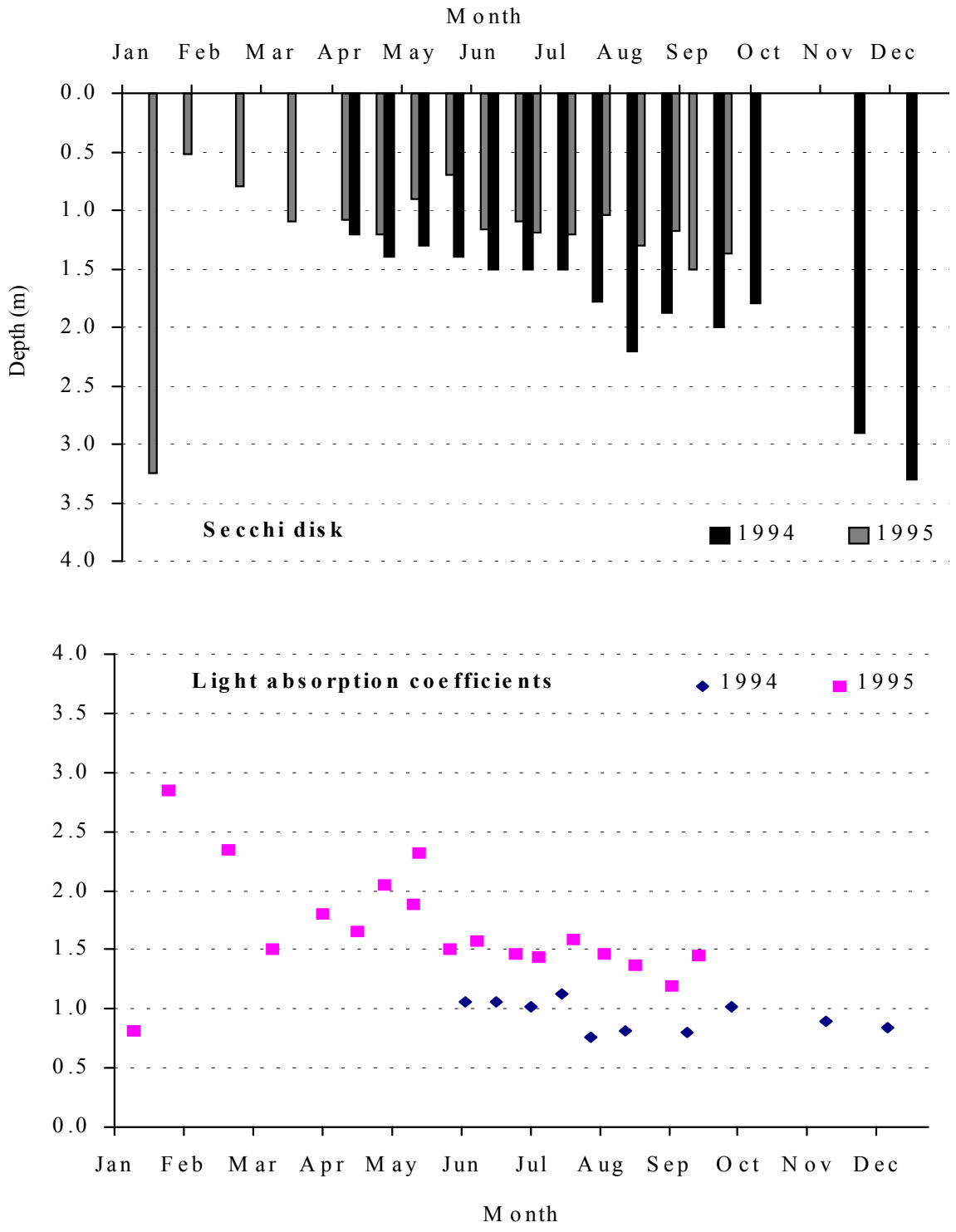


Figure 3. Secchi disk visibility and vertical light absorption coefficients at John Day Reservoir sampling stations in 1994 and 1995.

Overall mean light absorption coefficients varied from 0.77 to 1.06 in 1994 and from 0.82 to 2.85 in 1995 (Fig. 3). Vertical light absorption coefficients calculated for individual sampling stations are given in Appendix Table A-4, and the corresponding light transmittance values are shown in Appendix Table A-5. Differences in light absorption coefficients between reservoir locations were minimal in 1994 but more pronounced in 1995 (Appendix Fig. B-4). For data collected in 1994, plots of light transmittance vs. depth indicated that euphotic zone depth ranged from about 3.5 to 8 m.

Sediment Analysis

The majority of sediment samples obtained at upper reservoir sites were classified as silty sand. Median grain size of samples varied from 0.1 to 0.2 mm, volatile solids from 0.7 to 2.7%, and silt/clay from 11.4 to 31.4% (Fig. 4). Consistent differences in sediment parameters related to depth of sample collection (1 m, 3 m, or 5 m) were not evident. Sediment characteristics at individual sampling stations are detailed in Appendix Table A-6. Data tend to show higher organic content and smaller grain size at the Oregon shore stations than at the Washington shore sampling stations.

Chemical Variables

Dissolved Oxygen

Dissolved oxygen levels varied from 8.4 to 12.8 mg/L during the 1994-95 sampling period (Fig. 5). On most dates, average surface-to-bottom oxygen levels at upper-reservoir stations were slightly higher than at lower- and mid-reservoir stations (Appendix Fig. B-5). Station-to-station differences were typically minimal between stations in the same reservoir location sampled on the same date (Appendix Table A-7). Changes in oxygen content with depth are illustrated by the surface-to-bottom oxygen profiles shown for 1994 data in Appendix Figures B-6 and B-7. In 1994, dissolved oxygen decreased from surface to bottom from April to October, and increased from surface to bottom in November and December. At lower- and mid-reservoir stations, changes in dissolved oxygen were more rapid in the upper 10 m of the water column than in the zone from 10 m to the bottom. Variation in dissolved oxygen with depth at the upper-reservoir stations was comparable to that seen in the upper 5 m at lower- and mid-reservoir stations.

pH

Overall surface-to-bottom pH values (antilog means) at John Day Reservoir sampling stations (exclusive of the Crow Butte backwater) during the 1994-95 sampling period varied from 7.7 to 8.6 pH units (Fig. 6). At lower-, mid-, and upper-reservoir stations, pH values were generally similar, however at the Crow Butte backwater station, pH values were consistently higher (Appendix Fig. B-8 and Appendix Table A-8). In 1994, variation of pH within the water column was about 0.1 pH units at lower- and mid-reservoir sampling stations on most dates and slightly less than 0.1 pH units at upper-reservoir stations (Appendix Figs. B-9 and B-10).

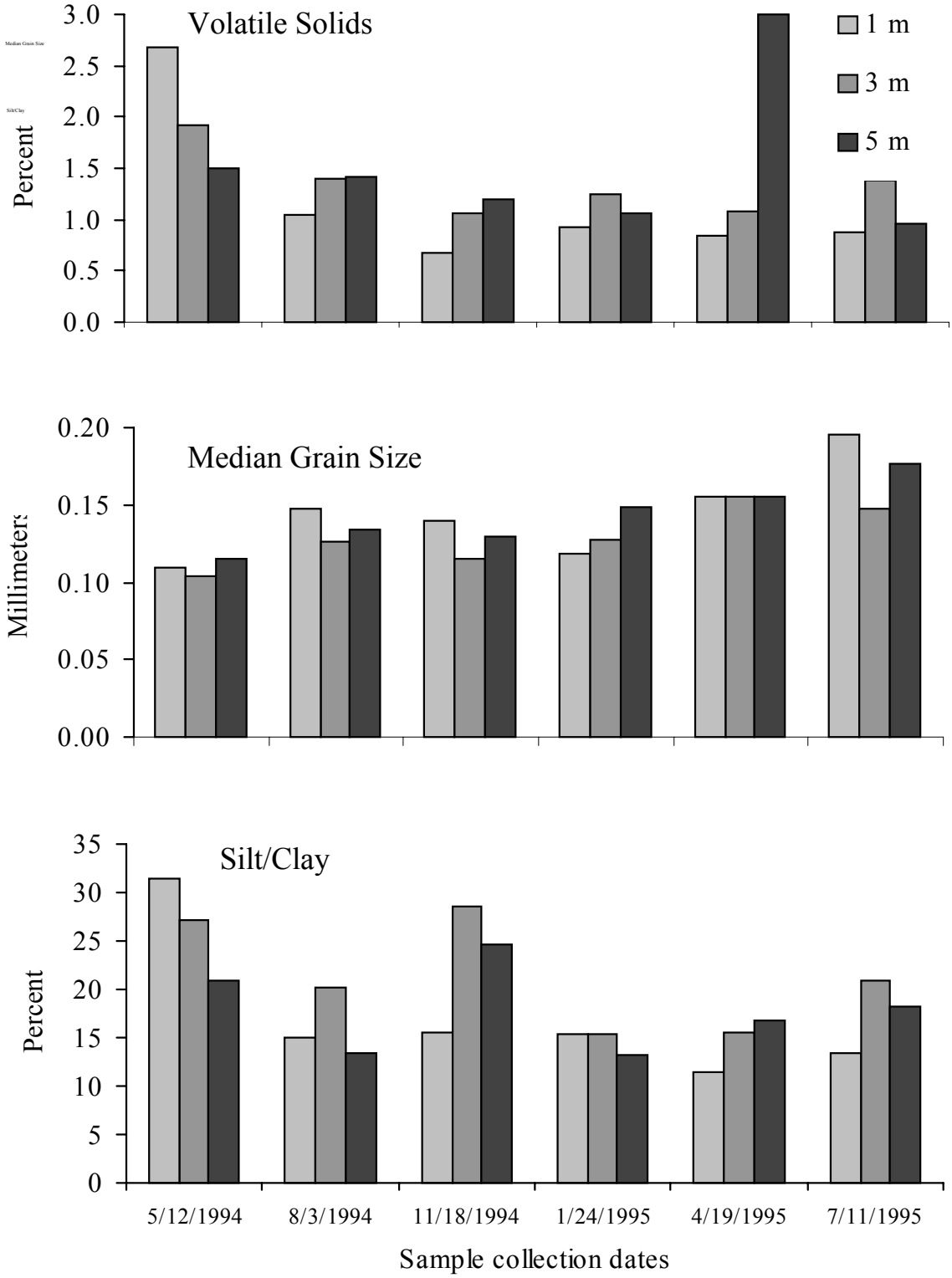
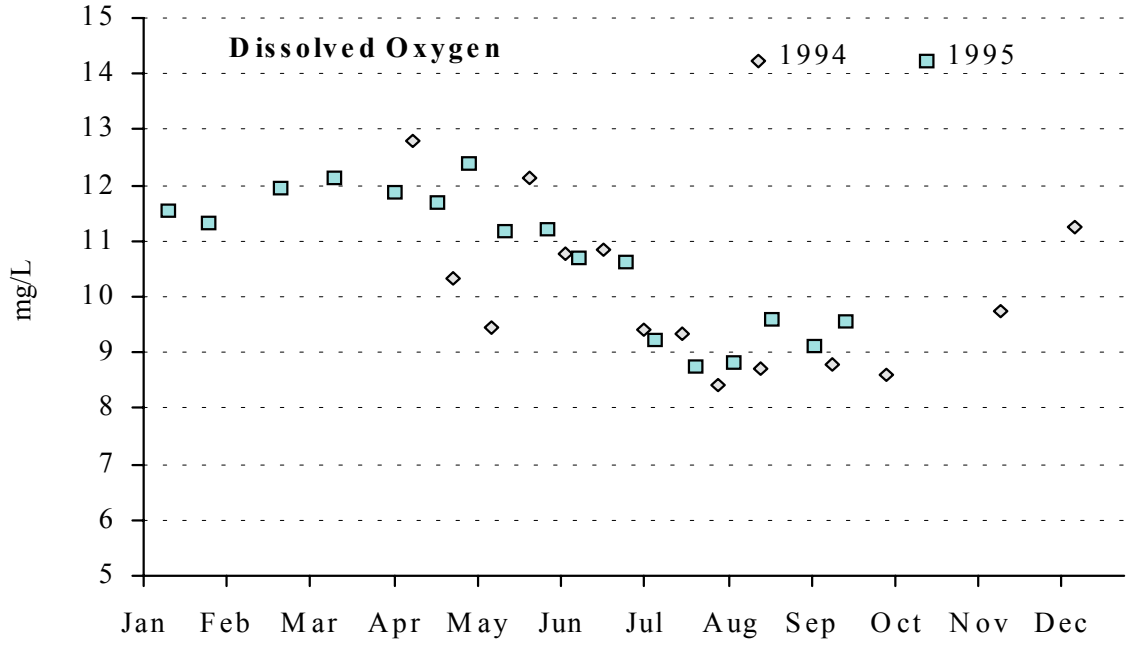


Figure 4. Volatile solids, median grain size, and percent silt/clay in sediment samples collected at upper John Day Reservoir sampling stations (1-, 3-, and 5-m depths), April 1994 to September 1995.



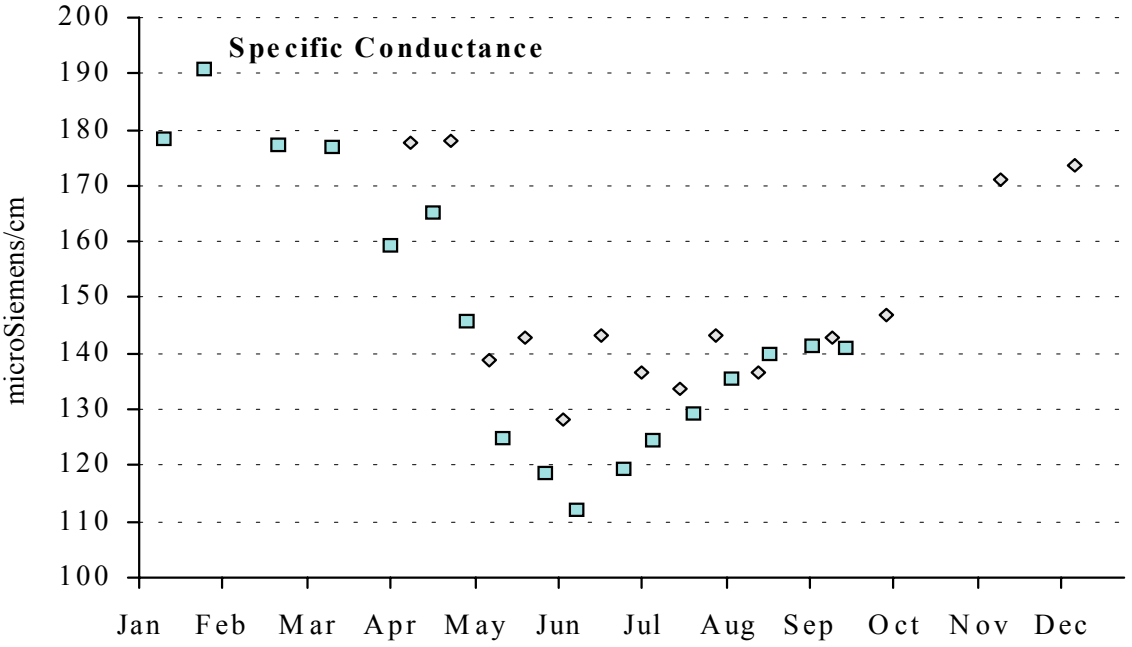


Figure 5. Comparison of 1994 and 1995 John Day Reservoir overall mean dissolved oxygen levels and specific conductance values.

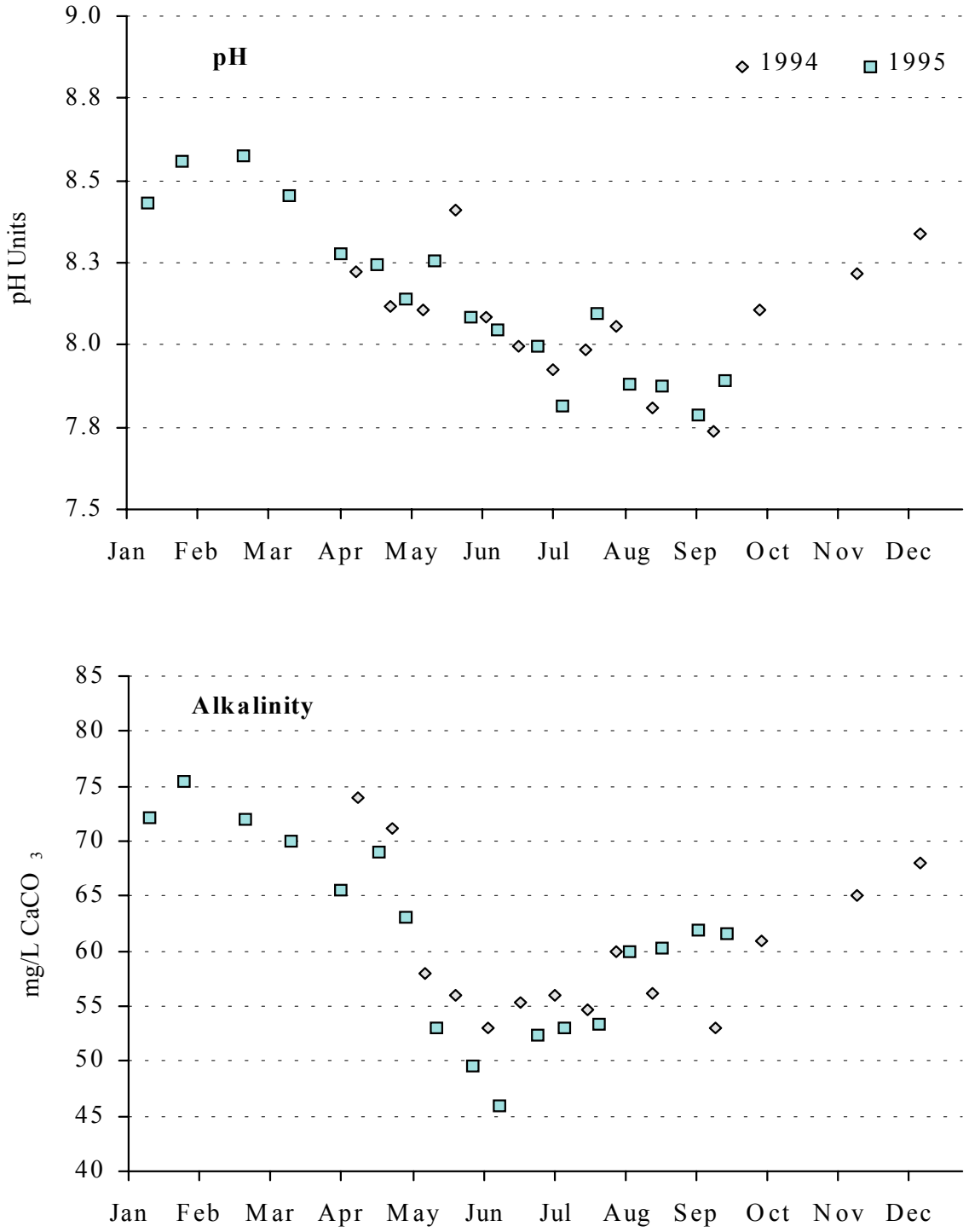


Figure 6. Comparison of 1994 and 1995 John Day Reservoir overall mean pH and alkalinity values.

Specific Conductance

Overall mean specific conductance measured at John Day Reservoir stations ranged from 112 to 191 µS/cm during the 1994-95 sampling period (Fig. 5). Specific conductance values differed only slightly between lower-, mid-, and upper-reservoir stations sampled during the same week, but higher specific conductance values were generally observed at the Crow Butte backwater than at other reservoir locations (Appendix Fig. B-5 and Appendix Table A-9). Specific conductance profiles (Appendix Figs. B-11 and B-12) show minimal variation in specific conductance with depth.

Alkalinity

Total alkalinity (averages of surface, mid-depth, and bottom samples) varied within the range of 46 to 75 mg/L $CaCO_3$ from April 1994 to September 1995 (Fig. 6). Total alkalinity values were similar at lower-, mid-, and upper-reservoir stations (Appendix Fig. B-8) and varied little with sample depth (Appendix Table A-10).

Silica

Silica concentrations varied from 4.9 to 17.7 mg/L during the 1994-95 sampling period (Fig. 7). Silica content was similar at lower-, mid-, and upper-reservoir locations, but present at lower levels in samples collected during the same weeks at the Crow Butte backwater (Appendix Fig. B-13). Silica content was similar in samples collected at different depths (Appendix Table A-11).

Major Ions

From April 1994 to September 1995, concentrations of major ions at John Day Reservoir stations varied within the following ranges: sulfate, 8.4 to 18.1 mg/L; chloride, 1.4 to 4.0 mg/L; calcium, 9.4 to 23.0 mg/L; magnesium, 3.4 to 7.0 mg/L; sodium, 3.5 to 11.2 mg/L; and potassium, 0.8 to 2.3 mg/L. Since specific conductance is a measure of ionic concentration, temporal changes in concentrations of major ions closely follow the curve given for specific conductance in Figure 5. Minimal variation was observed in concentrations of major ions between sampling stations or between depths from which samples were obtained (Appendix Tables A-12 through A-17).

Nitrogen and Phosphorus

Concentrations of ammonia nitrogen (NH_3-N), nitrite + nitrate nitrogen (NO_2+NO_3-N), and total nitrogen (includes organic nitrogen, NH_3-N, and NO_2+NO_3-N) at John Day Reservoir sampling stations are shown in Appendix Tables A-18 through A-20. Between April 1994 and September 1995, concentrations of these nitrogen forms (averages for lower-, mid-, and upper-reservoir stations) varied within the following ranges: NH_3-N, 0.01 to 0.06 mg/L; NO_2+NO_3-N, 0.01 to 0.55 mg/L; and total nitrogen, 0.20 to 0.88 mg/L. Consistent differences associated with reservoir location or depth of sample collection were not apparent.

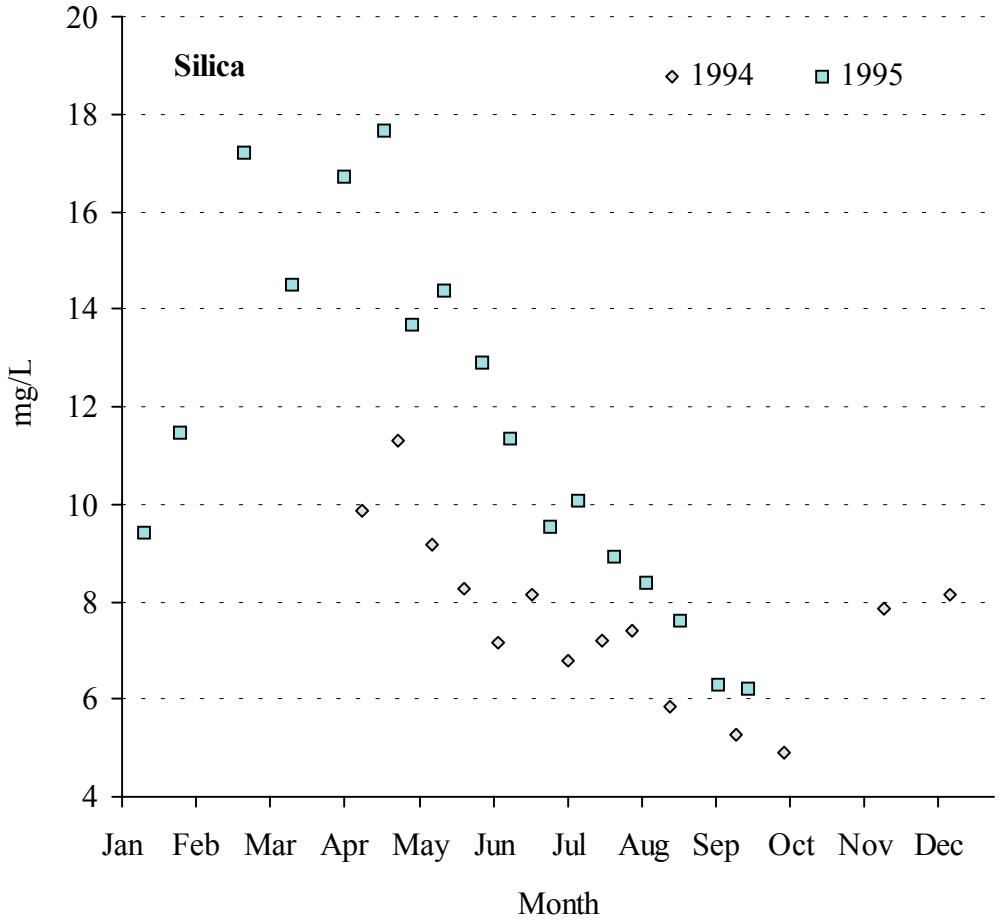


Figure 7. Silica content of John Day Reservoir in 1994 and 1995. Data points represent overall means for samples collected at lower-, mid-, and upper-reservoir sampling stations.

Orthophosphate phosphorus and total phosphorus concentrations (averages for lower-, mid-, and upper-reservoir stations) during the 1994-95 sampling period ranged from less than 1 to 27 µg/L and from 18 to 83 µg/L, respectively (Appendix Tables A-21 and A-22). Orthophosphate phosphorus content of samples collected from different depths shows occasional wide variation, but no consistent increasing or decreasing trends with depth. In contrast, total phosphorus concentrations frequently increased with depth of sample collection.

Biological Variables

Chlorophyll a

Chlorophyll *a* content of samples (averages for lower-, mid-, and upper-reservoir stations) during the 1994-95 sampling period ranged from 1.3 to 10.6 mg/m^3. Chlorophyll *a* levels peaked in April and May, declined during June, July, and August, and then increased during the fall months (Fig. 8). Chlorophyll *a* content of samples was highly variable between sampling stations visited on the same date, and between lower-, mid-, and upper-reservoir locations sampled during the same week (Appendix Fig. B-14). A greater level of primary production is suggested, however, at the Crow Butte backwater than at other reservoir locations. Appendix Tables A-23 and A-24 quantify the amounts chlorophyll a contributed by phytoplankton larger than 25 µm and smaller than 25 µm at each station throughout the study period.

Zooplankton

Average monthly densities of cladocerans, copepods, and rotifers at John Day Reservoir locations (exclusive of the Crow Butte backwater) during the 1994-95 sampling period are shown in Figure 9. Cladoceran densities ranged from 0.08 to 25 organisms/L, copepod densities from 0.4 to 42 organisms/L, and rotifer densities from 3 to 354 organisms/L. Appendix Tables A-25, A-26, and A-27 present monthly density data for zooplankters identified at lower- and mid-reservoir stations, upper-reservoir stations, and the Crow Butte backwater, respectively. Appendix Figures B-15 and B-16 compare densities of cladocerans and copepods between sampling locations in the open reservoir and the protected Crow Butte backwater. Appendix Table A-28 shows mean densities and standard deviations for individual sample locations and dates.

Cladocerans

Of ten cladoceran species identified in zooplankton samples collected at lower- and mid-reservoir stations, only *Bosmina longirostris* and *Daphnia thorata* were present at densities exceeding 1 organism/L in any month. Seventeen cladoceran species were identified from samples collected at upper-reservoir stations, including all 10 species identified at lower- and mid-reservoir stations. At upper-reservoir stations, as at lower- and mid-reservoir stations, *Bosmina longirostris* and *Daphnia thorata* were dominant. Other species typically occurred infrequently, at relatively low densities. In 1995, March-to-September cladoceran densities in the Crow Butte backwater peaked at 151 organisms/L, compared with maximum densities of 45 organisms/L at lower- and mid-reservoir stations, and 19 organisms/L at upper-reservoir stations during the same time period (Appendix Fig. B-15).

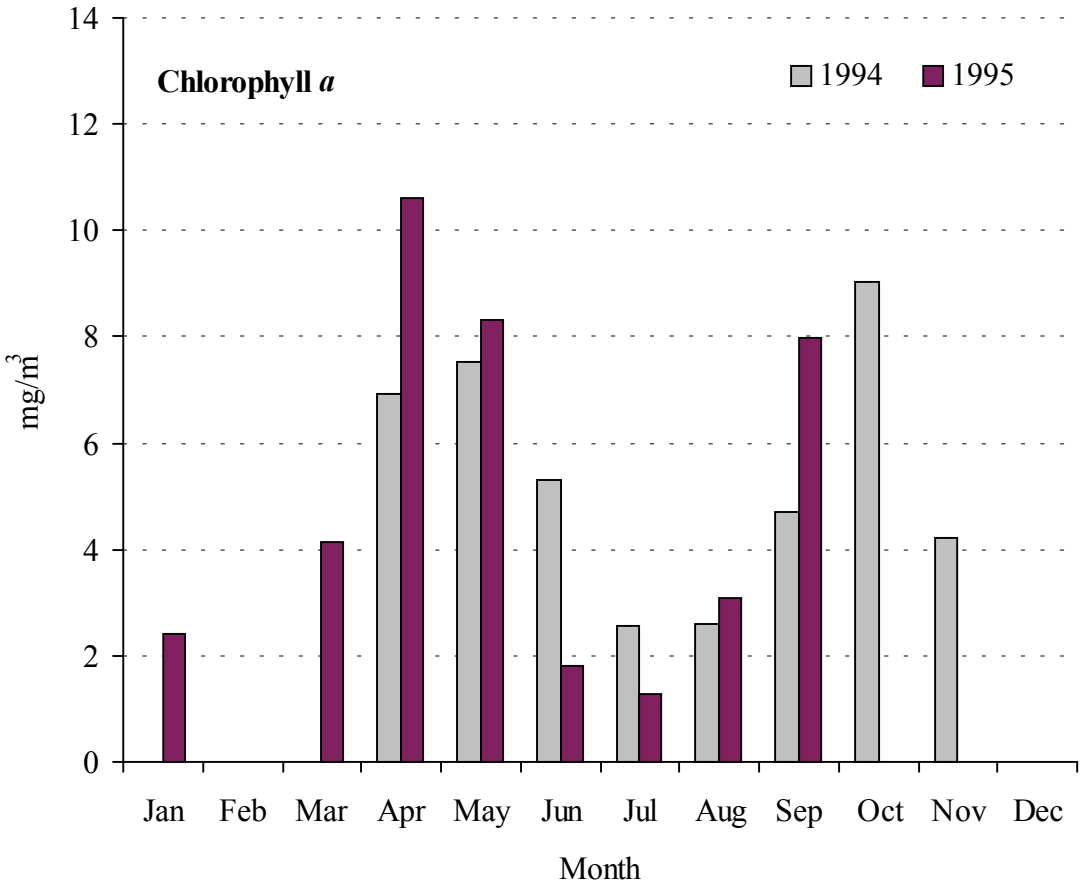


Figure 8. Chlorophyll a content of depth-integrated samples collected from John Day Reservoir during 1994 and 1995. Values shown are averages for samples collected from lower-, mid-, and upper-reservoir locations.

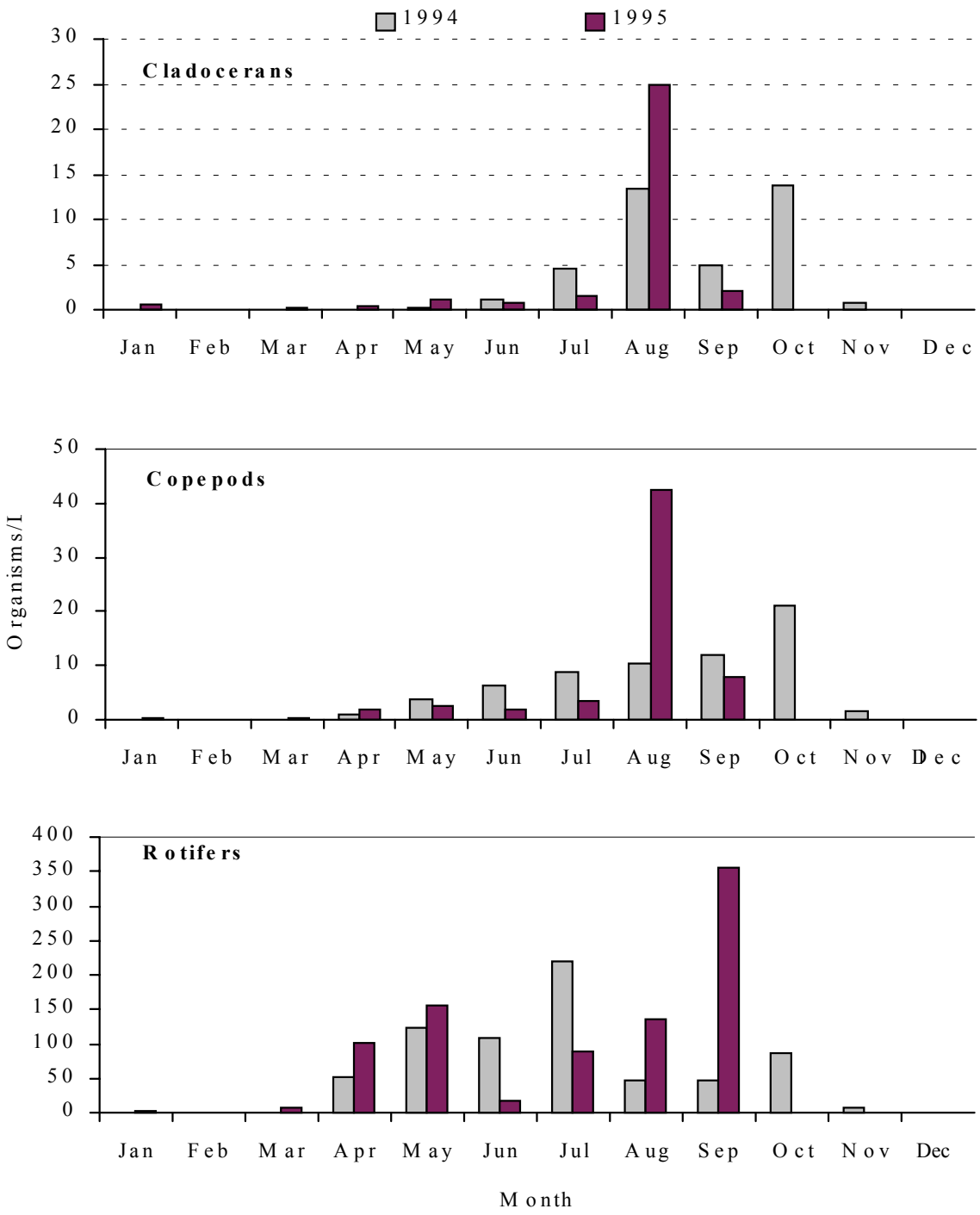


Figure 9. Zooplankton densities in John Day Reservoir during 1994 and 1995 (data from the Crow Butte backwater not included). No sample collection during February or December of either year.

Copepods

Collections contained predominantly immature life stages, including nauplii and copepodids. Calanoid copepodids were presumably developmental stages of *Leptodiaptomus ashlandi*, whereas cyclopoid copepodids were probably early life stages of *Diacyclops thomasi* or *Eucyclops agilis*, since these were the only mature copepod species identified in sample collections. *D. thomasi* was numerically dominant on most sample dates. Comparison of copepod densities between lower- and mid-reservoir, upper reservoir, and Crow Butte backwater locations from March to September 1995 shows slightly higher densities at lower- and mid-reservoir locations than at upper-reservoir stations. In general, however copepod densities at the Crow Butte backwater far exceeded those observed on the open river (Appendix Fig. B-16).

Rotifers

Twenty-seven rotifer taxa were identified from samples collected at lower- and mid-reservoir stations. Most taxa were rare, appearing infrequently and at very low densities. The dominant taxa, including *Keratella cochlearis*, *Polyarthra vulgaris*, and *Synchaeta pectinata,* were present at lower- and mid-reservoir stations on all sample dates.

A total of 38 rotifer taxa were identified in samples collected at upper-reservoir stations. Taxa found only at upper reservoir sites were typically present at very low densities and on only a few sample dates. *Keratella cochlearis*, *Polyarthra vulgaris*, and *Synchaeta pectinata*, the three most abundant rotifers at lower- and mid-reservoir stations, were also the dominant species at upper-reservoir stations. As with the cladocerans and copepods, rotifer densities at the Crow Butte backwater were greater than at open river locations from March to September 1995.

Benthic Invertebrates

Whereas monthly average benthic invertebrate densities varied from 3,200 to 10,700 organisms/m^2 (Table 2), average benthic invertebrate densities in samples from individual station/depth/date stratum varied from 130 to 32,480 organisms/m^2 (Appendix Table A-29). Data for individual benthic invertebrate samples are omitted from this report but are available in electronic format from NMFS, Point Adams Biological Field Station, P.O. Box 155, Hammond, OR 97121. A list of all benthic invertebrates identified in samples is given in Appendix Table A-30.

Table 2 summarizes densities of dominant benthic invertebrate taxa at John Day Reservoir sampling stations from April 1994 to September 1995. Sample collections were numerically dominated by oligochaetes, the amphipod *Corophium*, ostracods (seed shrimp), chironomids (non-biting midge larvae), nematodes, pelecypods (bivalve molluscs), Hydracarina (water mites), and nemerteans (proboscis worms). Samples collected in most months also contained relatively low densities of Ephemeroptera (mayflies), Trichoptera (caddisflies), Ceratopogonidae (biting midges), mysids (opossum shrimp), gastropods, turbellarians, and various other taxa. Density fluctuations were lower for numerically dominant taxa such as oligochaetes, chironomids, nematodes, and *Corophium* than for the less common benthic invertebrates.

Table 2. Dominant invertebrate taxa in benthic samples collected at John Day Reservoir sampling stations, April 1994 to September 1995.

	Mean number/m^2 *															
Taxon	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Jan	Mar	Apr	May	Jun	Jul	Aug	Sep
Annelida																
Oligochaeta	4,032	3,672	3,878	2,426	3,583	8,069	4,025	3,784	4,322	2,820	2,791	2,181	5,905	3,569	3,249	4,054
Other Annelida (3 taxa)	0	0	7	14	2	0	0	1	0	0	2	1	96	5	9	2
Crustacea																
Corophium spp.	804	311	1,511	884	1,158	292	184	331	303	206	132	134	690	626	737	612
Ostracoda	214	233	121	34	26	102	43	146	101	45	55	104	58	21	14	30
Mysidacea	2	10	13	15	30	1	0	0	1	0	1	1	1	2	5	1
Other Crustacea (4 taxa)	3	1	3	0	1	1	1	0	0	0	2	2	20	36	36	0
Insecta																
Chironomidae	287	522	980	511	648	560	657	561	602	233	319	187	1,181	1,123	1,101	443
Ephemeroptera	17	16	3	4	7	37	59	26	25	13	11	15	42	18	17	13
Diptera	60	5	3	5	0	1	5	0	0	0	0	1	1	0	0	1
Trichoptera	1	0	2	1	57	4	4	0	15	1	1	1	2	8	1	2
Ceratopogonidae	3	2	3	1	0	2	1	0	0	0	0	1	0	0	0	0
Zygoptera	1	1	0	0	0	2	5	0	1	0	16	0	4	16	27	0
Other Insecta (13 taxa)	2	0	2	4	1	2	2	1	0	0	1	1	1	3	2	3
Nematoda	283	434	322	145	210	735	360	528	401	256	317	235	310	170	245	268
Mollusca																
Pelecypoda	145	113	84	61	41	473	207	507	342	189	151	160	102	133	103	1,345
Gastropoda	5	3	3	2	2	8	7	23	5	6	2	2	2	2	17	5
Arachnida																
Prostigmata	155	94	62	126	414	70	81	118	190	98	67	100	219	114	49	10
Other Arachnida (2 taxa)	0	0	1	2	0	0	0	0	0	0	0	1	0	1	1	0
Nemertea	63	36	24	23	64	340	131	195	51	31	29	28	7	7	7	11
Cnidaria																
Hydra spp.	0	0	0	0	0	25	0	9	0	0	0	0	0	0	0	0
Platyhelminthes																
Turbellaria	11	8	2	2	0	2	0	3	7	0	12	3	0	0	2	7
Tardigrada	2	0	0	0	11	0	0	0	1	0	0	0	0	0	0	0
Nematomorpha	0	2	0	7	1	1	1	0	1	0	3	0	0	0	0	0
Unidentified	1	0	1	0	0	0	0	0	0	0	0	0	0	0	1	0
Total	6,091	5,463	7,025	4,267	6,256	10,727	5,773	6,233	6,368	3,898	3,912	3,158	8,641	5,854	5,623	6,807

* Grand means of samples collected at 1-, 3-, and 5-m depths at upper reservoir stations and at the Crow Butte backwater.

Benthic invertebrate densities varied by depth of sample collection in 1994. The overall mean density of samples collected at 3-m depths (8,100 organisms/m^2) was greater than the mean densities of samples collected at 1- and 5-m depths (5,400 and 5,900 organisms/m^2, respectively). Densities of *Corophium* varied with both survey date and depth in 1994. *Corophium* densities were lower in October and November than from June to August, and were less at 1-m depth than at 3- or 5-m depths. Temporal variation was not observed for chironomids or oligochaetes in 1994. These taxa did, however, show density differences with depth, appearing at higher densities in samples collected at the 5-m depth contour than in samples from 1-m or 3-m depths.

Resident and Anadromous Fishes

We completed 116 beach seine hauls, catching a total of 5,830 fishes at upper-reservoir stations from May 1994 to September 1995 (Table 3). Eleven common resident taxa were represented in the catch, along with three migratory species, American shad (*Alosa sapidissima)*, chinook salmon (*Oncorhynchus tshawytscha*), and coho salmon (*Oncorhynchus kisutch*). Except for a few adult prickly sculpins (*Cottus asper)*, suckers (*Catostomus* spp.), and common carp (*Cyprinus carpio*), catches consisted almost entirely of juvenile fishes.

American shad were the most numerous species in the catch (4,301 fish), followed by chinook salmon (584 fish), northern pikeminnow (*Ptychocheilus oregonensis)* (404 fish), smallmouth bass (*Micropterus dolomieui*) (152 fish), and sunfishes (*Lepomis* spp.) (89 fish). Juvenile shad attained a size large enough (about 35 mm) to be retained in the net in early August and were present in the catch from August to October in 1994 and from August to September in 1995. Chinook salmon subsampled from the catch in 1994 (n = 25) had a mean fork length of 74 mm (range 42 to 98 mm). Also in 1994, about 91% of the chinook salmon were captured during May, 8% during June, and less than 1% during early July. Chinook salmon were not present in the catch from late July through November. Both the size data and time of occurrence at upper-reservoir stations suggest that the chinook salmon we observed were 0-age upriver bright stock fall chinook. Prevalent resident species (northern pikeminnow, smallmouth bass, and sunfishes) were captured in greatest numbers during August, September, or October.

Table 3. Observed beach seine catch at upper John Day Reservoir stations, May 1994 to September 1995.

	Month and (number of beach seine sets)[a]													
Taxon	May (12)	Jun (11)	Jul (8)	Aug (12)	Sep (5)	Oct (5)	Nov (6)	Jan (6)	Apr (6)	May (7)	Jun (2)	Jul (12)	Aug (12)	Sept (12)
American shad	0	0	0	349	167	143	0	0	0	0	0	0	822	2,820
Common carp	4	1	2	3	0	3	1	0	2	0	0	8	1	7
Chinook salmon	255	24	2	0	0	0	0	0	16	178	103	6	0	0
Coho salmon	0	0	0	0	0	0	0	0	0	10	0	0	0	0
Crappies	0	0	0	1	5	0	11	0	0	0	0	0	0	0
Largemouth bass	0	0	1	0	0	0	1	0	0	0	0	1	0	114
Peamouth	0	0	0	2	3	3	0	0	0	0	0	0	0	0
Prickley sculpin	2	1	4	3	2	0	0	1	4	4	1	8	2	2
Smallmouth bass	0	0	0	89	39	10	3	0	0	0	2	0	7	2
Northern pikeminnow	1	0	3	13	29	137	2	1	0	0	0	0	22	196
Sunfishes[b]	0	0	0	0	0	77	0	0	0	0	0	0	0	12
Suckers	0	0	8	9	0	1	0	0	0	0	0	1	11	9
Whitefish	1	1	0	0	0	0	0	0	0	0	0	0	0	0
Yellow perch	0	10	1	2	0	2	0	4	1	1	3	4	4	1
Unidentified	0	0	4	1	0	0	0	0	0	0	0	3	0	0

[a] After 8 June 1994 all beach seine sets were made at the Long Walk Island (downstream) and Paterson Slough stations.
[b] Bluegill and pumpkinseed.

DISCUSSION

Physical Variables

Typical direct thermal stratification is characterized by a uniformly warmer upper layer, a middle layer in which temperature decreases at least 1°C for every 1-m increase in depth, and a lower layer of uniformly cooler water (Cole 1983). Within the Columbia River Basin, thermal stratification has been documented in Franklin D. Roosevelt Lake, the storage reservoir for the Grand Coulee Project (Stober et al. 1977), and in Brownlee Reservoir on the Snake River (Ebel and Koski 1968). In John Day Reservoir and in main-stem run-of-the-river reservoirs thermal stratification does not develop due to low water-retention times (Stober et al. 1979).

Vertical temperature profiles at John Day Reservoir sampling stations did not show thermal stratification, although there was a pronounced (about 2°C) warming from a depth of 15 m to the surface in spring. Temperature differences within the water column diminished through spring and summer. By winter, temperature profiles showed isothermal conditions.

In comparison with results of Funk et al. (1979) who noted less than a 1°C increase in temperature for passage of Snake River water through four reservoirs at time of maximal heat content, we noted differences of 0.3°C and 1.8°C between upper and lower John Day Reservoir locations in August 1994 and July 1995, respectively.

The extent to which light penetrates into waters is a function of turbidity and true color of the water. Available measures of water transparency include Secchi disk visibility, coefficients of absorption computed from readings taken with a subsurface photometer, and turbidity. General relationships exist between Secchi disk visibility, coefficients of absorption, and depth of the euphotic zone (Cole 1983). For example, the constant 1.7 divided by Secchi depth in meters often gives a result approximating the value of the vertical light absorption coefficient calculated from photometer readings. Depth of the euphotic zone may be estimated by multiplying Secchi depth by the factor 3.0.

Secchi disk visibility measured in John Day Reservoir varied from 0.5 to 3 m during the study. Variation was largely seasonal, reflecting higher turbidity during system-wide runoff in winter and spring and maximum clarity during winter periods when runoff and abundance of algae were at minimum levels. Station-to-station differences in Secchi depth were observed, but were usually due to local influences, such as turbid water entering from nearby tributaries such as the Umatilla or John Day Rivers or from wind-generated waves causing resuspension of fine sediments at some nearshore stations. An exception was noted, in that Secchi depth was usually less at the Crow Butte backwater than at nearby upper-reservoir stations, probably because of the greater abundance of phytoplankton at Crow Butte.

Vertical light extinction coefficients less than 1.0 indicate relatively clear waters. In comparison, average monthly vertical light extinction coefficients in John Day Reservoir from June 1994 to September 1995 varied from 0.77 to 2.85. Water clarity during the June to September period was greater in 1994 than in 1995 as shown by lower extinction coefficients.

Turbidity measurements reflected the station-to-station differences due to tributary inflow as well as differences in water clarity between 1994 and 1995. Turbidity decreased slightly from

upstream to downstream within the reservoir and increased from surface to bottom at the deeper sampling stations, trends which were likely due to settling of suspended materials.

Sediment samples were obtained at upper-reservoir stations selected for collection of soft-substrate benthic invertebrate samples. Although these locations were typical of much reservoir habitat, particularly the extensive shallow-water areas of the upper reservoir, other habitat types such as gravel and other hard substrates were not sampled. Li et al. (1981) conducted shoreline surveys of habitat along both the Washington and Oregon shores as part of a study of resident fish species in John Day Reservoir. Results of their study quantify shoreline habitat types and illustrate the increasing amount of hard substrate from upper to lower reservoir. Sediment analysis indicated differences in organic content and grain size between sampling stations used in our study. These differences are discussed in relation to benthic invertebrate collections described in a following section.

Chemical Variables

Other reservoir attributes for which surface-to-bottom profiles were taken at mid- and lower-reservoir stations included dissolved oxygen, pH, and specific conductance. Oxygen content of lakes typically decreases with depth during the growing season. Near-surface oxygen values often exceed saturation levels due to photosynthesis within the euphotic zone, whereas oxygen content at greater depths may be depleted due to respiration and decomposition of organic matter. The extent of oxygen decline with depth at John Day Reservoir sampling stations from spring to fall was slight, amounting to only about 0.5 mg/L in profiles covering 20- to 30-m depths. Expressed in terms of percent saturation, minimum and maximum oxygen concentrations observed in 1994 ranged from 86 to 116%. Funk et al. (1979) reported a much wider range of saturations (27 to 144%) from lower Snake River reservoirs during periods of pronounced oxygen stratification in 1977. Two differences were noted between upper and lower reservoir sampling stations: 1) oxygen content was slightly higher at upper-reservoir stations, and 2) oxygen saturation at upper-reservoir stations varied over a wider range than at lower reservoir stations.

Differences in vertical distribution of pH and alkalinity within the water column occur in some lakes as a result of photosynthetic utilization of CO_2 in the trophogenic zone and respiratory generation of CO_2 throughout the water column (Wetzel 1983). Where such differences exist, they tend to be marked more by pH differences than alkalinity changes within the water column. Changes observed for pH within the water column at lower- and mid-reservoir sampling stations were usually less than 0.3 pH units. These variations may reflect biological processes within the water column or may have resulted from the sampling technique (accuracy of the meter was ± 0.2 pH units; pH is difficult to measure accurately in waters of less than 200 µS/cm specific conductance).

Seasonal pH variation was characterized by highest values (8.6 pH units) in January and February and lowest values (7.7 pH units) in mid September. There were no clear differences in pH between upper-, mid-, or lower-reservoir sampling stations located near the main river channel. At the Crow Butte backwater, however, pH values differed from those at near-channel stations by 0.1 to 1.0 pH units (average 0.6 pH units) for samples collected in the same weeks. Since pH increases with the level of primary production, the higher pH values observed at Crow Butte suggest a higher level of productivity at that location than at other reservoir stations.

Specific conductance is a measure of the conductivity of water. Since conductivity depends on ion concentration, specific conductance is proportional to major ions concentration and, as such, is often used as a substitute for determination of total dissolved solids (Wetzel 1983). Specific conductance values measured in John Day Reservoir ranged from 112 to 191 µS/cm during the study period. Lowest specific conductance coincided with periods of high runoff and highest values were observed during low runoff periods in winter. Specific conductance was among the most stable water quality parameters relative to spatial variability. No discernable spatial differences were observed, either between stations or with depth of sample collection. Within the Columbia River system, specific conductance values reflect the qualities of individual drainage basins. Funk et al. (1979) reported ranges of 100 to 300 µS/cm and 50 to 70 µS/cm from the Snake and Columbia Rivers, respectively, at locations above their confluence. Values of specific conductance observed in John Day Reservoir reflect the composite nature of waters arriving from upstream.

The alkalinity of the Columbia River is essentially bicarbonate, derived from the weathering of sedimentary carbonate rocks within the drainage basin (Park et al. 1969). Some tributaries such as the Snake and Yakima Rivers are relatively rich in alkalinity, while others, such as the Clearwater River, are less alkaline. Park et al. (1969) found that alkalinity at the mouth of the Columbia River was relatively consistent, averaging about 1 meq/L regardless of river flow. Our sampling of John Day Reservoir showed similar average monthly alkalinity values (range 46 to 75 mg/L $CaCO_3$ or 0.9 to 1.5 meq/L). We did not observe biologically mediated decreases in alkalinity values from upstream to downstream locations as noted by Funk et al. (1979) from Rkm 248 downstream to Rkm 190 in the lower Snake River reservoir series, probably because of the comparatively short retention time of water in John Day Reservoir. Seasonal variation of alkalinity in John Day Reservoir was characterized by highest values during low river flow in winter and lowest values during high river flow.

Concentrations of the major cations (calcium, magnesium, sodium, and potassium), and the major anions (sulfate and chloride) in John Day Reservoir waters were similar to mean values calculated for North American rivers by Livingstone (1963). Also, the dominance relationships of cations and anions in John Day Reservoir waters were typical of those observed in many rivers and temperate zone open lake systems (i.e., calcium > magnesium ³ sodium > potassium and sulfate > chloride). Similarities in ionic concentration and dominance are common, particularly for rivers and open lakes with large drainage basins. Since ionic composition is derived primarily from weathering of soil and rock within drainage basins and because large basins tend to include a range of soil and rock types, many large rivers tend toward similar ionic content.

Concentrations of magnesium, sodium, potassium, and chloride undergo relatively minor changes from biotic utilization in the environment. Calcium and sulfate can exhibit marked seasonal and spatial dynamics as a result of microbial metabolism (Wetzel 1983). The major ions are usually considered to be in superabundance relative to biological needs (Funk et al. 1979). Stober et al. (1979), in a review of available Columbia River data, noted that sodium, chloride, and sulfate generally increased from the Canadian border to The Dalles, Oregon.

In John Day Reservoir, lowest concentrations of major ions coincided with periods of high runoff and highest concentrations were observed during low runoff periods in winter. There were no discernable differences between stations or within the water column.

In contrast to the common inverse relationship between discharge rate and concentration of some constituents of river waters, silica content is often relatively stable regardless of flow (Wetzel 1983). In John Day Reservoir, however, silica concentrations varied fourfold during the sampling season with minima noted in fall and maxima in spring. Although temporal changes in silica concentrations were roughly similar to those of the major ions, the decline in silica concentrations continued until early October, whereas ion concentrations tended to start increasing by July. Because the decline in silica continued throughout the growing season, it may have been more closely related to uptake of silica by diatom algae than to changes in river flow. Within John Day Reservoir there was little indication of spatial differences in silica concentrations between samples collected at near-channel stations or obtained from different depths within the water column. The silica content of samples collected at the Crow Butte backwater, however, averaged about 21% less than at nearby upper-reservoir stations, suggesting a higher degree of silica uptake by diatoms at Crow Butte that at other reservoir locations .

Nitrate and phosphate are the principal nutrients that support, and in some cases, limit primary production along the main stream Columbia River. Silica, an essential component in the cell walls of diatoms, is also considered a nutrient, but is seldom limiting to phytoplankton production. Temporal variation of phosphate, nitrate, and silica concentrations in the Columbia River system is characterized by maxima in winter and minima during summer months (Park et al. 1970). Summer nutrient levels reported by Park et al. (1970) showed a fourfold decrease in phosphate and a sevenfold decrease in nitrate between Pasco, Washington and Clatskanie, Oregon; this nutrient reduction was attributed to in-stream primary productivity.

Ammonia nitrogen results from bacterial decomposition of organic matter and, to a lesser extent, is introduced into water as an excretory product of aquatic animals. Ammonia is the nitrogen form most readily assimilated by algae and, as a consequence, ammonia levels often fluctuate rapidly. Probably because of the rapid uptake by algae, ammonia levels in John Day Reservoir varied widely among lower-, mid-, and upper-reservoir stations sampled during the same week. Spatial trends in ammonia levels, if present, were not readily apparent. Seasonal ammonia variation was characterized by highest levels from late April through October and lowest ammonia levels from November through early April.

Total oxidized nitrogen was determined as the sum of nitrite and nitrate nitrogen. Nitrite nitrogen is an intermediate stage in the nitrification of ammonia or the denitrification of nitrate. Because nitrite is readily oxidized, it is usually present in low concentrations. Nitrate nitrogen is present in surface runoff, groundwater, and precipitation and, in concentrations exceeding 0.3 mg/L, may result in algal blooms (Stober and Nakatani 1992). Seasonal variation in nitrite + nitrate nitrogen in John Day Reservoir was similar to that observed by Park et al. (1970). Nitrite + nitrate levels were at seasonal highs (0.5 to 0.6 mg/L) during winter, declined rapidly with the onset of the growing season in April, and eventually were reduced to low levels (<0.1 mg/L) at times during the growing season. Nitrite + nitrate values were consistently lower at the backwater habitat at Crow Butte than at near-channel sampling stations, most likely because of greater abundance of phytoplankton and lower turnover rate of water at the Crow Butte station.

Water samples collected from John Day Reservoir were also analyzed for total nitrogen content, defined as the combined concentrations of ammonia, nitrite + nitrate nitrogen, and organic

nitrogen compounds including proteins, peptides, nucleic acids, and urea. Seasonal variation in total nitrogen concentration was similar to that of nitrite + nitrate nitrogen. Total nitrogen values were similar at the near-channel sampling stations of the upper-, mid-, and lower-reservoir. In comparison, total nitrogen values at the Crow Butte backwater were usually greater than at the near-channel stations.

Wetzel (1983) states that the dissolved organic nitrogen of fresh waters often contributes over one-half of total soluble nitrogen. In comparison, the organic nitrogen component in reservoir waters was between 50 and 90% from April to September, then declined to 16% in January, February and March. This temporal variation of organic nitrogen was, as expected, essentially the opposite of nitrite + nitrate nitrogen which is at lowest levels during periods of high biotic activity late spring, summer, and fall.

Phosphorus is relatively less abundant than other requirements of biota, such as carbon, nitrogen, and oxygen. Because of this scarcity, phosphorus is commonly the first element to limit biological productivity (Wetzel 1983). Natural sources of phosphorus are via leaching from rocks and decomposition of organic matter; manmade sources include fertilizers, sewage, and detergents (Lind 1979). Phosphorus is lost through chemical precipitation to sediments and adsorption.

Laboratory analyses of John Day Reservoir samples included determinations of orthophosphate phosphorus and total phosphorus content. Orthophosphate phosphorus is directly utilizable by algae. Values observed in John Day Reservoir varied inversely to phytoplankton abundance, declining from maximum levels in winter and early spring (20 to 30 µg/L) to minimum values (<5 µg/L) from late spring through early fall. In comparison, Funk et al. (1979) observed mean orthophosphate phosphorus content ranging from 6 to 38 µg/L in lower Snake River reservoirs. Orthophosphate phosphorus values varied considerably between John Day Reservoir locations with no increasing or decreasing trends evident from upper to lower reservoir. Orthophosphate phosphorus levels at the Crow Butte backwater were typically lower than at other reservoir locations sampled during the same week.

Total phosphorus includes orthophosphate phosphorus as well as various organic forms bound in living organisms, especially algae (Wetzel 1983). Total phosphorus values observed in John Day Reservoir ranged from 18 to 83 µg/L at upper-, mid-, and lower-reservoir stations. This range of values compares with the range of 6 to 38 µg/L reported by Funk et al. (1979) in the Lower Snake River and seasonal mean concentrations of 5.9 to 8.3 µg/L reported by Teuscher et al. (1994) for oligotrophic lakes in the Sawtooth Valley of Idaho. Seasonal variation was similar to orthophosphate phosphorus, although somewhat less variation was observed between sampling stations. No increasing or decreasing trends were evident from upper to lower reservoir.

Biological Variables

Conversion of a free flowing river into a reservoir causes a fundamental change in sources of biotic energy. In the river environment, organic material originating elsewhere is metabolized by heterotrophic organisms, a process which may provide a high percentage of the energy input (Baxter 1977). This process is augmented by primary production in the periphyton (algae attached to river substrates) and by aquatic macrophytes.

Phytoplankton populations in rivers are typically low, due to continual removal by current. Following impoundment of the river, however, phytoplankton populations develop and contribute significantly to aquatic productivity, whereas periphyton populations and aquatic macrophytes are restricted to littoral zones and shallow-water areas of the newly-formed reservoir.

Measurement of the chlorophyll *a* content of reservoir waters provides an index of phytoplankton abundance which is, in turn, determined by factors including water clarity, nutrient levels, temperature, flow, and utilization by zooplankton. Chlorophyll *a* levels (monthly averages for lower-, mid-, and upper-reservoir stations) during the 1994-95 sampling period peaked in spring (April and May) and again in fall (September and October) a periodicity characteristic of Columbia River reservoirs (Stober and Nakatani 1992).

The maximum average chlorophyll *a* value observed during this study (10.6 mg/m^3 in April 1995) appears to be typical for a Columbia River system location. Peak chlorophyll *a* levels documented at other locations include 11.1 mg/m^3 in the forebay of Roosevelt Lake (Stober et al. 1977), 10.35 mg/m^3 in Rufus Woods Reservoir (Erickson et al. 1977), and 8.25 mg/m^3 at the Hanford Reach (Wolf et al. 1976). Funk et al. (1979) combined sampling data to obtain 2-year mean values of 4.9 and 2.5 mg/m^3 for Lower Granite Reservoir stations in 1976 and 1977, respectively.

Short water-retention times in run-of-the-river reservoirs are not conducive to buildup of phytoplankton populations (Erickson et al. 1977). Although John Day Reservoir is classified as a storage reservoir, retention times are short, ranging from 3.8 days at flows of 8,495 m^3/second to 11.2 days at river flow of 2,830 m^3/second (COE 1992). Based on chlorophyll *a* levels measured during study of John Day Reservoir in 1994 and 1995, phytoplankton populations are, as at other Columbia River locations, limited by water retention time. This contention is supported by comparison of average chlorophyll *a* levels between samples collected at upper-reservoir stations located near the main river channel and samples collected at the protected Crow Butte backwater during 1995. Peak chlorophyll *a* level observed at Crow Butte was 34.9 mg/m^3 during April 1995; an average of 12.3 mg/m^3 chlorophyll *a* was observed at upper-reservoir stations during the same week. In all other comparisons between samples collected during the same week at upper-reservoir stations and at Crow Butte, chlorophyll *a* values at Crow Butte were higher.

Factors commonly affecting rotifer abundance include temperature, food availability, and interactions with other zooplankters, particularly the cladocerans. In 1994, as water temperature at lower- and mid-reservoir stations increased from spring to summer, rotifer abundance also increased. In August and September 1994, although water temperature at lower- and mid-reservoir stations remained elevated (19 to 20°C), rotifer abundance declined to only 13% of the level observed during July 1994. This abrupt drop in rotifer abundance from July to September at mid- and lower-reservoir stations could not be explained by food availability, since chlorophyll *a* analysis indicated essentially constant levels of small plankton (<25 µm) throughout the April to November 1994 sampling period. Cladoceran abundance, however, increased from about 1 organism/L in June 1994 to 4.5 organisms/L in July 1994 and peaked at 23 organisms/L in August 1994. Rotifer abundance at lower- and mid-reservoir stations may have been reduced by feeding competition from the increasing cladoceran population, or through predation by insect larvae, copepods, or larval fish.

Mean cladoceran densities at John Day Reservoir stations from August to October and from May to September ranged from 2 to 25 organisms/L and from less than 1 to 25 organisms/L, respectively. Funk et al. (1979) reported low cladoceran concentrations (<1 to 46 organisms/L) in samples collected from lower Snake River reservoirs from August to October 1977. Samples collected at a John Day Reservoir backwater from May to September 1980 contained higher densities of cladocerans (<1 to 85 organisms/L)(Parente and Smith 1981).

The cladoceran densities we observed in John Day Reservoir were higher than densities typically observed in free-flowing rivers, but lower than expected for a non-oligotrophic lacustrine environment. Cladoceran population size and species composition are frequently regulated by predation. Selection by fish tends to remove larger forms such as *Daphnia* spp. from the population, resulting in dominance of small cladocerans such as *Bosmina* spp. When predation is not a factor, larger cladocerans tend to dominate smaller species. These interactions may explain two trends observed in our data: 1) in 1994 and 1995, *Daphnia* abundance declined and *Bosmina* became more abundant from August through the fall months, and 2) in 1994, the peak abundance of *Daphnia* at upper-reservoir stations was only about 10% of the peak observed at lower- and mid-reservoir stations. Increased predation in September and mid-October by young-of-the-year fish would be expected, as would relatively higher predation rates in the nearshore habitats adjacent to upper-reservoir stations.

Alternatively, variations in *Daphnia* and *Bosmina* populations may have been due to food availability or normal seasonal fluctuations.

Mean copepod densities at John Day Reservoir stations in 1994 varied from less than 1 to 42 organisms/L during the sampling period and were generally greater than copepod densities reported from Snake River reservoirs in 1977 (<1 to 12 organisms/L)(Funk et al. 1979) or from a John Day Reservoir backwater in 1980 (<1 to 8 organisms/L)(Parente and Smith 1981).

Copepod populations are commonly dominated by one species of cyclopoid copepod and one species of calanoid copepod (Thorp and Covich 1991). Aside from the occurrence of *Eucyclops agilis* on a single date at upper-reservoir stations, John Day Reservoir zooplankton samples obtained in 1994 contained only the cyclopoid copepod *Diacyclops thomasi*, the calanoid copepod *Leptodiaptomus ashlandi*, and copepodids presumed to be immature life stages of these species. Copepod densities were typically higher at lower- and mid- than at upper-reservoir stations, possibly due to differential predation rates. Although copepods are better able to evade capture than the cladocerans, copepods are prey for other copepods, as well as for larval, juvenile, and adult fish. Life cycles of copepods are variable, although developmental time is greater than in rotifers or cladocerans due to the progression through immature life stages. Copepod abundance generally increased from April to October at lower-, mid- and upper-reservoir sampling stations.

Benthic invertebrate data were highly variable, a consequence of differing substrate types within sampling areas and of the difficulties inherent in collecting representative benthic samples (Wetzel and Likens 1979). Mean total benthic invertebrate densities for samples collected at monthly intervals from April 1994 to November 1995 (no samples were collected in December 1994 or February 1995) ranged from 3,200 to 10,700 organisms/m^2. In comparison, benthic samples collected from a John Day Reservoir backwater in November 1979 and in June and

August 1980 contained relatively low mean densities of 140 to 1,540 organisms/m^2 (Parente and Smith 1981).

Benthic invertebrate populations were characterized by numerical dominance and relatively stable densities of oligochaetes, chironomids, and nematodes. Reproduction in oligochaetes and nematodes is usually considered to be approximately continuous, while chironomid species may exhibit one or more annual life cycles. Seasonal periodicity was not evident for chironomids in 1994 but definite peak abundances were noted in June, July, and August 1995. For most major taxa, densities were relatively high in spring, declined to seasonal lows during summer, then increased to relatively high levels in the fall. Taxa present at lower densities during the summer months included nemerteans, which are frequently most abundant in autumn, pelecypods, and ostracods. *Corophium* differed most notably from this seasonal trend in that higher *Corophium* densities were observed during the summer months than during fall.

Total benthic invertebrate densities varied by depth of sample collection in 1994. Overall mean densities at the 1- and 5-m depths (5,400 and 5,900 organisms/m^2) were less than overall mean density at the 3-m depth (8,100 organisms/m^2). These results, however, tend to obscure density vs. depth trends shown by individual taxa. Although most amphipods prefer waters less than 1 m deep (Pennak 1989), we noted higher *Corophium* densities at 5 m than at 1 m or 3 m. Nemerteans, or proboscis worms, were also present at higher densities at 5 m than at 1 m or 3 m. Densities of other taxa tended to decrease with depth. Oligochaetes, chironomids, and water mites were more abundant at 1- and 3-m depths than at the 5-m depth. Organic content at sample depths was similar, thus differences in oligochaete densities could not be explained on the basis of organic content. Similarly, we were unable to account for the greater densities of chironomids at shallower depths. Water mites, however, are typically most abundant in 1 to 2 m of water and are predaceous on worms and small insects (Pennak 1989), likely reasons for our observations of higher water-mite densities at the shallower depths.

Although station-to-station differences in benthic invertebrate densities were not considered in the study design, such differences were apparent during sampling, and later, during examination of the data. Stations differed noticeably with respect to bottom contour, current velocity, exposure to wave action, and substrate composition. The Sand Island and Big Blalock Island stations were characterized by steep bottom contours, high current velocities relative to other stations, and minimal wave action. The Long Walk Island stations, in comparison, were characterized by gentler bottom contours, relatively low current velocities, and moderate wave action. In regard to these characteristics, the Paterson Slough station was somewhat intermediate between groupings. The results of sediment analysis indicated that organic content at the Long Walk Island stations was about twice that of the Sand Island, Big Blalock Island, and Paterson Slough stations. Median grain size, on the other hand, was about 50% larger at the Big Blalock Island and Sand Island stations than at other stations.

With the exception of pelecypods, individual benthic invertebrate taxa tended to exhibit preference for either the Long Walk Island or Sand Island/Big Blalock Island habitat types. Oligochaetes were about twice as abundant at the Long Walk Island stations as at other stations, probably because of the higher organic content at Long Walk Island. Chironomids, nematodes, ostracods, Ephemeroptera larvae, and water mites were also more abundant at the Long Walk Island stations. Nemerteans and *Corophium*, however, were present at higher densities at Sand Island and at Big Blalock Island than at the Long Walk Island stations. *Corophium* densities

appeared more dependent on suitable current velocity than on grain size, since high *Corophium* densities were also present at the Paterson Slough station, despite the small grain size at this location.

Various studies have documented the importance of invertebrates in the diet of resident and anadromous fish in the Columbia River system. Griffith et al. (1995) found that while cladocerans (*Daphnia sch□dleri*) were the principal prey of Lake Roosevelt kokanee salmon during all seasons, chironomid larvae were important during summer. Parente and Smith (1981) found that from March to June, juvenile chinook salmon (mean length 75 mm) in a John Day Reservoir backwater were feeding on Cladocera (47% of stomach contents by volume), Ephemeroptera (35%), and Diptera (18%). Cladocerans and insects were the principal dietary components for subyearling chinook salmon captured at Jones Beach (RKm 75) in the upper Columbia River estuary (Kirn et al. 1986; Ledgerwood et al. 1990, 1991, 1994). For Columbia River estuary locations farther downstream, McCabe et al. (1983, 1986) concluded that *Corophium salmonis* was a primary prey for juvenile chinook salmon, coho salmon, and steelhead.

Previous research has documented use of littoral areas of John Day Reservoir by resident and anadromous species (Li et al. 1981). Although beach seines have often been used in littoral areas, catches may not be representative of true abundance or species composition (Parsley et al. 1989). Further, the selection of fishing site probably results in a degree of bias since few shoreline locations are suitable for seining due to substrate, current, or bottom contour. An indication that our seine catches may not be representative is the scarcity of adult fishes in the catch, particularly northern pikeminnow and smallmouth bass. Our results do serve to document use of littoral areas by juvenile salmon from April through July and by young-of-the-year of various resident species from summer through fall.

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APPENDIX A

Appendix Table A-1. Water temperatures (surface to bottom averages; °C) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/11/94	9.5	9.0	9.4						
4/13/94					9.0	8.8	8.7	9.1	9.2
4/26/94					11.3	11.3	11.4	11.3	11.6
5/10/94	14.7	13.4	13.3						
5/11/94					13.0	12.9	13.2	13.0	14.0
5/24/94					15.1	14.3	14.3	14.2	14.9
6/7/94	15.2	14.9							
6/8/94					15.2	14.7	14.7	14.5	14.7
6/9/94			14.9						
6/21/94					16.5	16.3	16.3	16.3	17.3
7/6/94	18.0	17.9	18.4						
7/7/94					18.2	18.0	17.9	18.1	18.7
7/20/94					20.9	21.0	20.9	20.9	21.2
8/2/94	22.0	22.0	21.8						
8/4/94					21.7	21.7	21.6	21.6	21.7
8/18/94					21.5	21.6	21.6	21.6	21.8
9/14/94					19.7	19.8	19.8	19.8	19.9
9/15/94	19.9	19.9	19.7						
10/4/94	19.4	19.3	18.9						
10/6/94					18.1	18.3	18.3	18.3	18.3
11/15/94	10.6	10.6	10.4						
11/17/94					10.1	10.2	10.2	10.1	9.8
12/13/94					6.3	6.4	6.4	6.3	6.2
12/14/94	6.6	6.5	6.0						
1/10/95	4.2	3.9	3.4						
1/25/95					3.6	3.6	3.6	3.6	3.5
2/21/95	4.8	3.4	3.4						
2/22/95					4.8	4.3	4.2	5.0	4.2
3/13/95	5.5	4.7	4.8						
3/15/95				7.8	6.0	5.5	5.6	6.1	5.9
4/4/95				10.6	7.9	7.8	8.0	7.8	8.3
4/18/95				10.5					
4/20/95					9.2	9.1	9.2	9.2	9.3
4/21/95	9.4	9.3	9.1						
5/1/95				12.2					
5/2/95									10.2
5/3/95					10.3	10.3	10.3	10.3	
5/15/95	12.9	12.6	12.8						
5/17/95				15.9	12.4	12.4	12.6	12.5	13.0
5/30/95				17.6					
5/31/95					15.2	15.4	15.5	15.3	16.2
6/12/95	15.6	15.6	15.4						
6/14/95					14.7	14.7	14.7	14.6	14.7

Appendix Table A-1. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
6/15/95				15.8					
6/28/95				18.0					
6/29/95					16.6	16.7	16.7	16.6	16.8
7/10/95	18.8	18.7	19.3						
7/12/95					19.1	19.0	18.9	19.1	19.0
7/13/95				20.5					
7/24/95				21.8					
7/25/95					20.7	20.5	20.5	20.7	20.7
8/8/95				20.4	20.3	20.4	20.4	20.3	20.2
8/10/95	21.1	21.1	20.4						
8/21/95				20.0					
8/22/95					19.3	19.6	19.1	19.3	19.6
9/5/95				20.9					
9/7/95					20.6	20.5	20.2	20.2	20.5
9/8/95	20.3	20.3	20.4						
9/19/95				21.0	21.1	20.9	20.7	20.7	20.8

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

Appendix Table A-2. Secchi disk readings (m) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/11/94	0.6	1.5	1.4						
4/12/94									
4/13/94					0.9	1.5	1.5	0.9	1.3
4/26/94					1.3	1.5	1.4	1.3	1.5
5/10/94	1.2	1.5	1.4						
5/11/94					1.4	1.4	1.4	1.4	1.4
5/12/94									
5/13/94									
5/24/94					1.2	1.5	1.5	1.4	1.4
6/7/94	1.5	1.5	--[b]						
6/8/94					1.2	1.5	1.5	1.4	1.8
6/9/94			1.5						
6/21/94					1.5	1.5	1.5	1.5	1.5
7/6/94	1.5	1.7	1.8						
7/7/94					1.4	1.4	1.4	1.4	1.4
7/20/94					1.7	1.8	2.0	1.7	1.7
8/2/94	2.7	3.4	2.1						
8/3/94									
8/4/94					1.4	2.3	2.0	2.0	1.5
8/18/94					1.7	2.1	2.1	1.7	1.8
9/13/94									
9/14/94					1.7	--	1.8	1.8	2.1
9/15/94	2.3	2.4	2.0						
10/4/94	2.1	2.3	1.5						
10/5/94									
10/6/94					1.8	1.7	1.7	1.7	1.8
11/15/94	3.0	3.3	3.0						
11/17/94					2.7	2.8	2.8	2.4	3.0
11/18/94									
12/13/94					3.2	4.2	3.5	2.9	4.4
12/14/94	--	2.7	3.2						
1/10/95	3.2	3.4	3.2						
1/24/95									
1/25/95					0.7	0.5	0.5	0.4	0.5
2/21/95	0.2	0.6	0.9						
2/22/95					0.5	1.1	1.1	0.5	1.1
3/13/95	0.9	1.2	1.4						
3/14/95									
3/15/95				0.9	0.8	0.9	0.9	0.8	1.2
4/4/95				0.9	1.1	1.2	1.1	1.1	0.9
4/18/95				0.9					
4/19/95									
4/20/95					1.2	--	1.4	1.2	1.2

Appendix Table A-2. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/21/95	0.8	1.3	1.4						
5/1/95				0.9					
5/2/95									1.1
5/3/95					0.5	0.9	1.0	0.5	
5/15/95	0.6	0.9	0.8						
5/16/95									
5/17/95				0.5	0.5	0.8	0.6	0.5	0.6
5/30/95				1.4					
5/31/95					0.9	1.2	1.2	1.1	1.4
6/12/95	1.1	1.4	1.2						
6/13/95									
6/14/95					1.1	1.1	1.1	0.9	1.1
6/15/95				1.5					
6/28/95				1.2					
6/29/95					1.1	1.2	1.2	1.1	1.4
7/10/95	1.4	1.4	1.2						
7/11/95									
7/12/95					0.9	1.4	1.2	1.1	1.4
7/13/95				0.9					
7/24/95				1.1					
7/25/95					0.8	1.1	1.2	1.1	1.1
8/8/95				0.9	1.2	1.4	1.4	1.2	1.2
8/9/95									
8/10/95	1.5	1.4	1.2						
8/21/95				1.7					
8/22/95					--	1.2	1.2	1.2	1.1
9/5/95				0.9					
9/6/95									
9/7/95					1.1	1.5	1.5	1.3	1.1
9/8/95	1.8	1.8	1.7						
9/19/95				1.1	1.4	1.5	1.5	1.2	1.2

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Dashes indicate data are not available.

Appendix Table A-3. Turbidity measurements (NTU) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	--[c]	5	5	4	5	--	4	5	5												
4/13/94												6	6	4	4	4	5	8	8	4	4
4/26/94												6	8	6	--	4	4	6	6	5	5
5/10/94	5	5	7	4	5	5	4	7	--												
5/11/94												6	10	6	5	5	5	--	6	5	5
5/24/94												7	8	6	6	7	7	8	8	6	6
6/7/94	5	5	2	5	6	--															
6/8/94												6	8	4	4	4	5	6	6	5	5
6/9/94							5	4	6												
6/21/94												4	5	5	5	5	5	5	5	4	4
7/6/94																					
7/7/94												6	8	5	6	5	5	5	7	5	6
7/20/94												5	6	4	4	4	4	5	5	4	5
8/2/94	2	2	5	1	2	4	2	3	8												
8/4/94												4	6	2	3	3	3	3	4	3	3
8/18/94												3	4	3	3	3	3	4	4	3	4
9/14/94												2	3	2	2	3	3	3	5	2	3
9/15/94	2	4	5	2	4	4	4	4	5												
10/4/94	3	3	4	3	3	6	4	5	6												
10/6/94												3	3	3	3	3	3	3	5	2	3
11/15/94	2	2	7	2	2	7	2	2	3												
11/17/94												2	5	2	3	2	3	2	3	2	2
12/13/94												2	2	1	1	2	2	2	2	1	1
12/14/94	2	2	2	2	2	2	2	2	3												
1/10/95	2	2	2	2	2	2	2	2	2												
1/25/95												24	24	21	21	22	22	25	26	21	20
2/21/95	42	39	37	11	11	11	8	9	8												
2/22/95												24	42	9	8	8	8	26	32	8	8
3/13/95	9	8	7	6	6	6	6	6	6												
3/15/95										7	8	13	13	8	8	7	7	12	12	7	6
4/4/95										8	8	11	11	9	9	9	9	10	10	10	10
4/18/95										10	11										

Appendix Table A-3. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												9	9	6	6	6	6	8	8	6	7
4/21/95	10	9	10	6	7	7	7	7	7												
5/1/95										8	9										
5/2/95																				8	7
5/3/95												27	26	8	8	7	7	25	26		
5/15/95	10	10	13	7	8	9	9	10	13												
5/17/95										10	11	16	17	13	12	12	12	19	18	11	12
5/30/95										4	4										
5/31/95												7	9	6	5	6	5	7	7	6	5
6/12/95	7	7	15	5	6	8	5	6	12												
6/14/95												10	11	7	6	7	7	8	8	7	7
6/15/95										5	7										
6/28/95										4	5										
6/29/95												8	7	6	5	6	5	7	7	4	5
7/10/95	4	5	8	7	6	7	6	5	7												
7/12/95												9	13	8	6	9	5	7	6	5	6
7/13/95										10	8										
7/24/95										7	12										
7/25/95												8	13	7	6	6	7	7	8	6	6
8/8/95										7	8	7	8	7	7	6	7	6	6	6	6
8/10/95	4	5	6	4	4	6	6	6	7												
8/21/95										2	7										
8/22/95												7	8	5	6	5	5	5	6	5	6
9/5/95										6	6										
9/7/95												4	6	3	5	4	5	--	--	4	5
9/8/95	3	5	8	3	5	7	3	4	5												
9/19/95										4	6	4	7	4	4	4	4	5	5	4	4

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-4. Vertical light absorption coefficients at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
6/7/94	1.045[b]	0.984							
6/8/94					1.334	0.978	0.940	1.102	1.058
6/9/94			1.093						
6/21/94					1.124	1.002	1.038	1.128	1.007
7/6/94	0.979	1.079	0.914						
7/7/94					1.178	1.036	1.016	1.039	0.986
7/20/94					1.227	0.980	0.982	1.351	1.087
8/2/94	0.666	0.679	0.678						
8/4/94					1.042	0.732	0.774	0.899	0.836
8/18/94					0.814	0.732	0.744	1.044	0.717
9/14/94					0.883	0.834	0.848	0.873	0.791
9/15/94	0.652	0.627	0.844						
10/4/94	0.984	0.845	1.014						
10/6/94					1.110	1.050	1.050	1.194	0.999
11/15/94	0.841	0.813	0.854						
11/17/94					1.009	0.953	0.902	0.976	0.890
12/13/94					0.941	0.869	0.878	0.947	0.845
12/14/94	0.822	0.854	0.741						
1/10/95	0.891	0.832	0.777						
1/25/95					2.986	2.626	2.810	3.070	2.734
2/21/95	5.707	1.765	1.569						
2/22/95					2.843	1.438	1.495	2.951	1.535
3/13/95	1.440	1.283	1.213						
3/15/95					2.075	1.517	1.515	2.019	1.535
4/4/95				1.856	1.825	1.846	1.681	1.938	1.741
4/18/95				1.826					
4/20/95					1.902	1.606	1.594	1.868	1.513
4/21/95	1.785	1.470	1.602						
5/1/95				1.764					
5/2/95									1.749
5/3/95					2.961	1.718	1.684	2.994	
5/15/95	2.004	1.658	1.943						
5/17/95				2.290	2.773	2.239	2.010	2.724	1.842
5/30/95				1.257					
5/31/95					1.653	1.474	1.411	1.656	1.345
6/12/95	1.490	1.180	1.348						
6/14/95					2.125	1.717	1.494	1.970	1.651
6/15/95				1.846					
6/28/95				1.276					
6/29/95					1.666	1.391	1.304	1.665	1.318
7/10/95	1.161	1.195	1.450						
7/12/95					1.894	1.358	1.435	1.638	1.535
7/13/95				1.990					

Appendix Table A-4. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
7/24/95				1.571					
7/25/95					1.749	1.523	1.511	1.674	1.461
8/8/95				1.735	1.635	1.396	1.483	1.501	1.424
8/10/95	1.429	1.220	1.575						
8/21/95				1.058					
8/22/95					1.595	1.279	1.255	1.299	1.392
9/5/95				1.393					
9/7/95					1.347	1.286	1.377	1.368	1.280
9/8/95	1.093	1.033	1.039						
9/19/95				1.319	1.470	1.358	1.471	1.387	1.572

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Absorption coefficients per meter expressed as: $k = \frac{\ln I_0 - \ln I_z}{z}$ where I_0 = light intensity at zero depth, I_z = light intensity at depth z, and z = depth of reading.

Appendix Table A-5. Light transmittance at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
6/7/94	35.60[b]	38.26							
6/8/94					26.51	38.56	39.48	33.64	35.06
6/9/94			34.39						
6/21/94					32.71	37.19	35.92	32.79	36.85
7/6/94	38.30	34.79	40.28						
7/7/94					33.08	35.61	36.31	35.50	37.39
7/20/94					29.37	37.94	37.67	26.33	34.18
8/2/94	51.42	51.40	50.89						
8/4/94					35.31	48.16	46.24	40.70	43.46
8/18/94					44.50	48.10	47.58	37.92	48.87
9/14/94					41.39	43.71	43.14	41.93	45.53
9/15/94	52.18	53.39	43.01						
10/4/94	38.28	43.26	36.60						
10/6/94					33.26	35.28	35.48	30.61	37.21
11/15/94	44.44	45.71	43.10						
11/17/94					37.97	40.21	41.71	38.49	42.45
12/13/94					40.51	43.39	43.08	40.24	44.51
12/14/94	46.10	44.17	49.46						
1/10/95	43.18	45.32	46.79						
1/25/95					6.86	8.94	7.67	6.51	8.31
2/21/95	0.38	17.54	21.42						
2/22/95					6.43	24.97	23.00	5.95	22.14
3/13/95	24.66	28.46	31.26						
3/15/95					12.99	22.61	22.97	13.68	22.30
4/4/95				16.68	16.38	17.10	20.30	15.01	19.44
4/18/95				18.03					
4/20/95					15.88	21.49	21.36	16.23	24.43
4/21/95	18.19	23.72	20.66						
5/1/95				18.95					
5/2/95									18.04
5/3/95					6.47	18.44	19.31	5.77	
5/15/95	14.05	19.74	15.36						
5/17/95				12.00	7.06	11.57	14.44	7.21	16.12
5/30/95				29.22					
5/31/95					19.77	23.87	25.09	19.66	26.66
6/12/95	23.29	31.84	26.96						
6/14/95					12.70	19.21	22.95	15.09	20.38
6/15/95				17.28					
6/28/95				28.96					
6/29/95					19.70	25.58	27.92	19.70	27.90
7/10/95	32.28	31.47	24.52						
7/12/95					15.55	26.46	24.61	20.03	22.98
7/13/95				14.07					

Appendix Table A-5. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater		Upper reservoir			
Date	352	355	390	424	443a	443b	445	446	449
7/24/95				21.46					
7/25/95					17.85	22.52	22.66	19.22	23.68
8/8/95				19.75	20.05	25.27	23.62	23.02	24.52
8/10/95	26.40	36.09	22.13						
8/21/95				35.10					
8/22/95					20.81	28.67	29.34	27.93	25.89
9/5/95				25.05					
9/7/95					27.03	28.68	27.22	27.01	30.15
9/8/95	35.21	38.09	37.26						
9/19/95				27.23	23.68	26.85	23.97	28.52	22.33

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Percentage of radiant energy passing through each meter expressed as: $T = 100\, e^{-k}$ where $k =$ the vertical light absorption coefficient.

Appendix Table A-6. Sediment characteristics of samples collected at upper John Day Reservoir sampling stations during 1994 and 1995.

Date	Station	Depth (m)	Volatile solids (%)	Median grain size (mm)	Silt/Clay (%)
5/12/94	Long Walk Island (downstream site)	1	7.2	0.16	5.8
5/12/94	Big Blalock Island	1	0.8	0.12	12.4
5/12/94	Long Walk Island (upstream site)	1	1.9	0.12	16.6
5/12/94	Paterson Slough	1	0.8	0.04	90.8
5/12/94	Long Walk Island (downstream site)	3	2.9	0.07	46.0
5/12/94	Big Blalock Island	3	1.2	0.17	11.1
5/12/94	Long Walk Island (upstream site)	3	2.7	0.07	43.0
5/12/94	Paterson Slough	3	0.9	0.11	8.5
5/12/94	Long Walk Island (downstream site)	5	2.8	0.08	36.0
5/12/94	Big Blalock Island	5	1.2	0.13	25.7
5/12/94	Long Walk Island (upstream site)	5	1.1	0.16	6.2
5/12/94	Paterson Slough	5	0.9	0.09	16.0
8/3/94	Long Walk Island (downstream site)	1	0.6	0.17	4.3
8/3/94	Sand Island	1	0.5	0.22	7.2
8/3/94	Big Blalock Island	1	0.8	0.15	9.9
8/3/94	Long Walk Island (upstream site)	1	2.6	0.08	38.5
8/3/94	Paterson Slough	1	0.7	0.12	15.4
8/3/94	Long Walk Island (downstream site)	3	2.3	0.07	41.0
8/3/94	Sand Island	3	1.1	0.17	7.8
8/3/94	Big Blalock Island	3	1.0	0.14	24.7
8/3/94	Long Walk Island (upstream site)	3	1.6	0.10	13.4
8/3/94	Paterson Slough	3	1.0	0.15	14.4
8/3/94	Long Walk Island (downstream site)	5	2.9	0.08	32.9
8/3/94	Sand Island	5	1.0	0.17	5.9
8/3/94	Big Blalock Island	5	0.8	0.17	3.8
8/3/94	Long Walk Island (upstream site)	5	1.3	0.15	13.6
8/3/94	Paterson Slough	5	1.1	0.10	10.4
11/18/94	Long Walk Island (downstream site)	1	0.5	0.16	6.1
11/18/94	Sand Island	1	0.7	0.16	10.1
11/18/94	Big Blalock Island	1	0.6	0.17	8.6
11/18/94	Long Walk Island (upstream site)	1	0.9	0.07	45.6
11/18/94	Paterson Slough	1	0.7	0.14	7.5
11/18/94	Long Walk Island (downstream site)	3	1.6	0.12	28.3
11/18/94	Sand Island	3	0.5	0.15	16.3
11/18/94	Big Blalock Island	3	0.7	0.14	21.8
11/18/94	Long Walk Island (upstream site)	3	1.7	0.06	48.5
11/18/94	Paterson Slough	3	0.8	0.10	27.7
11/18/94	Long Walk Island (downstream site)	5	1.8	0.08	27.2
11/18/94	Sand Island	5	0.7	0.19	12.2
11/18/94	Big Blalock Island	5	0.8	0.12	26.1
11/18/94	Long Walk Island (upstream site)	5	1.5	0.18	16.5
11/18/94	Paterson Slough	5	1.2	0.07	41.6
1/24/95	Long Walk Island (downstream site)	1	0.5	0.17	6.4
1/24/95	Sand Island	1	0.9	0.15	7.3
1/24/95	Big Blalock Island	1	0.9	0.10	16.9
1/24/95	Long Walk Island (upstream site)	1	1.4	0.08	28.3

Appendix Table A-6. Continued.

Date	Station	Depth (m)	Volatile solids (%)	Median grain size (mm)	Silt/Clay (%)
1/24/95	Paterson Slough	1	0.9	0.09	17.5
1/24/95	Long Walk Island (downstream site)	3	1.9	0.08	28.1
1/24/95	Sand Island	3	0.8	0.17	5.4
1/24/95	Big Blalock Island	3	0.7	0.17	6.5
1/24/95	Long Walk Island (upstream site)	3	2.1	0.08	29.2
1/24/95	Long Walk Island (downstream site)	5	1.8	0.08	27.5
1/24/95	Sand Island	5	0.4	0.25	2.7
1/24/95	Big Blalock Island	5	0.6	0.21	1.6
1/24/95	Long Walk Island (upstream site)	5	1.6	0.09	27.3
1/25/95	Paterson Slough	3	0.7	0.13	8.0
1/25/95	Paterson Slough	5	0.9	0.11	7.2
4/18/95	Crow Butte	1	2.6	0.05	84.0
4/18/95	Crow Butte	3	1.6	0.06	60.6
4/18/95	Crow Butte	5	3.0	0.05	71.6
4/19/95	Long Walk Island (downstream site)	1	0.7	0.17	8.2
4/19/95	Sand Island	1	0.3	0.23	5.6
4/19/95	Big Blalock Island	1	0.9	0.14	9.3
4/19/95	Long Walk Island (upstream site)	1	1.7	0.09	29.3
4/19/95	Paterson Slough	1	0.6	0.15	4.7
4/19/95	Long Walk Island (downstream site)	3	1.7	0.09	29.2
4/19/95	Sand Island	3	0.5	0.27	4.4
4/19/95	Big Blalock Island	3	0.6	0.19	6.1
4/19/95	Long Walk Island (upstream site)	3	1.8	0.09	28.5
4/19/95	Paterson Slough	3	0.8	0.14	9.7
4/19/95	Long Walk Island (downstream site)	5	2.2	0.08	33.7
4/19/95	Sand Island	5	9.6	0.27	6.3
4/19/95	Big Blalock Island	5	0.7	0.17	10.7
4/19/95	Long Walk Island (upstream site)	5	1.7	0.14	26.1
4/19/95	Paterson Slough	5	0.8	0.12	7.0
7/11/95	Long Walk Island (downstream site)	1	0.8	0.16	7.1
7/11/95	Sand Island	1	0.7	0.19	6.7
7/11/95	Big Blalock Island	1	0.9	0.34	7.0
7/11/95	Long Walk Island (upstream site)	1	1.5	0.13	21.8
7/11/95	Paterson Slough	1	0.5	0.16	24.2
7/11/95	Long Walk Island (downstream site)	3	2.1	0.08	35.1
7/11/95	Sand Island	3	0.5	0.26	2.9
7/11/95	Big Blalock Island	3	1.0	0.16	32.0
7/11/95	Long Walk Island (upstream site)	3	2.6	0.09	28.9
7/11/95	Paterson Slough	3	0.7	0.15	6.0
7/11/95	Long Walk Island (downstream site)	5	2.2	0.08	32.2
7/11/95	Sand Island	5	0.4	0.27	1.9
7/11/95	Big Blalock Island	5	0.4	0.24	13.3
7/11/95	Long Walk Island (upstream site)	5	0.9	0.17	7.7
7/11/95	Paterson Slough	5	0.9	0.12	36.2
7/13/95	Crow Butte	1	1.8	0.08	35.3
7/13/95	Crow Butte	3	1.6	0.14	20.1
7/13/95	Crow Butte	5	3.0	0.06	55.3

Appendix Table A-7. Dissolved oxygen levels (surface to bottom averages; mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/11/94	12.4	12.8	13.0						
4/13/94					12.5	13.1	12.9	12.3	13.0
4/26/94					10.1	10.4	10.5	10.1	10.5
5/10/94	10.9	11.5	9.1						
5/11/94					8.1	8.2	8.1	8.0	8.2
5/24/94					12.1	12.2	12.3	11.8	12.3
6/7/94	10.4	10.8							
6/8/94					11.0	11.3	11.4	10.7	11.2
6/9/94			10.6						
6/21/94					10.8	10.8	10.8	10.6	11.2
7/6/94	9.0	9.1	9.5						
7/7/94					9.7	9.7	9.8	9.5	10.2
7/20/94					9.2	9.4	9.4	9.2	9.5
8/2/94	8.0	8.1	8.4						
8/4/94					8.5	8.9	8.8	8.8	8.8
8/18/94					8.7	8.7	8.6	8.5	9.0
9/14/94					9.3	9.3	9.3	9.6	9.8
9/15/94	8.1	8.3	8.7						
10/4/94	8.3	8.4	8.9						
10/6/94					9.5	9.4	9.7	10.0	10.2
11/15/94	9.5	9.8	9.9						
11/17/94					9.3	9.6	9.7	9.7	10.5
12/13/94					10.7	10.5	10.3	10.8	10.3
12/14/94	11.4	11.6	11.7						
1/10/95	11.3	11.7	11.6						
1/25/95					11.4	11.5	11.2	11.4	11.1
2/21/95	11.6	12.1	12.1						
2/22/95					11.8	12.1	11.9	11.8	11.8
3/13/95	11.9	12.4	12.4						
3/15/95				12.1	11.6	11.8	12.1	11.7	12.3
4/4/95				13.1	11.7	11.9	11.8	11.9	11.9
4/18/95				12.1					
4/20/95					11.8	11.9	11.9	11.7	12.1
4/21/95	11.4	11.6	11.7						
5/1/95				11.8					
5/2/95									12.3
5/3/95					12.1	12.5	12.7	12.2	
5/15/95	10.5	11.1	11.5						
5/17/95				10.1	10.8	11.3	11.4	10.9	11.4
5/30/95				12.1					
5/31/95					10.9	11.2	11.3	11.1	11.6
6/12/95	10.3	10.6	10.7						
6/14/95					10.4	11.1	11.2	10.6	11.3

Appendix Table A-7. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
6/15/95				10.1					
6/28/95				10.3					
6/29/95					10.2	10.7	10.9	10.3	11.1
7/10/95	9.0	9.1	9.4						
7/12/95					9.2	9.2	9.1	9.2	9.2
7/13/95				9.3					
7/24/95				9.0					
7/25/95					8.8	8.7	8.7	8.7	8.8
8/8/95				9.5	9.0	9.0	8.9	9.0	8.9
8/10/95	8.6	8.6	8.9						
8/21/95				10.3					
8/22/95					9.6	9.5	9.5	9.7	9.7
9/5/95				8.4					
9/7/95					9.4	9.3	9.1	9.3	9.6
9/8/95	8.6	8.8	9.2						
9/19/95				8.3	9.8	9.5	9.3	9.5	9.6

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

Appendix Table A-8. pH values (antilog means) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/11/94	8.4	8.5	8.5						
4/13/94					8.3	8.3	8.3	8.3	8.4
4/26/94					8.5	8.6	8.6	8.5	8.7
5/10/94	8.6	8.6	8.6						
5/11/94					8.4	8.5	8.6	8.4	8.8
5/24/94					8.5	8.5	8.4	8.3	8.6
6/7/94	8.3	8.4							
6/8/94					8.3	8.1	8.2	8.2	8.3
6/9/94			8.2						
6/21/94					8.2	8.2	8.2	8.2	8.4
7/6/94	8.2	8.2	8.3						
7/7/94					8.2	8.1	8.1	8.1	8.4
7/20/94					8.0	8.1	8.1	8.1	8.2
8/2/94	8.0	8.1	8.1						
8/4/94					8.1	8.2	8.2	8.2	8.3
8/18/94					8.1	8.1	8.0	8.0	8.2
9/14/94					8.4	8.4	8.5	8.5	8.5
9/15/94	8.1	8.1	8.2						
10/4/94	8.3	8.3	8.4						
10/6/94					8.5	8.5	8.6	8.6	8.7
11/15/94	8.1	8.1	8.1						
11/17/94					8.0	8.1	8.1	8.1	8.1
12/13/94					8.1	8.2	8.2	8.1	8.2
12/14/94	8.1	8.1	8.0						
1/10/95	8.0	8.0	8.1						
1/25/95					8.0	8.0	8.0	8.0	8.0
2/21/95	8.0	8.0	8.0						
2/22/95					8.0	8.0	8.0	8.0	8.0
3/13/95	7.9	7.9	8.0						
3/15/95				8.3	7.9	7.9	7.9	7.9	8.0
4/4/95				8.8	7.8	7.8	7.8	7.8	7.9
4/18/95				8.6					
4/20/95					8.0	8.0	8.0	8.0	8.1
4/21/95	7.9	8.0	8.0						
5/1/95				8.6					
5/2/95									8.1
5/3/95					8.0	8.1	8.2	8.1	
5/15/95	8.0	8.1	8.1						
5/17/95				8.8	7.9	8.0	8.1	7.9	8.2
5/30/95				8.7					
5/31/95					7.8	7.9	7.9	7.8	8.0
6/12/95	7.8	7.9	7.8						
6/14/95					7.7	7.8	7.8	7.7	7.8

Appendix Table A-8. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
6/15/95				8.0					
6/28/95				8.1					
6/29/95					7.8	7.9	7.9	7.8	8.0
7/10/95	7.8	7.7	7.9						
7/12/95					7.8	7.8	7.7	7.8	7.7
7/13/95				8.6					
7/24/95				8.6					
7/25/95					7.8	7.7	7.7	7.7	7.7
8/8/95				8.5	7.9	7.9	7.8	7.8	7.8
8/10/95	7.9	7.9	8.0						
8/21/95				8.8					
8/22/95					8.1	8.0	8.0	8.2	8.2
9/5/95				8.7					
9/7/95					8.2	8.1	8.0	8.1	8.2
9/8/95	8.1	8.2	8.3						
9/19/95				8.4	8.4	8.3	8.3	8.3	8.4

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

Appendix Table A-9. Specific conductance (temperature compensated to 25°C; surface to bottom averages; µS/cm) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Sampling stations[a]								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
4/11/94	171	179	178						
4/13/94					181	181	177	182	181
4/26/94					179	178	176	179	177
5/10/94	142	139	134						
5/11/94					140	140	141	141	142
5/24/94					144	142	142	144	143
6/7/94	129	128							
6/8/94					130	128	129	129	128
6/9/94			128						
6/21/94					168	136	137	139	137
7/6/94	136	136	135						
7/7/94					138	137	137	139	138
7/20/94					133	134	134	134	134
8/2/94	144	144	144						
8/4/94					142	141	141	142	140
8/18/94					137	136	136	137	136
9/14/94					144	143	142	143	142
9/15/94	144	144	143						
10/4/94	146	147	145						
10/6/94					149	148	148	150	148
11/15/94	172	171	169						
11/17/94					177	175	173	173	168
12/13/94					171	166	164	170	165
12/14/94	178	179	176						
1/10/95	184	178	175						
1/25/95					196	188	188	195	187
2/21/95	170	191	183						
2/22/95					171	169	167	165	167
3/13/95	175	181	187						
3/15/95				189	174	162	158	176	155
4/4/95				170	154	159	161	163	160
4/18/95				162					
4/20/95					173	172	172	173	172
4/21/95	150	160	168						
5/1/95				159					
5/2/95									149
5/3/95					143	147	147	143	
5/15/95	126	133	128						
5/17/95				147	121	103	123	119	124
5/30/95				126					
5/31/95					119	118	119	118	119
6/12/95	120	117	112						

Appendix Table A-9. Continued.

	Sampling stations								
	Lower/Mid reservoir			Backwater	Upper reservoir				
Date	352	355	390	424	443a	443b	445	446	449
6/14/95					104	106	106	106	107
6/15/95				112					
6/28/95				121					
6/29/95					120	118	119	120	119
7/10/95	125	125	127						
7/12/95					121	121	121	123	122
7/13/95				107					
7/24/95				132					
7/25/95					129	129	129	130	129
8/8/95				137	135	135	135	136	134
8/10/95	138	135	135						
8/21/95				138					
8/22/95					140	139	140	141	140
9/5/95				144					
9/7/95					141	141	140	143	140
9/8/95	142	141	142						
9/19/95				148	142	140	140	142	140

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

Appendix Table A-10. Alkalinity measurements (mg/L $CaCO_3$) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	71	75	75	76	76	76	75	74	73												
4/13/94												75	75	75	75	74	74	73	75	73	73
4/26/94												72	72	68	71	71	72	72	71	72	70
5/10/94	59	58	59	57	56	57	57	57	55												
5/11/94												57	58	62	58	59	--[c]	57	61	58	59
5/24/94												56	56	56	57	55	55	56	56	56	57
6/7/94	56	56	53	54	54	53															
6/8/94												54	55	53	53	53	52	54	53	53	53
6/9/94							52	52	53												
6/21/94												55	55	55	56	55	55	56	56	56	55
7/6/94	55	56	56	56	56	56	56	55	56												
7/7/94												56	56	56	55	56	56	56	56	56	56
7/20/94												55	55	55	55	55	55	54	54	54	55
8/2/94	62	63	62	62	61	61	61	60	60												
8/4/94												60	59	59	58	59	58	59	59	58	58
8/18/94												56	57	56	56	56	56	56	56	56	56
9/14/94												54	54	53	53	53	53	54	54	53	53
9/15/94	53	53	53	53	53	54	54	53	54												
10/4/94	60	60	60	61	61	60	60	60	61												
10/6/94												61	61	61	61	60	61	61	61	60	61
11/15/94	64	65	69	65	65	66	63	64	63												
11/17/94												67	66	66	66	65	65	66	66	65	64
12/13/94												69	68	68	67	67	67	67	67	67	67
12/14/94	70	71	70	70	70	70	69	69	69												
1/10/95	74	74	74	72	71	72	71	72	71												
1/25/95												76	80	70	75	75	75	76	76	75	76
2/21/95	76	75	76	77	76	75	77	77	76												
2/22/95												67	68	69	68	68	68	66	65	68	70
3/13/95	76	77	77	69	70	71	76	74	75												
3/15/95										73	72	67	67	66	66	65	65	67	67	66	65
4/4/95										71	72	66	66	65	65	65	66	66	66	65	66
4/18/95										70	70										

Appendix Table 10. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												71	70	70	71	71	70	71	71	71	71
4/21/95	66	67	66	66	66	67	69	69	69												
5/1/95										68	68										
5/2/95																				64	64
5/3/95												61	62	63	64	64	64	62	62		
5/15/95	57	56	56	55	55	55	53	54	54												
5/17/95										66	66	51	52	53	53	53	53	51	50	52	53
5/30/95										54	53										
5/31/95												50	49	49	50	50	50	49	49	50	50
6/12/95	51	48	48	48	48	48	46	46	46												
6/14/95												43	43	45	45	45	45	45	45	46	45
6/15/95										49	46										
6/28/95										53	52										
6/29/95												52	52	52	52	53	53	52	52	53	53
7/10/95	55	53	53	53	53	53	54	54	54												
7/12/95												52	52	52	52	52	52	52	52	52	52
7/13/95										55	55										
7/24/95										53	53										
7/25/95												54	53	53	53	53	53	54	54	53	53
8/8/95										60	61	60	60	60	59	59	59	60	60	59	59
8/10/95	62	60	60	60	60	60	60	60	60												
8/21/95										60	61										
8/22/95												60	60	60	60	60	60	61	61	61	60
9/5/95										63	63										
9/7/95												62	61	61	61	61	61	62	62	61	61
9/8/95	62	63	62	62	62	62	63	63	62												
9/19/95										64	65	62	62	61	61	61	61	62	62	62	62

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-11. Silica concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	16.5	8.8	8.2	9.1	8.7	8.0	8.6	9.6	9.5												
4/13/94												10.2	10.3	10.3	10.4	10.2	10.1	10.8	11.1	10.8	10.3
4/26/94												10.9	11.7	11.4	11.5	11.2	11.5	11.4	11.9	10.8	10.6
5/10/94	18.0	9.2	9.3	8.5	8.7	8.7	7.7	8.2	8.5												
5/11/94												9.4	9.7	8.6	8.6	8.6	--[c]	9.4	9.5	8.3	8.6
5/24/94												8.6	8.8	7.9	8.1	8.1	7.9	9.0	9.0	7.4	7.9
6/7/94	11.2	11.6	7.7	2.1	2.5	7.3															
6/8/94												6.2	5.2	7.1	7.8	5.8	7.8	5.4	8.3	7.9	7.9
6/9/94							7.5	7.5	7.5												
6/21/94												8.1	8.2	8.1	8.0	8.3	8.2	8.3	8.3	7.8	8.0
7/6/94	6.7	6.9	7.2	6.8	6.9	6.9	6.3	6.4	6.5												
7/7/94												7.0	7.1	7.2	7.1	7.1	7.2	7.2	7.2	6.7	6.7
7/20/94												7.3	7.3	7.3	7.3	7.3	7.0	7.3	7.0	7.0	7.0
8/2/94	7.5	7.5	8.0	7.5	7.5	7.8	7.3	7.3	7.5												
8/4/94												7.5	7.5	7.3	7.3	7.0	7.0	7.3	7.3	6.8	7.0
8/18/94												6.0	6.0	5.8	5.8	5.8	5.8	5.8	6.0	5.5	5.8
9/14/94												5.3	5.5	5.3	5.3	5.3	5.3	5.3	5.3	5.3	5.3
9/15/94	5.0	5.3	5.3	5.3	5.3	5.3	5.0	5.3	5.3												
10/4/94	5.3	5.3	5.3	5.3	5.3	5.3	4.6	4.6	4.6												
10/6/94												5.0	5.0	4.8	4.8	4.8	4.8	4.8	4.8	4.8	4.8
11/15/94	7.4	7.4	8.7	7.6	7.6	7.6	7.8	7.8	7.8												
11/17/94												8.2	8.2	8.2	8.2	8.0	8.0	8.5	8.2	7.6	7.8
12/13/94												8.2	8.2	7.5	7.5	7.5	7.5	8.4	8.2	7.5	7.5
12/14/94	8.4	8.2	8.8	8.4	8.6	8.6	8.2	7.9	8.4												
1/10/95	10.0	10.7	10.7	9.3	9.8	9.3	8.9	8.9	8.9												
1/25/95												13.0	12.6	10.7	10.7	10.7	10.5	12.6	12.8	10.5	10.5
2/21/95	19.0	18.1	17.6	27.5	26.5	26.5	13.0	13.0	13.3												
2/22/95												15.6	15.8	12.1	11.9	11.9	11.9	16.7	16.5	23.5	24.0
3/13/95	22.1	14.8	13.3	13.0	13.5	13.5	13.9	13.7	13.9												
3/15/95										10.5	10.8	17.1	17.1	13.9	13.9	13.0	13.0	17.4	17.4	11.9	12.1
4/4/95										11.2	11.2	16.7	16.7	16.7	16.8	16.7	16.5	17.0	17.0	16.5	16.5
4/18/95										12.8	11.8										

Appendix Table A-11. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												17.8	17.8	17.0	17.0	17.2	17.2	17.8	17.8	17.2	15.4
4/21/95	23.0	21.7	17.5	16.7	16.5	16.7	17.2	17.0	17.2												
5/1/95										10.8	11.3										
5/2/95																				13.5	12.9
5/3/95												15.1	15.4	12.4	12.4	12.7	12.1	15.1	15.1		
5/15/95	18.6	14.6	14.6	14.3	14.6	14.8	14.0	14.3	15.1												
5/17/95										9.7	9.7	15.9	16.4	12.9	12.9	9.7	10.0	16.2	16.2	11.9	11.9
5/30/95										11.5	12.4										
5/31/95												13.3	13.3	13.0	13.0	12.7	12.4	13.3	13.3	12.4	12.4
6/12/95	14.2	12.4	12.4	11.5	11.5	11.8	11.2	11.2	11.2												
6/14/95												11.2	11.2	10.3	10.3	10.3	10.3	10.9	10.9	10.0	10.0
6/15/95										8.8	10.0										
6/28/95										8.7	9.0										
6/29/95												10.2	10.2	9.6	9.6	9.0	9.0	10.2	10.2	8.7	8.7
7/10/95	10.2	10.2	10.2	11.1	10.5	10.5	10.2	9.9	10.2												
7/12/95												9.3	9.3	10.2	9.9	9.6	9.9	9.6	9.6	9.9	9.9
7/13/95										7.8	8.1										
7/24/95										6.9	6.9										
7/25/95												9.0	9.0	9.0	9.0	9.0	9.0	9.0	9.0	8.4	8.7
8/8/95										6.6	6.6	8.6	8.6	8.6	8.6	8.6	8.6	8.6	8.6	8.6	8.6
8/10/95	8.6	8.3	8.3	8.1	8.3	8.3	8.1	8.3	8.3												
8/21/95										5.7	6.0										
8/22/95												7.5	7.5	7.8	7.8	7.8	7.8	7.5	7.5	7.2	7.5
9/5/95										4.9	4.9										
9/7/95												6.6	6.9	6.9	6.9	6.9	6.9	6.9	7.2	6.3	6.9
9/8/95	6.0	6.3	6.3	5.7	6.6	6.6	5.7	5.7	6.0												
9/19/95										5.1	5.4	6.0	6.3	6.0	6.3	6.3	6.0	6.3	6.6	6.3	6.0

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-12. Sulfate concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	9.9	13.7	13.9	13.6	15.5	14.1	12.9	13.6	13.5												
4/13/94												14.0	14.2	14.7	14.8	14.3	13.3	14.4	17.0	14.2	15.5
4/26/94												13.5	15.6	15.5	13.6	13.7	15.6	13.8	14.3	15.8	15.7
5/10/94	7.8	10.5	10.2	10.2	9.9	--[c]	9.9	10.2	10.4												
5/11/94												10.4	10.1	10.6	12.0	10.6	--	10.2	11.6	10.5	10.7
5/24/94												10.7	10.6	10.6	10.1	10.4	10.6	10.2	10.4	10.7	10.8
6/7/94	7.8	7.5	8.6	8.9	9.0	8.5															
6/8/94												8.4	8.5	8.6	8.5	8.4	9.2	8.4	8.3	8.3	8.3
6/9/94							8.3	8.4	8.5												
6/21/94												--	--	--	--	--	--	--	--	--	--
7/6/94	8.8	8.8	8.9	8.9	8.9	8.9	8.7	8.8	8.8												
7/7/94												9.1	9.2	9.2	9.1	--	--	--	--	--	--
7/20/94												8.6	8.5	8.8	8.6	8.7	8.8	8.6	8.6	8.7	8.7
8/2/94	9.9	10.0	10.1	9.8	10.1	10.0	10.1	10.1	10.1												
8/4/94												10.1	9.9	9.9	9.8	9.8	10.1	9.9	10.0	9.7	9.7
8/18/94												9.1	9.4	9.1	9.0	9.1	9.1	9.2	9.2	9.1	9.0
9/14/94												9.8	10.0	10.0	9.9	9.8	9.9	9.9	9.9	10.1	9.8
9/15/94	9.8	9.7	9.6	9.7	9.9	9.7	9.4	9.9	9.8												
10/4/94	12.1	14.5	11.7	10.5	10.7	10.7	10.5	10.8	11.8												
10/6/94												11.1	11.1	11.3	11.1	12.1	11.2	11.4	11.3	11.7	11.6
11/15/94	13.5	13.5	12.9	13.6	13.4	12.8	13.4	13.3	13.5												
11/17/94												13.8	14.4	14.3	14.4	13.9	14.0	13.9	13.9	13.0	12.9
12/13/94												12.7	13.0	12.5	12.3	12.2	12.3	12.8	12.8	12.5	12.4
12/14/94	14.2	14.2	14.0	14.3	14.3	14.4	13.8	13.9	14.1												
1/10/95	14.0	13.5	13.6	14.1	14.0	13.7	13.0	13.3	13.2												
1/25/95												15.1	15.3	14.0	14.2	14.4	14.3	15.1	15.0	14.0	13.7
2/21/95	11.5	12.0	12.3	15.0	15.1	14.9	15.2	15.4	15.3												
2/22/95												12.7	12.0	12.3	12.3	12.2	11.8	11.8	11.8	12.0	12.0
3/13/95	11.5	16.7	18.4	17.8	19.0	18.2	19.4	19.2	19.4												
3/15/95										19.6	19.4	17.4	17.4	15.7	15.7	15.1	15.1	17.6	17.8	14.4	14.6
4/4/95										14.9	14.9	14.4	14.4	13.6	13.8	13.6	13.8	14.4	14.4	14.4	14.4
4/18/95										18.3	18.1										

Appendix Table A-12. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												19.0	19.0	18.5	18.1	19.0	19.3	19.0	19.1	19.1	19.0
4/21/95	14.1	14.7	17.2	17.6	17.6	17.8	19.1	19.1	18.8												
5/1/95										14.4	14.4										
5/2/95																				13.1	13.1
5/3/95												12.5	12.5	12.7	12.7	12.5	12.5	12.2	12.2		
5/15/95	13.9	15.4	14.9	14.2	13.7	13.7	15.7	15.4	15.4												
5/17/95										13.4	13.7	10.8	10.5	10.8	10.8	10.5	10.8	10.8	10.5	11.6	12.4
5/30/95										13.4	13.4										
5/31/95												12.9	13.2	12.9	13.2	12.7	12.9	13.2	12.9	13.4	13.4
6/12/95	13.7	13.9	9.0	9.0	9.0	9.2	13.2	13.4	13.4												
6/14/95												12.1	12.1	11.9	11.9	11.9	12.1	11.9	12.1	12.1	12.1
6/15/95										13.2	12.4										
6/28/95										12.7	12.7										
6/29/95												12.7	12.5	12.3	12.3	12.1	12.1	12.5	12.5	11.9	11.9
7/10/95	10.6	11.2	10.9	10.9	11.2	11.2	11.2	11.2	11.2												
7/12/95												10.6	10.6	10.6	10.6	10.3	10.6	10.6	10.9	10.9	10.9
7/13/95										11.2	11.5										
7/24/95										10.6	10.6										
7/25/95												10.3	10.6	10.0	10.3	9.7	9.7	10.3	10.3	10.6	10.9
8/8/95										11.4	11.4	11.6	11.6	11.2	11.6	10.9	10.9	11.6	11.6	11.9	11.9
8/10/95	11.4	11.6	11.6	11.9	12.1	11.6	11.9	11.9	11.9												
8/21/95										10.6	10.6										
8/22/95												10.6	10.3	10.6	10.6	10.6	10.3	10.3	10.6	10.6	10.6
9/5/95										11.2	11.2										
9/7/95												11.0	11.0	10.7	10.7	10.7	10.7	11.0	11.0	10.7	11.0
9/8/95	10.7	10.7	11.0	11.0	11.0	11.0	11.0	11.0	11.0												
9/19/95										13.7	14.1	13.5	13.5	13.0	13.0	13.5	13.2	13.5	13.5	13.2	13.0

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-13. Chloride concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	3.3	3.7	3.7	3.8	3.8	3.8	3.7	3.7	4.3												
4/13/94												4.3	4.3	4.2	4.4	4.2	4.0	4.3	4.9	4.2	4.5
4/26/94												4.6	4.9	5.4	4.7	4.7	5.1	5.3	5.1	4.9	4.7
5/10/94	2.3	2.8	2.7	2.9	2.5	--[c]	2.5	2.5	2.1												
5/11/94												2.6	3.0	2.9	3.4	2.6	--	2.9	3.2	2.8	2.9
5/24/94												2.7	2.7	2.7	2.9	3.6	2.6	2.8	2.7	2.8	2.6
6/7/94	1.3	1.2	1.4	1.5	1.3	1.3															
6/8/94												3.2	1.3	1.2	1.2	1.2	1.3	1.3	1.2	1.2	1.3
6/9/94							2.1	1.9	1.2												
6/21/94												--	--	--	--	--	--	--	--	--	--
7/6/94	1.3	1.3	1.3	1.3	1.3	1.3	1.2	1.2	1.2												
7/7/94												1.6	1.6	1.6	1.6	--	--	--	--	--	--
7/20/94												2.0	2.0	2.1	2.1	2.1	2.1	2.1	2.0	2.1	2.1
8/2/94	2.9	2.9	2.6	2.9	2.8	3.2	2.7	2.7	3.0												
8/4/94												2.7	3.0	2.6	2.5	2.9	2.6	3.0	2.7	2.4	2.4
8/18/94												1.6	1.6	1.6	1.6	1.7	1.6	1.6	1.7	1.6	1.9
9/14/94												1.9	2.0	2.0	2.0	1.9	2.0	2.0	2.0	1.9	2.0
9/15/94	1.9	2.3	2.0	1.9	1.9	1.9	2.5	1.9	2.0												
10/4/94	3.6	4.5	2.1	2.3	2.4	2.0	1.9	1.9	2.3												
10/6/94												2.1	2.5	2.1	2.8	2.1	2.3	2.2	2.7	2.3	3.1
11/15/94	3.5	3.5	3.2	3.5	3.5	3.5	3.5	3.5	3.5												
11/17/94												3.7	4.0	4.0	4.0	3.7	3.7	3.7	3.7	3.2	3.2
12/13/94												3.1	3.1	2.8	3.0	2.9	3.2	3.0	3.4	3.0	3.1
12/14/94	3.9	3.8	3.8	4.0	3.9	3.9	3.5	3.7	3.8												
1/10/95	3.9	3.8	3.7	3.9	3.9	3.6	3.4	3.3	3.3												
1/25/95												4.5	4.5	4.1	4.1	4.1	4.1	4.6	4.5	4.0	3.8
2/21/95	3.6	3.7	3.6	4.8	4.5	4.4	4.2	4.4	4.7												
2/22/95												3.4	3.3	2.9	3.2	3.0	2.9	3.6	3.3	2.8	3.0
3/13/95	2.4	3.7	4.0	4.0	4.2	4.2	4.7	4.5	4.7												
3/15/95										5.0	5.0	4.5	4.5	3.2	3.2	2.9	2.9	4.5	4.7	2.4	2.4
4/4/95										4.0	4.0	3.5	3.5	3.5	3.5	3.5	3.5	3.7	3.7	3.5	3.5
4/18/95										3.7	3.7										

Appendix Table A-13. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5
4/21/95	2.4	2.7	3.5	3.5	3.5	3.5	4.0	4.0	4.0												
5/1/95										3.7	3.5										
5/2/95																				2.9	3.2
5/3/95												2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7		
5/15/95	2.2	2.7	2.7	2.4	2.4	2.7	2.2	2.2	2.2												
5/17/95										3.2	3.2	2.4	2.4	1.9	1.9	2.4	1.9	2.4	2.4	1.7	1.7
5/30/95										2.7	2.7										
5/31/95												2.7	2.7	2.4	2.4	2.4	2.4	2.7	2.7	2.4	2.4
6/12/95	3.2	3.0	3.0	3.0	3.0	2.7	2.7	2.7	2.7												
6/14/95												2.4	2.5	2.0	2.0	2.0	2.0	2.2	2.2	2.0	2.0
6/15/95										2.7	2.0										
6/28/95										1.0	0.7										
6/29/95												0.7	1.0	0.4	0.4	0.1	0.1	0.7	0.7	0.1	0.1
7/10/95	1.0	2.3	2.3	2.3	1.0	1.0	1.3	1.3	1.3												
7/12/95												1.0	1.0	1.0	1.0	2.3	1.0	1.0	5.8	1.0	1.0
7/13/95										1.3	1.3										
7/24/95										2.6	2.6										
7/25/95												1.8	1.8	1.8	2.1	2.1	2.1	2.1	2.1	2.1	2.1
8/8/95										2.6	2.6	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1
8/10/95	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1	2.1												
8/21/95										2.1	2.4										
8/22/95												2.1	2.4	2.1	2.1	2.4	2.4	2.1	2.1	2.1	2.1
9/5/95										2.9	2.9										
9/7/95												2.4	2.4	2.4	2.4	2.4	2.4	2.4	2.4	2.1	2.4
9/8/95	2.4	2.4	2.4	2.4	2.4	2.4	2.4	2.1	2.1												
9/19/95										2.8	3.8	2.5	2.5	2.5	2.5	2.3	2.3	2.8	2.5	2.3	2.3

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-14. Calcium concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	19.0	22.0	22.2	21.7	22.2	22.1	21.8	21.5	21.9												
4/13/94												21.6	21.5	21.6	21.8	21.7	21.8	22.0	22.0	21.9	22.0
4/26/94												19.2	19.5	19.4	19.5	19.5	19.5	19.6	19.4	19.6	19.6
5/10/94	16.2	17.0	17.2	16.9	16.8	16.7	16.8	16.8	17.0												
5/11/94												16.5	16.8	17.4	17.2	16.9	--[c]	16.8	16.9	17.3	17.3
5/24/94												16.3	16.1	16.2	16.2	16.2	16.3	16.4	16.4	16.4	16.4
6/7/94	14.3	14.1	14.3	15.5	15.6	14.5															
6/8/94												15.2	15.1	15.0	15.2	15.3	15.6	15.3	15.1	15.1	15.0
6/9/94							14.7	14.8	15.1												
6/21/94												15.7	15.8	15.6	15.5	15.5	15.7	15.4	15.5	15.6	15.2
7/6/94	15.6	16.1	15.8	15.6	15.7	15.7	15.7	16.0	15.9												
7/7/94												15.5	15.8	15.6	15.8	15.8	15.7	15.6	15.9	15.7	15.8
7/20/94												16.1	16.1	16.0	15.8	16.1	16.0	16.1	16.1	15.8	16.0
8/2/94	16.8	16.7	17.0	17.1	18.1	17.2	17.0	17.1	17.1												
8/4/94												16.8	16.7	16.7	16.7	16.7	16.7	16.7	16.6	16.7	16.7
8/18/94												17.4	17.4	18.2	17.4	17.2	17.3	17.3	17.4	17.3	17.3
9/14/94												17.3	17.3	17.3	17.2	17.2	17.0	17.2	17.1	17.2	17.2
9/15/94	17.4	17.4	17.4	17.5	17.5	17.4	17.1	16.4	17.3												
10/4/94	17.3	17.1	17.3	17.2	16.9	17.2	17.3	17.4	17.5												
10/6/94												17.5	17.4	17.4	17.0	17.4	17.2	17.5	17.6	18.5	18.4
11/15/94	19.0	19.0	20.0	20.0	19.0	20.0	19.0	19.0	19.0												
11/17/94												20.0	20.0	19.0	19.0	19.0	19.0	19.0	19.0	19.0	19.0
12/13/94												20.0	18.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0
12/14/94	20.0	21.0	21.0	21.0	20.0	21.0	21.0	20.0	21.0												
1/10/95	21.0	21.0	21.0	21.0	21.0	21.0	21.0	21.0	21.0												
1/25/95												23.0	22.0	22.0	22.0	22.0	22.0	22.0	21.0	22.0	22.0
2/21/95	21.0	21.0	20.0	23.0	23.0	23.0	23.0	23.0	23.0												
2/22/95												20.0	20.0	21.0	21.0	20.0	20.0	20.0	20.0	21.0	22.0
3/13/95	19.0	19.0	19.0	21.0	21.0	21.0	22.0	22.0	22.0												
3/15/95										22.0	22.0	20.0	19.0	20.0	20.0	20.0	20.0	19.0	19.0	20.0	20.0
4/4/95										21.0	21.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0	20.0
4/18/95										20.0	20.0										

Appendix Table A-14. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												20.0	20.0	21.0	21.0	21.0	21.0	20.0	20.0	21.0	21.0
4/21/95	19.0	19.0	19.0	21.0	21.0	21.0	20.0	20.0	20.0												
5/1/95										19.1	19.1										
5/2/95																				18.4	18.5
5/3/95												17.3	17.4	18.4	18.4	18.5	18.4	18.0	18.0		
5/15/95	13.6	22.1	16.3	14.8	14.9	14.7	13.9	13.7	13.9												
5/17/95										16.7	16.4	13.6	13.1	13.8	13.8	12.3	11.1	13.2	13.1	14.8	15.5
5/30/95										11.5	11.5										
5/31/95												10.1	10.1	10.0	10.0	10.6	10.4	10.1	10.0	10.6	10.5
6/12/95	10.8	10.3	10.3	10.6	10.3	10.6	10.6	10.5	10.3												
6/14/95												9.4	9.4	10.0	9.8	10.0	10.0	10.3	10.0	10.3	10.3
6/15/95										10.7	10.2										
6/28/95										13.8	14.1										
6/29/95												13.5	13.6	14.2	14.0	14.3	14.1	13.9	13.8	14.5	14.4
7/10/95	13.8	14.0	13.6	13.7	13.7	13.5	14.1	14.0	14.1												
7/12/95												13.7	13.9	14.5	14.4	15.1	14.7	13.9	13.7	15.4	14.4
7/13/95										14.8	14.4										
7/24/95										13.9	13.7										
7/25/95												14.0	14.2	14.1	14.1	14.3	14.1	13.9	14.1	14.0	14.2
8/8/95										16.8	16.8	16.9	16.6	16.4	16.8	17.0	16.8	16.5	16.7	16.7	16.6
8/10/95	16.5	16.3	16.2	16.3	16.1	16.1	16.8	16.6	16.7												
8/21/95										16.7	16.5										
8/22/95												16.8	16.8	16.6	16.7	16.8	16.6	16.5	16.6	16.7	16.7
9/5/95										17.4	17.6										
9/7/95												17.4	17.2	17.0	17.2	17.1	16.9	17.4	17.3	17.2	17.2
9/8/95	17.2	17.4	17.1	17.0	17.1	17.0	17.3	17.4	17.4												
9/19/95										17.5	17.9	17.2	17.1	17.2	17.1	17.3	16.9	17.4	17.4	18.0	17.0

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-15. Magnesium concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	6.1	6.7	6.6	6.4	6.2	6.1	6.6	6.5	6.5												
4/13/94												6.0	6.0	6.2	6.1	6.5	6.4	6.8	6.6	6.6	6.5
4/26/94												6.3	6.3	6.1	6.3	6.6	6.6	6.6	6.2	6.3	6.3
5/10/94	5.7	5.0	5.0	4.8	4.8	4.8	4.7	4.9	4.8												
5/11/94												5.0	5.0	5.0	5.0	4.9	--[c]	5.0	5.0	5.0	5.0
5/24/94												5.0	4.9	4.9	4.9	4.9	4.9	5.0	5.0	5.0	4.9
6/7/94	4.3	4.8	4.7	4.2	4.2	4.5															
6/8/94												4.4	4.5	4.3	4.3	4.3	4.4	4.4	4.4	4.3	4.4
6/9/94							4.3	4.3	4.4												
6/21/94												4.6	4.6	4.5	4.5	4.5	4.5	4.5	4.5	4.5	4.5
7/6/94	4.4	4.6	4.5	4.5	4.5	4.4	4.4	4.5	4.5												
7/7/94												4.5	4.7	4.6	4.6	4.7	4.3	4.6	4.6	4.6	4.7
7/20/94												4.6	4.6	4.5	4.5	4.5	4.5	4.5	4.6	4.5	4.5
8/2/94	4.8	4.8	4.8	4.9	5.2	5.0	5.0	4.9	4.9												
8/4/94												4.8	4.8	4.8	4.8	4.8	4.9	4.7	4.8	4.8	4.8
8/18/94												4.8	4.7	4.7	4.8	4.6	4.4	4.8	4.8	4.8	4.9
9/14/94												5.1	5.1	5.0	5.0	5.0	5.0	5.0	5.0	5.0	5.0
9/15/94	5.0	5.3	5.0	5.0	5.0	5.0	5.0	5.1	5.0												
10/4/94	5.2	5.2	5.2	5.0	5.2	5.2	5.2	5.1	5.2												
10/6/94												5.3	5.3	5.3	5.3	5.3	5.3	5.4	5.2	5.3	5.3
11/15/94	5.6	5.6	7.0	5.6	5.6	5.6	5.6	5.6	5.6												
11/17/94												5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
12/13/94												5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
12/14/94	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6												
1/10/95	7.0	7.0	7.0	7.0	7.0	7.0	7.0	7.0	7.0												
1/25/95												7.0	7.0	7.0	7.0	7.0	7.0	7.0	7.0	7.0	7.0
2/21/95	6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5	6.5												
2/22/95												5.2	5.2	5.2	5.2	5.2	5.2	6.5	6.5	5.2	5.2
3/13/95	6.5	6.5	6.5	5.2	5.2	5.2	6.5	6.5	6.5												
3/15/95										6.5	6.5	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2
4/4/95										6.5	6.5	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2	5.2
4/18/95										6.5	6.5										

Appendix Table A-15. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												5.2	5.2	5.2	5.2	6.5	6.5	5.2	5.2	5.2	5.2
4/21/95	6.5	6.5	6.5	5.2	5.2	5.2	5.2	5.2	5.2												
5/1/95										4.7	4.6										
5/2/95																				4.8	4.8
5/3/95												4.3	4.3	4.5	4.4	4.7	4.5	4.3	4.2		
5/15/95	4.6	4.5	4.5	4.5	4.5	4.4	4.4	4.4	4.4												
5/17/95										5.1	5.0	4.2	4.1	4.4	4.1	4.2	4.1	4.0	4.0	4.2	4.2
5/30/95										4.3	4.4										
5/31/95												4.0	4.0	4.0	4.0	4.1	4.1	4.0	3.9	4.1	4.1
6/12/95	4.3	4.4	4.3	4.0	4.0	4.0	3.8	3.7	3.8												
6/14/95												3.4	3.4	3.5	3.6	3.5	3.5	3.6	3.6	3.6	3.6
6/15/95										4.0	4.0										
6/28/95										4.0	4.0										
6/29/95												3.9	3.9	3.9	3.9	4.0	4.0	4.0	4.0	4.0	4.0
7/10/95	4.1	3.9	4.0	4.0	3.9	4.0	4.1	4.0	4.0												
7/12/95												3.8	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.9	3.9
7/13/95										4.1	4.1										
7/24/95										3.9	3.9										
7/25/95												3.7	3.7	4.0	3.9	3.9	4.0	3.8	3.8	3.8	3.8
8/8/95										4.6	4.7	4.6	4.6	4.6	4.6	4.7	4.6	4.7	4.6	4.6	4.6
8/10/95	4.8	4.6	4.6	4.6	4.6	4.6	4.6	4.7	4.6												
8/21/95										4.9	4.9										
8/22/95												4.8	4.8	4.8	4.9	4.8	4.8	4.9	4.9	4.8	4.8
9/5/95										5.1	5.1										
9/7/95												5.0	4.9	5.0	4.9	5.0	5.0	5.1	5.0	5.0	5.0
9/8/95	5.0	5.1	5.0	5.0	5.1	5.0	5.1	5.0	5.0												
9/19/95										5.1	5.3	5.0	5.0	5.0	4.8	4.9	5.0	5.1	5.0	5.0	5.0

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-16. Sodium concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	6.1	6.1	5.9	6.1	5.8	5.8	6.3	6.2	6.1												
4/13/94												6.8	6.9	6.6	6.7	6.5	6.4	6.9	7.1	6.9	6.9
4/26/94												7.2	7.2	7.1	7.1	7.1	7.0	7.2	7.2	7.0	6.9
5/10/94	6.1	5.3	5.3	5.1	4.9	5.0	4.6	4.6	4.6												
5/11/94												5.3	5.2	5.1	5.1	5.0	--[c]	5.3	5.3	4.9	5.1
5/24/94												5.7	5.7	5.5	5.5	5.5	5.5	5.6	5.6	5.6	5.5
6/7/94	4.6	4.7	4.0	4.0	3.9	4.0															
6/8/94												4.0	4.0	3.7	3.7	3.8	3.7	4.0	3.9	3.7	3.7
6/9/94							3.6	3.7	3.6												
6/21/94												4.0	4.1	4.1	4.1	4.1	4.1	4.2	4.2	4.0	4.2
7/6/94	3.6	3.6	3.6	3.6	3.6	3.6	3.5	3.5	3.5												
7/7/94												4.1	4.0	4.1	4.1	4.1	4.2	4.1	4.1	4.0	4.1
7/20/94												4.5	4.5	4.6	4.7	4.7	4.7	4.7	4.7	4.7	4.8
8/2/94	5.4	5.4	5.5	5.4	5.5	5.6	5.7	5.7	5.6												
8/4/94												5.6	5.6	5.5	5.5	5.5	5.5	5.6	5.6	5.3	5.3
8/18/94												3.9	3.9	3.9	3.9	3.9	3.9	4.0	3.9	3.9	3.9
9/14/94												4.6	4.6	4.5	4.5	4.5	4.5	4.6	4.6	4.5	4.5
9/15/94	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4	4.4												
10/4/94	5.1	5.1	5.1	5.1	5.1	5.1	4.8	4.8	4.8												
10/6/94												5.2	5.2	5.2	5.1	5.2	5.2	5.5	5.4	5.3	5.3
11/15/94	8.6	8.4	8.3	7.4	7.4	7.1	8.0	7.7	7.6												
11/17/94												8.3	8.1	8.1	8.1	8.3	8.1	8.1	8.1	7.4	7.6
12/13/94												6.2	6.0	6.3	5.9	6.0	5.9	5.9	6.2	6.3	5.9
12/14/94	8.1	7.4	7.1	7.3	7.0	7.1	7.0	6.7	7.0												
1/10/95	9.8	9.8	8.4	8.4	8.4	8.4	8.4	8.4	8.4												
1/25/95												9.8	9.8	9.8	8.4	9.8	11.2	11.2	11.2	9.8	9.8
2/21/95	11.2	11.3	11.2	9.0	8.9	8.8	10.7	10.7	10.7												
2/22/95												8.2	8.2	10.0	10.0	7.7	7.8	8.0	8.0	7.2	7.3
3/13/95	8.0	7.9	7.8	7.3	7.9	7.9	8.8	8.5	8.8												
3/15/95										8.0	8.0	8.6	8.6	7.0	7.0	6.4	6.4	9.0	8.9	5.6	5.6
4/4/95										8.0	8.0	8.7	8.8	7.2	7.2	7.0	7.0	9.0	8.9	5.8	5.8
4/18/95										7.9	7.9										

Appendix Table A-16. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												8.8	8.8	7.2	7.2	7.2	7.2	9.0	9.0	5.8	5.8
4/21/95	7.9	7.8	7.8	7.5	7.9	7.9	8.9	8.6	8.8												
5/1/95										7.4	7.6										
5/2/95																				6.4	6.5
5/3/95												6.5	6.6	6.0	6.3	6.0	6.0	6.3	6.4		
5/15/95	6.8	6.9	6.9	6.6	6.5	6.5	6.2	6.2	6.2												
5/17/95										6.8	7.0	6.0	6.1	5.5	5.5	5.5	5.3	6.2	6.2	5.1	5.2
5/30/95										5.1	5.1										
5/31/95												5.6	5.6	5.1	5.1	5.0	4.9	5.6	5.7	4.9	4.8
6/12/95	6.0	5.7	5.7	5.5	5.4	5.4	5.0	5.1	5.0												
6/14/95												4.7	4.6	4.1	4.1	4.1	4.1	4.5	4.4	3.8	3.9
6/15/95										5.0	4.6										
6/28/95										4.7	4.5										
6/29/95												4.7	4.7	4.1	4.1	3.7	3.7	4.7	4.8	3.7	3.6
7/10/95	4.6	4.4	4.4	4.3	4.4	4.3	4.6	4.7	4.7												
7/12/95												4.2	4.3	4.4	4.3	4.4	4.2	4.2	4.3	4.2	4.3
7/13/95										4.8	4.8										
7/24/95										5.5	5.4										
7/25/95												4.6	4.5	4.6	4.5	4.6	4.7	4.8	4.5	4.6	4.7
8/8/95										5.1	5.2	4.7	4.8	4.7	4.8	4.6	4.8	4.7	4.6	4.8	4.7
8/10/95	5.0	4.6	4.7	4.6	4.5	4.6	4.7	4.7	4.7												
8/21/95										4.9	5.1										
8/22/95												4.9	5.0	4.8	4.9	4.9	5.0	5.0	5.0	4.7	4.9
9/5/95										5.5	5.6										
9/7/95												5.0	5.1	5.0	5.0	5.0	5.0	5.1	5.2	4.9	5.0
9/8/95	5.1	5.2	5.2	5.2	5.2	5.3	4.9	5.0	5.1												
9/19/95										5.6	6.0	5.3	5.1	4.9	5.1	5.0	5.0	5.1	5.3	4.9	5.0

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-17. Potassium concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	1.5	1.4	1.4	1.4	1.4	1.4	1.5	1.5	1.5												
4/13/94												1.6	1.6	1.6	1.5	1.5	1.5	1.6	1.6	1.6	1.6
4/26/94												1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7
5/10/94	1.4	1.2	1.9	1.2	1.2	1.2	1.1	1.1	1.1												
5/11/94												1.2	1.2	1.2	1.2	1.2	--[c]	1.2	1.2	1.2	1.2
5/24/94												1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.2	1.1
6/7/94	1.1	1.0	0.9	0.9	1.0	0.9															
6/8/94												0.9	0.9	0.9	0.8	0.9	0.9	0.9	0.9	0.8	0.8
6/9/94							0.9	0.8	0.9												
6/21/94												0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
7/6/94	0.8	0.8	0.8	0.9	0.8	0.8	0.8	0.8	0.8												
7/7/94												0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
7/20/94												0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
8/2/94	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9												
8/4/94												0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9	0.9
8/18/94												1.0	1.0	1.0	0.9	0.9	0.9	1.0	1.0	1.0	1.0
9/14/94												1.1	1.1	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
9/15/94	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0												
10/4/94	1.1	1.1	1.1	1.1	1.1	1.2	1.1	1.1	1.1												
10/6/94												1.1	1.1	1.1	1.1	1.1	1.1	1.2	1.2	1.1	1.1
11/15/94	1.4	1.4	1.5	1.4	1.4	1.5	1.4	1.4	1.4												
11/17/94												1.5	1.5	1.5	1.5	1.4	1.4	1.4	1.4	1.4	1.4
12/13/94												1.2	1.2	1.0	1.0	1.0	1.0	1.1	1.1	1.2	1.0
12/14/94	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3	1.3												
1/10/95	1.4	1.4	1.3	1.3	1.3	1.3	1.2	1.3	1.2												
1/25/95												1.7	1.5	1.5	1.6	1.5	1.6	1.6	1.5	1.5	1.5
2/21/95	2.3	2.3	2.2	2.0	2.0	2.0	2.0	2.0	2.0												
2/22/95												2.1	1.9	1.7	1.6	1.6	1.6	1.9	2.0	1.6	1.6
3/13/95	1.7	1.8	1.9	1.8	1.9	1.9	2.0	2.0	2.0												
3/15/95										1.8	1.8	2.1	2.1	1.7	1.7	1.6	1.6	2.2	2.2	1.4	1.4
4/4/95										1.7	1.7	2.0	2.0	1.8	1.8	1.6	1.6	2.2	2.2	1.4	1.4
4/18/95										1.7	1.7										

Appendix Table A-17. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												1.9	1.9	1.7	1.7	1.6	1.6	1.8	2.0	1.3	1.3
4/21/95	1.7	1.7	1.8	1.8	1.8	1.9	1.9	1.9	2.0												
5/1/95										1.6	1.6										
5/2/95																				1.5	1.5
5/3/95												1.6	1.6	1.5	1.4	1.4	1.4	1.6	1.6		
5/15/95	1.5	1.5	1.5	1.5	1.5	1.5	1.7	1.4	1.4												
5/17/95										1.4	1.5	1.5	1.5	1.3	1.3	1.2	1.2	1.5	1.5	1.2	1.2
5/30/95										1.3	1.3										
5/31/95												1.3	1.3	1.3	1.3	1.2	1.2	1.3	1.3	1.2	1.2
6/12/95	1.4	1.4	1.3	1.3	1.3	1.3	1.2	1.2	1.2												
6/14/95												1.2	1.2	1.0	1.1	1.0	1.1	1.1	1.1	1.0	1.0
6/15/95										1.2	1.1										
6/28/95										1.0	1.0										
6/29/95												1.0	1.0	0.9	0.9	0.9	0.9	1.0	1.0	0.9	0.9
7/10/95	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.2												
7/12/95												1.1	1.1	1.1	1.1	1.0	1.1	1.1	1.1	1.1	1.0
7/13/95										1.2	1.2										
7/24/95										1.2	1.1										
7/25/95												1.0	1.0	1.0	1.0	1.1	1.0	1.0	1.0	1.0	1.0
8/8/95										1.1	1.2	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1
8/10/95	1.2	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1												
8/21/95										1.1	1.1										
8/22/95												1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1	1.1
9/5/95										1.2	1.2										
9/7/95												1.2	1.1	1.1	1.1	1.1	1.1	1.2	1.1	1.1	1.1
9/8/95	1.1	1.1	1.1	1.1	1.1	1.2	1.1	1.1	1.1												
9/19/95										1.2	1.3	1.2	1.2	1.1	1.1	1.1	1.1	1.2	1.4	1.1	1.1

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-18. Ammonia nitrogen (NH_3-N) concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	0.02	0.02	0.01	0.01	0.01	0.01	0.02	0.01	0.01												
4/13/94												0.01	0.01	0.01	0.00	0.01	0.01	0.01	0.01	0.01	0.00
4/26/94												0.02	0.02	0.01	0.02	0.02	0.02	0.00	0.02	0.00	0.01
5/10/94	0.04	0.03	0.03	0.02	0.03	0.03	0.02	0.02	0.03												
5/11/94												0.02	0.03	0.02	0.03	0.03	0.03	0.03	0.03	0.02	0.03
5/24/94												0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
6/7/94	0.06	0.05	0.01	0.05	0.05	0.01															
6/8/94												0.03	0.03	0.02	0.02	0.03	0.03	0.03	0.03	0.03	0.03
6/9/94							0.05	0.05	0.04												
6/21/94												0.03	0.03	0.04	0.04	0.04	0.03	0.04	0.03	0.04	0.03
7/6/94	0.04	0.04	0.05	0.04	0.04	0.01	0.04	0.04	0.03												
7/7/94												0.03	0.03	0.03	0.03	0.03	0.03	0.02	0.01	0.02	0.02
7/20/94												0.04	0.03	0.04	0.03	0.03	0.03	0.03	0.03	0.03	0.04
8/2/94	0.07	0.05	0.05	0.05	0.06	0.05	0.05	0.04	0.04												
8/4/94												0.03	0.05	0.04	0.04	0.04	0.04	0.03	0.03	0.04	0.04
8/18/94												0.03	0.03	0.03	0.04	0.03	0.04	0.04	0.04	0.02	0.03
9/14/94												0.01	0.03	0.02	0.02	0.02	0.02	0.02	0.01	0.01	0.01
9/15/94	0.03	0.04	0.04	0.04	0.01	0.02	0.02	0.02	0.02												
10/4/94	0.07	0.06	0.06	0.06	0.05	0.06	0.05	0.05	0.05												
10/6/94												0.05	0.05	0.04	0.04	0.04	0.04	0.04	0.04	0.03	0.03
11/15/94	0.02	0.02	0.03	0.02	0.02	0.02	0.01	0.02	0.02												
11/17/94												--[c]	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
12/13/94												0.01	0.01	0.01	0.01	0.01	0.01	0.00	0.01	0.01	0.01
12/14/94	0.02	0.01	0.01	0.02	0.02	0.02	0.01	0.01	0.01												
1/10/95	0.02	0.02	0.02	0.01	0.02	0.01	0.01	0.01	0.02												
1/25/95												0.02	0.02	0.01	0.01	0.02	0.01	0.02	0.01	0.02	0.02
2/21/95	0.04	0.03	0.02	0.02	0.02	0.02	0.03	0.03	0.02												
2/22/95												0.02	0.02	0.03	0.02	0.03	0.02	0.03	0.02	0.03	0.02
3/13/95	0.01	0.01	0.02	0.01	0.02	0.02	0.01	0.01	0.01												
3/15/95										0.03	0.03	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.03	0.01	0.01
4/4/95										0.03	0.02	0.01	0.01	0.01	0.01	0.02	0.01	0.01	0.01	0.01	0.01
4/18/95										0.07	0.06										

Appendix Table A-18. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03	0.03
4/21/95	0.03	0.03	0.03	0.02	0.02	0.02	0.03	0.03	0.03												
5/1/95										0.03	0.03										
5/2/95																				0.01	0.00
5/3/95												0.01	0.01	0.02	0.02	0.02	0.02	0.01	0.01		
5/15/95	0.02	0.02	0.04	0.01	0.03	0.03	0.02	0.03	0.01												
5/17/95										0.04	0.03	0.02	0.01	0.01	0.03	0.01	0.02	0.01	0.02	0.01	0.01
5/30/95										0.08	0.09										
5/31/95												0.02	0.02	0.01	0.01	0.02	0.02	0.01	0.01	0.02	0.01
6/12/95	0.03	0.02	0.02	0.01	0.04	0.02	0.03	0.02	0.01												
6/14/95												0.02	0.02	0.02	0.03	0.00	0.00	0.02	0.02	0.01	0.02
6/15/95										0.02	0.03										
6/28/95										0.06	0.03										
6/29/95												0.02	0.01	0.01	0.01	0.00	0.01	0.01	0.01	0.01	0.01
7/10/95	0.04	0.04	0.04	0.05	0.04	0.04	0.04	0.04	0.04												
7/12/95												0.03	0.03	0.04	0.06	0.05	0.02	0.01	0.01	0.01	0.01
7/13/95										0.01	0.02										
7/24/95										0.00	0.00										
7/25/95												0.02	0.02	0.01	0.01	0.01	0.01	0.02	0.02	0.00	0.00
8/8/95										0.01	0.01	0.02	0.02	0.01	0.02	0.01	0.02	0.02	0.01	0.03	0.02
8/10/95	0.03	0.03	0.03	0.02	0.03	0.03	0.02	0.02	0.01												
8/21/95										0.06	0.06										
8/22/95												0.04	0.02	0.02	0.02	0.02	0.02	0.04	0.03	0.06	0.03
9/5/95										0.07	0.06										
9/7/95												0.04	0.05	0.04	0.04	0.03	0.04	0.05	0.05	0.04	0.03
9/8/95	0.04	0.03	0.02	0.06	0.02	0.00	0.05	0.06	0.05												
9/19/95										0.00	0.00	0.04	0.02	0.00	0.00	0.01	0.02	0.04	0.03	0.00	0.00

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-19.Nitrite + nitrate nitrogen (NO_2+NO_3-N) concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	0.10	0.17	0.17	0.17	0.15	0.17	0.19	0.21	0.21												
4/13/94												0.26	0.25	0.26	0.26	0.25	0.25	0.25	0.28	0.25	0.24
4/26/94												0.23	0.23	0.22	0.22	0.22	0.22	0.25	0.25	0.19	0.20
5/10/94	0.03	0.09	0.10	0.07	0.08	0.08	0.55	0.09	0.09												
5/11/94												0.12	0.13	0.10	0.10	0.10	0.10	0.13	0.14	0.07	0.08
5/24/94												0.10	0.11	0.09	0.09	0.09	0.09	0.12	0.13	0.06	0.07
6/7/94	0.02	0.02	0.02	0.02	0.03	0.03															
6/8/94												0.09	0.09	0.07	0.07	0.06	0.06	0.09	0.09	0.06	0.06
6/9/94							0.03	0.04	0.05												
6/21/94												0.07	0.07	0.06	0.06	0.06	0.06	0.08	0.08	0.04	0.04
7/6/94	0.01	0.02	0.01	0.01	0.01	0.02	0.01	0.01	0.01												
7/7/94												0.06	0.05	0.06	0.05	0.04	0.04	0.06	0.06	0.04	0.04
7/20/94												0.03	0.03	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
8/2/94	0.03	0.03	0.04	0.03	0.03	0.03	0.02	0.03	0.04												
8/4/94												0.05	0.05	0.04	0.04	0.04	0.04	0.05	0.05	0.03	0.04
8/18/94												0.05	0.05	0.04	0.04	0.04	0.04	0.05	0.05	0.02	0.03
9/14/94												0.02	0.03	0.02	0.02	0.02	0.02	0.02	0.02	0.01	0.01
9/15/94	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01												
10/4/94	0.28	0.29	0.33	0.21	0.23	0.12	0.08	0.10	0.35												
10/6/94												0.33	0.08	0.07	0.08	0.09	0.01	0.07	0.19	0.21	0.22
11/15/94	0.20	0.21	0.16	0.20	0.21	0.21	0.22	0.23	0.23												
11/17/94												--[c]	0.27	0.24	0.24	0.24	0.24	0.28	0.27	0.23	0.23
12/13/94												0.28	0.28	0.22	0.22	0.22	0.22	0.27	0.27	0.22	0.22
12/14/94	0.27	0.26	0.27	0.28	0.28	0.28	0.26	0.26	0.27												
1/10/95	0.29	0.28	0.28	0.29	0.29	0.29	0.26	0.26	0.27												
1/25/95												0.49	0.53	0.46	0.42	0.38	0.43	0.51	0.54	0.43	0.46
2/21/95	0.46	0.48	0.49	0.52	0.54	0.54	0.54	0.54	0.54												
2/22/95												0.63	0.60	0.50	0.51	0.49	0.49	0.58	0.58	0.47	0.48
3/13/95	0.23	0.47	0.52	0.51	0.53	0.54	0.56	0.55	0.55												
3/15/95										0.49	0.49	0.65	0.62	0.51	0.51	0.47	0.47	0.64	0.67	0.42	0.42
4/4/95										0.29	0.28	0.52	0.52	0.49	0.49	0.52	0.53	0.54	0.54	0.51	0.49
4/18/95										0.19	0.20										

Appendix Table A-19. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												0.35	0.35	0.30	0.30	0.31	0.33	0.34	0.35	0.32	0.31
4/21/95	0.24	0.26	0.35	0.35	0.36	0.37	0.26	0.34	0.34												
5/1/95										0.15	0.15										
5/2/95																				0.29	0.27
5/3/95												0.29	0.25	0.22	0.24	0.19	0.21	0.25	0.24		
5/15/95	0.14	0.21	0.20	0.17	0.19	0.22	0.16	0.16	0.18												
5/17/95										0.06	0.07	0.26	0.22	0.13	0.18	0.14	0.20	0.26	0.24	0.12	0.13
5/30/95										0.04	0.05										
5/31/95												0.16	0.16	0.15	0.15	0.11	0.11	0.15	0.16	0.13	0.14
6/12/95	0.11	0.13	0.13	0.11	0.12	0.12	0.12	0.13	0.11												
6/14/95												0.13	0.13	0.11	0.11	0.11	0.11	0.13	0.12	0.10	0.11
6/15/95										0.07	0.08										
6/28/95										0.06	0.06										
6/29/95												0.11	0.11	0.11	0.11	0.07	0.10	0.12	0.10	0.07	0.08
7/10/95	0.10	0.10	0.10	0.11	0.11	0.12	0.10	0.10	0.12												
7/12/95												0.11	0.11	0.11	0.11	0.12	0.12	0.11	0.11	0.12	0.13
7/13/95										0.01	0.01										
7/24/95										0.02	0.01										
7/25/95												0.10	0.11	0.10	0.12	0.11	0.11	0.09	0.12	0.09	0.01
8/8/95										0.00	0.00	0.08	0.08	0.08	0.08	0.10	0.08	0.08	0.08	0.08	0.07
8/10/95	0.03	0.03	0.03	0.03	0.03	0.03	0.10	0.04	0.04												
8/21/95										0.00	0.00										
8/22/95												0.04	0.05	0.06	0.06	0.08	0.07	0.05	0.05	0.03	0.06
9/5/95										0.00	0.00										
9/7/95												0.08	0.09	0.08	0.08	0.08	0.08	0.08	0.09	0.09	0.09
9/8/95	0.01	0.02	0.03	0.01	0.03	0.03	0.01	0.02	0.02												
9/19/95										0.01	0.01	0.05	0.08	0.07	0.07	0.08	0.07	0.08	0.09	0.07	0.07

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-20. Total nitrogen concentrations (mg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	0.41	0.62	0.49	0.50	0.48	0.49	0.49	0.53	0.53												
4/13/94												0.54	0.56	0.54	0.53	0.53	0.54	0.54	0.59	0.53	0.51
4/26/94												0.47	0.47	0.49	0.47	0.47	0.47	0.49	0.50	0.44	0.49
5/10/94	0.31	0.33	0.36	0.28	0.33	0.33	0.35	0.31	0.33												
5/11/94												0.36	0.45	0.33	0.33	0.33	0.34	0.34	0.39	0.31	0.33
5/24/94												0.33	0.38	0.30	0.32	0.30	0.32	0.38	0.40	0.33	0.35
6/7/94	0.28	0.25	0.42	0.25	0.25	0.55															
6/8/94												0.28	0.28	0.27	0.22	0.25	0.27	0.27	0.32	0.27	0.30
6/9/94							0.25	0.25	0.27												
6/21/94												0.23	0.25	0.22	0.22	0.22	0.22	0.23	0.25	0.20	0.22
7/6/94	0.20	0.21	0.36	0.38	0.21	0.26	0.25	0.25	0.25												
7/7/94												0.28	0.30	0.25	0.26	0.25	0.28	0.26	0.35	0.26	0.30
7/20/94												0.23	0.23	0.18	0.18	0.21	0.21	0.21	0.23	0.20	0.28
8/2/94	0.25	0.23	0.18	0.21	0.20	0.23	0.21	0.21	0.27												
8/4/94												0.27	0.27	0.20	0.23	0.37	0.20	0.21	0.21	0.21	0.21
8/18/94												0.28	0.30	0.25	0.22	0.22	0.30	0.23	0.27	0.17	0.18
9/14/94												0.37	0.30	0.28	0.26	0.25	0.19	0.21	0.23	0.25	0.25
9/15/94	0.21	0.21	0.32	0.28	0.28	0.19	0.18	0.23	0.26												
10/4/94	0.28	0.26	0.30	0.25	0.23	0.26	0.25	0.23	0.25												
10/6/94												0.28	0.27	0.26	0.35	0.30	0.28	0.44	0.33	0.30	0.35
11/15/94	0.34	0.36	0.36	0.36	0.37	0.44	0.41	0.41	0.41												
11/17/94												--[c]	0.46	0.39	0.37	0.41	0.39	0.44	0.44	0.37	0.37
12/13/94												0.40	0.40	0.33	0.37	0.35	0.35	0.42	0.40	0.33	0.35
12/14/94	0.39	0.37	0.39	0.40	0.40	0.39	0.39	0.39	0.40												
1/10/95	0.41	0.42	0.39	0.44	0.42	0.41	0.35	0.41	0.41												
1/25/95												0.42	0.70	0.55	0.53	0.60	0.53	0.63	0.67	0.60	0.53
2/21/95	0.93	0.76	0.68	0.64	0.66	0.64	0.62	0.64	0.62												
2/22/95												0.70	0.87	0.60	0.62	0.58	0.58	0.76	0.76	0.58	0.56
3/13/95	0.60	0.52	0.50	0.60	0.66	0.68	0.60	0.78	0.60												
3/15/95										0.85	0.78	0.78	0.82	0.64	0.62	0.62	0.60	0.80	0.85	0.56	0.60
4/4/95										0.80	0.91	0.84	1.08	0.84	0.84	0.87	0.85	0.91	0.91	0.84	0.80
4/18/95										0.75	0.67										

Appendix Table A-20. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												0.61	0.61	0.58	0.61	0.58	0.56	0.54	0.59	0.58	0.58
4/21/95	0.38	0.46	0.50	0.58	0.58	0.59	0.59	0.59	0.61												
5/1/95										0.66	0.68										
5/2/95																				0.53	0.51
5/3/95												0.63	0.64	0.50	0.53	0.55	0.55	0.55	0.57		
5/15/95	0.51	0.55	0.55	0.50	0.51	0.48	0.44	0.50	0.46												
5/17/95										0.61	0.59	0.61	0.51	0.44	0.46	0.48	0.44	0.50	0.50	0.42	0.50
5/30/95										0.37	0.35										
5/31/95												0.39	0.37	0.35	0.35	0.33	0.33	0.33	0.35	0.31	0.33
6/12/95	0.43	0.35	0.41	0.34	0.35	0.35	0.35	0.35	0.34												
6/14/95												0.32	0.35	0.30	0.50	0.34	0.32	0.32	0.32	0.28	0.27
6/15/95										0.35	0.35										
6/28/95										0.30	0.27										
6/29/95												0.29	0.45	0.31	0.25	0.22	0.23	0.32	0.40	0.26	0.25
7/10/95	0.30	0.04	0.29	0.30	0.29	0.26	0.35	0.30	0.30												
7/12/95												0.33	0.35	0.32	0.29	0.38	0.35	0.35	0.30	0.32	0.33
7/13/95										0.45	0.33										
7/24/95										0.41	0.39										
7/25/95												0.33	0.35	0.29	0.29	0.29	0.31	0.29	0.37	0.29	0.29
8/8/95										0.30	0.33	0.25	0.26	0.25	0.25	0.23	0.23	0.25	0.26	0.30	0.25
8/10/95	0.22	0.20	0.20	0.18	0.20	0.20	0.23	0.22	0.22												
8/21/95										0.34	0.38										
8/22/95												0.29	0.29	0.29	0.29	0.21	0.23	0.29	0.30	0.34	0.29
9/5/95										0.40	0.35										
9/7/95												0.28	0.31	0.28	0.31	0.32	0.32	0.32	0.35	0.28	0.32
9/8/95	0.23	0.28	0.32	0.28	0.28	0.32	0.28	0.25	0.28												
9/19/95										0.23	0.34	0.26	0.28	0.25	0.26	0.26	0.28	0.29	0.28	0.26	0.26

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-21.Orthophosphate concentrations (µg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	1.0	1.0	1.0	0.1	1.0	1.0	0.1	1.0	1.0												
4/13/94												3.0	3.0	3.0	3.0	4.0	4.0	2.0	3.0	3.0	2.0
4/26/94												3.0	4.0	5.0	3.0	3.0	--[c]	2.0	4.0	4.0	--
5/10/94	0.1	0.1	0.1	0.1	0.1	1.0	0.1	0.1	0.1												
5/11/94												1.0	1.0	3.0	4.0	1.0	1.0	3.0	2.0	3.0	1.0
5/24/94												1.0	1.0	0.1	0.1	3.0	0.1	1.0	1.0	0.1	0.1
6/7/94	2.0	3.0	5.0	3.0	3.0	5.0															
6/8/94												3.0	4.0	7.0	7.0	3.0	2.0	5.0	5.0	4.0	2.0
6/9/94							3.0	3.0	3.0												
6/21/94												5.0	6.0	5.0	5.0	5.0	5.0	6.0	6.0	4.0	5.0
7/6/94	5.0	6.0	8.0	4.0	5.0	5.0	3.0	2.0	3.0												
7/7/94												4.0	4.0	3.0	3.0	4.0	2.0	3.0	4.0	0.1	2.0
7/20/94												0.1	0.1	1.0	2.0	0.1	0.1	0.1	1.0	0.1	0.1
8/2/94	13.0	11.0	15.0	10.0	10.0	11.0	5.0	6.0	7.0												
8/4/94												6.0	8.0	8.0	5.0	5.0	7.0	7.0	7.0	4.0	5.0
8/18/94												8.0	9.0	10.0	3.0	12.0	10.0	8.0	11.0	0.1	6.0
9/14/94												0.1	0.1	0.1	0.1	0.1	0.1	6.0	4.0	0.1	0.1
9/15/94	9.0	7.0	7.0	5.0	6.0	6.0	3.0	3.0	1.0												
10/4/94	5.0	6.0	6.0	7.0	17.0	6.0	1.0	0.1	2.0												
10/6/94												5.0	1.0	0.1	15.0	0.1	0.1	0.1	0.1	0.1	10.0
11/15/94	14.0	15.0	14.0	15.0	15.0	16.0	16.0	17.0	17.0												
11/17/94												--	20.0	18.0	18.0	18.0	18.0	21.0	20.0	17.0	17.0
12/13/94												12.0	12.0	10.0	11.0	12.0	11.0	13.0	12.0	10.0	9.0
12/14/94	13.0	12.0	13.0	14.0	15.0	16.0	12.0	15.0	13.0												
1/10/95	14.0	14.0	14.0	14.0	14.0	14.0	11.0	12.0	12.0												
1/25/95												28.0	29.0	26.0	27.0	26.0	25.0	29.0	29.0	24.0	24.0
2/21/95	22.0	30.0	30.0	18.0	21.0	20.0	16.0	15.0	17.0												
2/22/95												22.0	26.0	15.0	16.0	15.0	15.0	26.0	24.0	16.0	15.0
3/13/95	17.0	20.0	20.0	19.0	19.0	20.0	20.0	20.0	22.0												
3/15/95										3.0	2.0	31.0	34.0	21.0	23.0	21.0	20.0	33.0	32.0	15.0	16.0
4/4/95										0.1	0.1	22.0	22.0	19.0	20.0	21.0	23.0	22.0	23.0	20.0	20.0
4/18/95										1.0	0.1										

Appendix Table A-21. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												11.0	12.0	11.0	10.0	11.0	10.0	13.0	13.0	8.0	8.0
4/21/95	15.0	14.0	14.0	13.0	13.0	14.0	10.0	9.0	10.0												
5/1/95										1.0	1.0										
5/2/95																				4.0	5.0
5/3/95												24.0	11.0	3.0	2.0	3.0	5.0	10.0	11.0		
5/15/95	7.0	6.0	5.0	2.0	7.0	7.0	4.0	6.0	8.0												
5/17/95										1.0	0.1	14.0	15.0	8.0	7.0	7.0	7.0	15.0	16.0	4.0	4.0
5/30/95										1.0	1.0										
5/31/95												9.0	10.0	7.0	7.0	8.0	7.0	8.0	12.0	2.0	5.0
6/12/95	4.0	4.0	4.0	4.0	5.0	2.0	1.0	4.0	4.0												
6/14/95												5.0	4.0	3.0	3.0	4.0	5.0	5.0	3.0	0.1	1.0
6/15/95										0.1	0.1										
6/28/95										0.1	0.1										
6/29/95												4.0	6.0	4.0	4.0	0.1	7.0	3.0	5.0	0.1	2.0
7/10/95	10.0	12.0	11.0	14.0	10.0	11.0	12.0	11.0	10.0												
7/12/95												15.0	11.0	13.0	15.0	38.0	29.0	13.0	12.0	9.0	9.0
7/13/95										8.0	5.0										
7/24/95										3.0	3.0										
7/25/95												7.0	9.0	8.0	8.0	10.0	9.0	7.0	9.0	6.0	8.0
8/8/95										0.1	0.1	5.0	5.0	6.0	6.0	6.0	6.0	6.0	5.0	3.0	5.0
8/10/95	6.0	6.0	7.0	3.0	4.0	6.0	0.1	0.1	1.0												
8/21/95										0.1	0.1										
8/22/95												0.1	0.1	1.0	0.1	1.0	0.1	0.1	0.1	0.1	0.1
9/5/95										2.0	1.0										
9/7/95												0.1	1.0	1.0	2.0	3.0	2.0	2.0	2.0	2.0	2.0
9/8/95	0.1	3.0	4.0	1.0	2.0	1.0	0.1	0.1	0.1												
9/19/95										0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-22. Total phosphorus concentrations (µg/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

	Lower/Mid reservoir[a]									Backwater[a]		Upper reservoir[a]									
	352			355			390			424		443a		443b		445		446		449	
Date	S[b]	M[b]	B[b]	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/11/94	36.0	46.0	34.0	24.0	26.0	30.0	24.0	34.0	34.0												
4/13/94												34.0	34.0	24.0	30.0	30.0	44.0	34.0	44.0	26.0	26.0
4/26/94												34.0	38.0	34.0	28.0	28.0	34.0	34.0	34.0	32.0	52.0
5/10/94	70.0	28.0	90.0	18.0	22.0	24.0	22.0	70.0	80.0												
5/11/94												32.0	68.0	34.0	32.0	28.0	28.0	32.0	38.0	24.0	28.0
5/24/94												36.0	66.0	26.0	30.0	34.0	30.0	36.0	42.0	20.0	26.0
6/7/94	22.0	22.0	51.0	22.0	26.0	80.0															
6/8/94												22.0	26.0	16.0	16.0	16.0	12.0	18.0	18.0	16.0	18.0
6/9/94							18.0	16.0	22.0												
6/21/94												16.0	18.0	18.0	16.0	18.0	18.0	22.0	22.0	12.0	22.0
7/6/94	22.0	26.0	43.0	22.0	22.0	36.0	18.0	26.0	38.0												
7/7/94												42.0	38.0	22.0	18.0	22.0	26.0	22.0	36.0	22.0	38.0
7/20/94												22.0	28.0	12.0	18.0	18.0	22.0	16.0	28.0	18.0	26.0
8/2/94	20.0	20.0	16.0	20.0	16.0	22.0	16.0	20.0	30.0												
8/4/94												22.0	22.0	20.0	20.0	14.0	16.0	16.0	22.0	16.0	22.0
8/18/94												27.0	38.0	23.0	27.0	25.0	36.0	32.0	46.0	21.0	29.0
9/14/94												44.0	51.0	39.0	41.0	39.0	43.0	43.0	44.0	39.0	39.0
9/15/94	39.0	46.0	46.0	36.0	44.0	41.0	41.0	43.0	48.0												
10/4/94	44.0	53.0	48.0	39.0	41.0	50.0	44.0	46.0	53.0												
10/6/94												43.0	53.0	43.0	48.0	48.0	32.0	50.0	50.0	46.0	48.0
11/15/94	28.0	30.0	39.0	28.0	34.0	64.0	35.0	34.0	44.0												
11/17/94												--[c]	53.0	37.0	34.0	53.0	40.0	41.0	41.0	35.0	35.0
12/13/94												25.0	36.0	21.0	32.0	21.0	25.0	25.0	27.0	25.0	20.0
12/14/94	23.0	23.0	23.0	25.0	29.0	29.0	29.0	27.0	30.0												
1/10/95	23.0	23.0	23.0	26.0	28.0	25.0	23.0	32.0	28.0												
1/25/95												51.0	64.0	45.0	45.0	55.0	40.0	53.0	77.0	72.0	94.0
2/21/95	112.0	130.0	96.0	53.0	49.0	51.0	41.0	43.0	43.0												
2/22/95												94.0	185.0	43.0	43.0	43.0	41.0	110.0	141.0	47.0	43.0
3/13/95	57.0	49.0	47.0	35.0	49.0	49.0	57.0	75.0	57.0												
3/15/95										63.0	57.0	79.0	92.0	47.0	51.0	49.0	47.0	92.0	98.0	41.0	47.0
4/4/95										25.0	20.0	42.0	47.0	62.0	50.0	74.0	44.0	52.0	90.0	50.0	53.0
4/18/95										108.0	103.0										

Appendix Table A-22. Continued.

	Lower/Mid reservoir									Backwater		Upper reservoir									
	352			355			390			424		443a		443b		445		446		449	
Date	S	M	B	S	M	B	S	M	B	S	B	S	B	S	B	S	B	S	B	S	B
4/20/95												85.0	85.0	74.0	85.0	80.0	68.0	63.0	80.0	74.0	74.0
4/21/95	74.0	91.0	91.0	74.0	74.0	74.0	85.0	80.0	85.0												
5/1/95										53.0	55.0										
5/2/95																				46.0	34.0
5/3/95												89.0	93.0	36.0	42.0	40.0	42.0	65.0	72.0		
5/15/95	42.0	55.0	61.0	44.0	51.0	42.0	59.0	57.0	42.0												
5/17/95										57.0	46.0	80.0	53.0	40.0	40.0	46.0	57.0	53.0	51.0	44.0	53.0
5/30/95										21.0	21.0										
5/31/95												23.0	32.0	32.0	27.0	30.0	30.0	30.0	34.0	21.0	25.0
6/12/95	40.0	42.0	59.0	38.0	33.0	52.0	47.0	47.0	47.0												
6/14/95												61.0	66.0	38.0	47.0	52.0	42.0	38.0	38.0	35.0	29.0
6/15/95										33.0	43.0										
6/28/95										23.0	27.0										
6/29/95												40.0	67.0	35.0	35.0	28.0	28.0	45.0	63.0	27.0	32.0
7/10/95	35.0	45.0	35.0	28.0	27.0	30.0	37.0	28.0	37.0												
7/12/95												38.0	42.0	38.0	33.0	55.0	43.0	42.0	37.0	27.0	32.0
7/13/95										65.0	35.0										
7/24/95										48.0	50.0										
7/25/95												34.0	66.0	31.0	31.0	31.0	31.0	34.0	43.0	29.0	36.0
8/8/95										33.0	43.0	23.0	28.0	20.0	22.0	22.0	22.0	25.0	25.0	22.0	22.0
8/10/95	22.0	23.0	25.0	22.0	23.0	25.0	25.0	25.0	23.0												
8/21/95										14.0	25.0										
8/22/95												23.0	34.0	20.0	21.0	21.0	23.0	20.0	25.0	18.0	23.0
9/5/95										43.0	44.0										
9/7/95												23.0	34.0	23.0	34.0	30.0	32.0	28.0	39.0	23.0	34.0
9/8/95	23.0	36.0	46.0	27.0	28.0	41.0	27.0	20.0	28.0												
9/19/95										18.0	39.0	20.0	30.0	18.0	21.0	21.0	23.0	25.0	28.0	23.0	21.0

[a] Sampling stations: River kilometer (RKm) 352 = LePage; RKm 355 = Towal; RKm 390 = Arlington; RKm 424 = Crow Butte; RKm 443a = Long Walk Island (downstream station); RKm 443b = Sand Island; RKm 445 = Big Blalock Island; RKm 446 = Long Walk Island (upstream station); and RKm 449 = Paterson Slough.

[b] Denotes sampling depths: S = surface; M = mid-depth; and B = sample collected just off the bottom.

[c] Dashes indicate data are not available.

Appendix Table A-23. Monochromatic chlorophyll *a* content (mg/m^3) of depth-integrated samples collected at lower-reservoir, mid-reservoir, and backwater locations in John Day Reservoir, April 1994 to September 1995.

	RKm 352[a]			RKm 355			RKm 390			RKm 424		
Date	P1[b]	P2[c]	Tot.[d]	P1	P2	Tot.	P1	P2	Tot.	P1	P2	Tot.
4/11/94	1.5	7.3	8.8	2.7	13.2	15.9	2.1	2.0	4.1			
5/10/94	2.1	10.2	12.3	1.6	4.4	6.0	4.4	3.1	7.5			
6/7/94	2.5	2.7	5.2	3.5	2.2	5.7						
6/9/94							2.3	5.2	7.5			
7/6/94	0.9	1.3	2.2	1.5	0.8	2.3	1.2	0.4	1.6			
8/2/94	1.3	0.6	1.9				2.9	0.0	2.9			
9/15/94	1.9	0.5	2.4	1.6	0.8	2.4	2.9	2.7	5.6			
10/4/94	2.9	1.6	4.5	4.0	1.1	5.1	3.5	3.7	7.2			
11/15/94	2.0	2.7	4.7	2.3	3.0	5.3	2.4	1.5	3.9			
1/10/95	1.9	0.6	2.5	2.2	0.5	2.7	2.0	1.2	3.2			
3/13/95	2.3	0.6	2.9	3.3	1.0	4.3	4.0	1.5	5.5			
4/4/95										12.1	8.9	21.0
4/18/95										8.2	26.7	34.9
4/21/95	3.1	4.2	7.3	5.2	7.5	12.7	4.8	9.0	13.8			
5/1/95										15.7	7.0	22.7
5/15/95	1.3	1.8	3.1	1.5	9.0	10.5	3.2	0.7	3.9			
5/17/95												
5/30/95										5.1	3.5	8.6
6/12/95	1.2	2.6	3.8	1.0	0.8	1.8	0.4	0.7	1.1			
6/15/95										0.8	1.0	1.8
6/28/95										0.8	2.5	3.3
7/10/95	0.4	0.6	1.0	0.4	0.7	1.1	0.9	0.6	1.5			
7/13/95										1.1	1.4	2.5
7/24/95										1.4	3.8	5.2
8/8/95										1.8	2.8	4.6
8/10/95	0.7	1.1	1.8	0.7	1.2	1.9	1.3	2.2	3.5			
8/21/95										3.3	4.0	7.3
9/5/95										3.9	6.6	10.5
9/8/95	3.3	3.5	6.8	3.4	4.0	7.4	2.8	5.9	8.7			
9/19/95										4.6	6.8	11.4

[a] Sampling stations: RKm 352 = LePage, RKm 355 = Towal, RKm 390 = Arlington, RKm 424 = Crow Butte backwater.

[b] Monochromatic chlorophyll *a* content of phytoplankton passing through a 25µm net.

[c] Estimated monochromatic chlorophyll *a* content of phytoplankton retained by a 25µm net.

[d] Monochromatic chlorophyll *a* content of unfiltered samples.

Appendix Table A-24. Monochromatic chlorophyll *a* content (mg/m^3) of depth-integrated samples collected at locations in the upper John Day Reservoir, April 1994 to September 1995.

	RKm 443a[a]			RKm 443b			RKm 445			RKm 446			RKm 449		
Date	P1[b]	P2[c]	Tot.[d]	P1	P2	Tot.	P1	P2	Tot.	P1	P2	Tot.	P1	P2	Tot.
4/13/94	2.7	7.8	10.5				1.2	4.3	5.5	3.2	7.6	10.8			
4/26/94	2.0	1.8	3.8	0.6	2.4	3.0	0.2	1.8	2.0	1.3	1.0	2.3	0.7	0.4	1.1
5/11/94	1.6	6.5	8.1	2.0	7.4	9.4	2.0	5.7	7.7	3.1	3.5	6.6	2.7	0.2	2.9
5/24/94	1.1	1.2	2.3	1.7	4.7	6.4	2.3	2.9	5.2	0.8	5.5	6.3	2.8	1.5	4.3
6/8/94	4.3	0.6	4.9	2.1	3.4	5.5	1.5	0.8	2.3	3.3	1.5	4.8	0.9	3.0	3.9
6/21/94	0.3	0.5	0.8	0.8	0.5	1.3				1.1	1.0	2.1	1.1	0.2	1.3
7/7/94	1.6	2.1	3.7	1.6	1.6	3.2	1.6	3.2	4.8	2.1	2.4	4.5	2.1	4.6	6.7
7/20/94	1.6	1.3	2.9	1.8	0.3	2.1	1.9	0.8	2.7	2.4	0.3	2.7	2.7	1.0	3.7
8/4/94	2.1	0.8	2.9				1.9	0.2	2.1	1.9	0.0	1.9	1.3	0.6	1.9
8/18/94	2.1	3.2	5.3	2.1	1.4	3.5	1.6	2.7	4.3	1.9	2.1	4.0	2.1	2.2	4.3
9/14/94	0.8	0.0	0.8				3.2	4.0	7.2	4.5	4.0	8.5	4.0	3.7	7.7
10/6/94	3.7	10.2	13.9	4.3	9.1	13.4	4.3	10.4	14.7	5.6	12.6	18.2	4.5	10.7	15.2
11/17/94	2.3	2.2	4.5	1.6	2.5	4.1	1.7	1.0	2.7	2.1	1.9	4.0	2.7	1.0	3.7
1/25/95	1.2	0.2	1.4	1.1	0.4	1.5									
3/15/95	2.5	0.3	2.8	2.2	0.9	3.1	2.5	1.1	3.6	1.9	0.4	2.3	3.2	1.4	4.6
4/4/95	3.1	0.9	4.0	2.1	2.3	4.4	2.7	1.4	4.1	1.7	0.8	2.5	2.9	1.9	4.8
4/20/95	5.2	7.1	12.3	4.1	7.0	11.1	4.9	7.1	12.0	4.9	7.2	12.1	12.4	1.4	13.8
5/3/95	5.0	9.4	14.4	4.8	11.8	16.6	4.3	10.7	15.0	5.3	7.4	12.7	5.1	10.5	15.6
5/17/95				1.7	3.3	5.0	0.7	4.3	5.0	0.5	0.6	1.1	1.1	6.6	7.7
5/31/95	2.5	2.2	4.7	3.5	2.5	6.0	2.9	2.4	5.3	3.5	0.5	4.0	4.0	1.3	5.3
6/14/95	0.4	0.3	0.7	0.6	0.1	0.7	0.4	0.5	0.9	0.7	0.1	0.8	0.8	0.7	1.5
6/29/95	1.3	1.5	2.8	1.4	1.0	2.4	1.1	1.3	2.4	0.7	0.2	0.9	0.9	0.5	1.4
7/12/95	1.1	0.4	1.5	0.9	0.2	1.1	0.7	0.2	0.9	0.9	0.1	1.0			
7/25/95	0.6	0.8	1.4	0.7	0.5	1.2	0.7	0.4	1.1	0.8	0.5	1.3	0.9	0.5	1.4
8/8/95	0.9	1.1	2.0	0.9	0.9	1.8	1.0	0.9	1.9	1.0	1.0	2.0	1.0	1.0	2.0
8/22/95	2.9	3.8	6.7	2.7	2.7	5.4	2.1	2.3	4.4	3.6	3.3	6.9	3.0	2.6	5.6
9/7/95	2.6	5.6	8.2	2.0	6.4	8.4	2.2	4.3	6.5	2.7	4.8	7.5	2.7	4.1	6.8
9/19/95	1.9	8.2	10.1	2.8	5.8	8.6	3.1	5.0	8.1	3.1	5.7	8.8	2.7	5.7	8.4

[a] Sampling stations: RKm 443a = Long Walk Island (downstream station), RKm 443b = Sand Island, RKm 445 = Big Blalock Island, RKm 446 = Long Walk Island (upstream station, and RKm 449 = Paterson Slough.

[b] Monochromatic chlorophyll *a* content of phytoplankton passing through a 25µm net.

[c] Estimated monochromatic chlorophyll *a* content of phytoplankton retained by a 25µm net.

[d] Monochromatic chlorophyll *a* content of unfiltered samples.

Appendix Table A-25.Zooplankton densities at lower and middle John Day Reservoir sampling stations, April 1994 to September 1995.

Taxon	Mean number/L															
	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Jan	Mar	Apr	May	Jun	Jul	Aug	Sep
Cladocera																
Alona quadrangularis	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.01	0.00	0.00	0.00	0.00	0.08	0.00	0.00	0.00
Alona rectangula	0.00	0.00	0.00	0.00	0.27	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Alona rustica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00
Bosmina longirostris	0.00	0.07	0.82	2.16	0.55	3.32	8.04	0.96	0.12	0.14	0.32	1.13	0.96	1.23	1.54	2.69
Ceriodaphnia quadrangula	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.06	0.03	0.00	0.00
Chydorus sphaericus	0.02	0.00	0.03	0.04	0.02	0.04	0.06	0.02	0.06	0.00	0.00	0.00	0.06	0.00	0.00	0.00
Daphnia retrocurva	0.00	0.00	0.00	0.00	0.00	0.38	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.14	38.08	0.00
Daphnia thorata	0.00	0.02	0.21	1.73	19.58	0.06	0.03	0.00	0.00	0.00	0.00	0.00	0.06	0.28	1.46	0.00
Leptodora kindtii	0.00	0.00	0.03	0.00	0.27	0.11	0.02	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00
Leydigia leydigi	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Immature	0.00	0.02	0.06	0.60	2.29	0.03	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.33	4.21	0.00
Totals	0.02	0.11	1.15	4.53	22.98	3.98	8.36	1.00	0.20	0.14	0.32	1.13	1.22	2.06	45.34	2.69
Copepoda																
Calanoid copepodid	0.02	0.00	0.03	0.10	0.19	0.19	0.91	0.01	0.02	0.00	0.00	0.00	0.00	0.08	0.00	0.00
Cyclopoid copepodid	0.05	0.23	2.31	0.74	1.01	1.80	4.12	0.13	0.06	0.09	0.35	0.23	0.50	1.54	1.92	0.46
Diacyclops thomasi	0.01	0.34	0.76	1.19	2.10	0.99	2.95	0.14	0.03	0.00	0.27	0.29	0.06	0.42	10.43	1.12
Harpacticoid	0.00	0.00	0.00	0.03	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00
Leptodiaptomus ashlandi	0.03	0.00	0.12	0.11	0.32	0.07	1.14	0.03	0.00	0.00	0.13	0.00	0.00	0.00	0.26	0.00
Nauplii	0.34	1.76	4.00	8.54	9.97	11.15	18.10	1.81	0.38	0.17	1.08	1.24	1.77	2.19	65.63	10.98
Totals	0.45	2.33	7.22	10.71	13.64	14.20	27.22	2.12	0.49	0.26	1.83	1.76	2.38	4.23	78.24	12.56
Rotifera																
Asplanchna girodi	0.26	0.37	0.36	0.12	0.00	0.04	1.26	0.26	0.24	0.00	0.08	0.21	0.25	0.05	0.00	1.56
Bdelloid	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00
Brachionus angularis	0.00	0.00	0.04	0.09	0.26	0.08	0.00	0.00	0.00	0.04	0.00	0.00	0.08	2.07	1.07	1.39
Brachionus calyciflorus	0.22	0.59	0.03	0.00	0.00	2.45	0.57	1.03	0.15	0.32	0.23	0.90	0.00	0.00	0.00	0.68
Brachionus caudatus	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Brachionus quadridentatus	0.08	0.03	0.00	0.00	0.05	0.20	0.15	0.00	0.04	0.00	0.00	0.00	0.00	0.05	0.00	0.00

Appendix Table A-25. Continued.

Taxon	Mean number/L															
	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Jan	Mar	Apr	May	Jun	Jul	Aug	Sep
Rotifera (continued)																
Collotheca libera	0.00	0.00	0.06	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Collotheca spp.	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.46
Conochiloides dossaurius	0.00	0.00	0.03	6.49	1.97	0.11	0.78	2.47	0.05	0.00	0.00	0.00	0.18	0.05	0.00	5.24
Conochilus unicornis	0.00	0.00	0.03	0.37	0.43	0.12	0.74	0.05	0.00	0.00	0.00	0.20	0.31	0.09	0.15	0.21
Euchlanis calpidia	0.03	0.03	0.07	0.10	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.16	0.17	0.03	0.00	0.00
Euchlanis dialata	8.63	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Filinia longiseta	0.34	0.49	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.19	0.20	0.03	0.00	0.00	0.00
Gastropus hypotus	0.00	0.00	0.00	0.00	0.00	0.41	0.42	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Kellicottia bostoniensis	0.00	0.00	6.01	0.00	0.00	0.00	0.00	0.00	0.02	0.13	0.00	0.00	0.00	0.00	0.00	0.00
Kellicottia longispina	0.28	0.83	2.64	9.98	0.00	0.00	0.00	0.01	0.00	0.10	0.10	3.39	2.61	2.94	0.21	0.00
Keratella cochlearis	23.24	62.21	65.65	115.28	3.98	3.51	2.58	0.30	0.13	0.77	21.76	81.36	6.50	11.29	5.19	67.14
Keratella quadrata	5.36	9.65	1.15	1.61	0.00	0.02	0.00	0.01	0.34	0.21	2.64	1.25	0.26	0.00	0.00	0.00
Keratella serrulata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.00	0.00	0.00	0.00
Lecane spp.	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.96
Mytlinia mucronatum	0.00	0.02	0.65	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Notholca acuminata	5.64	2.10	17.64	0.21	0.00	0.00	0.00	0.00	0.04	0.16	0.91	18.99	1.00	0.00	0.00	0.00
Polyarthra vulgaris	4.27	48.77	29.32	56.25	13.33	21.83	39.79	1.40	0.21	0.73	33.39	22.87	4.14	12.34	114.81	280.82
Pompholyx sulcata	0.00	0.00	0.00	0.03	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.00	0.00
Synchaeta pectinata	0.20	1.14	34.96	77.08	1.31	0.41	0.69	3.60	2.48	4.96	22.97	20.46	10.48	85.20	6.35	6.80
Trichocerca spp.	0.01	0.00	0.46	6.17	13.37	1.20	3.65	0.16	0.02	0.00	0.00	0.00	0.12	6.52	24.69	49.58
Trichotria pocillum	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.21	0.03	0.00	0.00	0.20
Totals	48.60	126.23	159.19	273.78	34.80	30.38	50.63	9.34	3.72	7.42	82.38	150.20	26.16	120.74	152.47	418.04

Appendix Table A-26. Zooplankton densities at upper John Day Reservoir sampling stations, April 1994 to September 1995.

Taxon	Mean number/L															
	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Jan	Mar	Apr	May	Jun	Jul	Aug	Sep
Cladocera																
Alona afflinis	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Alona costata	0.00	0.12	0.00	0.00	0.00	0.00	0.00	0.01	0.03	0.00	0.00	0.00	0.00	0.01	0.00	0.00
Alona guttata	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03	0.03	0.00	0.00	0.00
Alona quadrangularis	0.00	0.00	0.00	0.05	0.04	0.00	0.00	0.01	0.00	0.00	0.00	0.06	0.02	0.00	0.00	0.00
Alona rectangula	0.00	0.00	0.06	0.05	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Alona rustica	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00
Bosmina longirostris	0.13	0.08	0.82	1.43	0.23	5.27	18.45	0.43	1.32	0.18	0.22	1.17	0.09	0.27	0.22	0.34
Camptocercus rectirostris	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Ceriodaphnia quadrangula	0.00	0.00	0.02	0.07	0.02	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00
Chydorus sphaericus	0.02	0.01	0.04	0.06	0.01	0.27	0.08	0.04	0.00	0.05	0.00	0.10	0.00	0.00	0.00	0.00
Daphnia retrocurva	0.00	0.00	0.00	0.00	1.57	0.06	0.22	0.00	0.00	0.00	0.00	0.00	0.05	0.05	0.66	0.22
Daphnia thorata	0.02	0.07	0.09	0.78	1.82	0.12	0.03	0.01	0.00	0.00	0.00	0.09	0.11	0.04	0.00	0.11
Diaphanasoma birgeii	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.06	0.09
Ilypcryptus spinifer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.01	0.00	0.00
Kurzia latissima	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Leptodora kindtii	0.00	0.02	0.00	0.05	0.00	0.02	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Leydigia leydigi	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Immature	0.04	0.00	0.05	0.02	0.54	0.00	0.03	0.01	0.00	0.00	0.00	0.11	0.03	0.01	0.16	0.03
Totals	0.24	0.30	1.08	2.51	4.27	5.76	18.91	0.52	1.35	0.23	0.23	1.58	0.35	0.39	1.10	0.79
Copepoda																
Calanoid copepodid	0.03	0.01	0.06	0.03	0.07	0.90	0.43	0.00	0.00	0.00	0.00	0.03	0.00	0.02	0.00	0.03
Cyclopoid copepodid	0.16	0.10	0.55	1.05	1.61	2.70	2.49	0.07	0.00	0.04	0.13	0.13	0.06	0.19	0.61	0.88
Diacyclops thomasi	0.49	0.08	0.16	0.24	0.58	0.71	1.76	0.04	0.00	0.00	0.06	0.08	0.04	0.10	0.30	0.54
Eucyclops agilis	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Harpacticoid	0.02	0.10	0.02	0.02	0.00	0.08	0.08	0.07	0.02	0.15	0.00	0.09	0.02	0.02	0.00	0.00
Leptodiaptomus ashlandi	0.06	0.00	0.00	0.07	0.03	0.25	1.20	0.01	0.00	0.00	0.00	0.00	0.01	0.05	0.06	0.03
Nauplii	2.75	4.66	5.06	3.60	5.62	4.62	7.54	0.35	0.44	0.46	0.86	3.05	1.69	1.53	2.25	1.82
Totals	3.52	4.95	5.85	5.01	7.91	9.26	13.50	0.54	0.46	0.65	1.05	3.38	1.82	1.91	3.22	3.31

Appendix Table A-26. Continued.

Taxon	Mean number/L															
	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Jan	Mar	Apr	May	Jun	Jul	Aug	Sep
Rotifera																
Ascomorpha ovalis	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Asplanchna girodi	0.35	0.28	0.07	0.11	0.00	0.12	1.13	0.00	0.02	0.00	0.02	0.04	0.00	0.00	0.00	1.46
Asplanchna sieboldi	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00
Bdelloid	0.04	0.01	0.35	0.11	0.01	0.00	0.00	0.01	0.00	0.00	0.04	0.29	0.37	0.32	0.32	0.00
Brachionus angularis	0.00	0.00	0.18	0.18	0.31	0.07	0.03	0.00	0.00	0.03	0.00	0.13	0.22	0.39	0.52	0.30
Brachionus calyciflorus	3.76	0.09	0.04	0.05	0.01	2.66	0.20	0.02	0.00	0.08	1.22	0.37	0.00	0.00	0.05	0.08
Brachionus caudatus	0.01	0.00	0.01	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Brachionus patulus	0.00	0.00	0.00	0.00	0.02	0.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Brachionus quadridentatus	0.42	0.00	0.01	0.02	0.08	0.35	0.18	0.00	0.00	0.00	0.00	0.02	0.02	0.00	0.00	0.00
Brachionus rubens	0.00	0.00	0.01	0.02	0.00	0.00	0.04	0.00	0.00	0.04	0.03	0.00	0.05	0.00	0.00	0.00
Collotheca libera	0.00	0.00	0.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Collotheca spp.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.60	0.84
Colurella spp.	0.01	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.08	0.18
Conochiloides dossaurius	0.01	0.00	0.00	4.65	0.94	0.40	3.46	0.95	0.00	0.00	0.00	0.08	0.06	0.10	3.58	8.53
Conochilus unicornis	0.00	0.00	0.20	0.24	0.55	0.67	1.06	0.04	0.00	0.00	0.00	0.00	0.03	0.00	0.49	0.94
Euchlanis calpidia	0.34	0.19	0.88	0.83	0.04	0.25	1.02	0.21	0.00	0.02	0.03	0.19	0.05	0.13	0.27	0.00
Euchlanis dialata	8.66	0.21	0.30	0.06	0.00	0.00	0.08	0.00	0.00	0.00	0.00	0.00	0.00	0.02	0.00	0.00
Filinia longiseta	3.04	0.47	0.01	0.00	0.52	0.00	0.00	0.01	0.03	0.02	0.58	1.33	0.03	0.00	0.08	0.10
Gastropus hypotus	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Gastropus stylifer	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.04
Hexarthra mira	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.01	0.00	0.00
Kellicottia bostoniensis	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.00	0.01	0.00	0.00
Kellicottia longispina	1.13	1.12	4.89	0.82	0.17	0.00	0.00	0.02	0.00	0.04	0.45	3.21	1.58	0.65	0.04	0.03
Keratella cochlearis	57.45	56.48	15.76	9.80	5.09	4.22	2.99	0.12	0.09	0.39	15.16	68.70	3.36	1.87	3.85	10.17
Keratella quadrata	10.10	4.65	0.46	0.18	0.01	0.00	0.04	0.00	0.16	0.42	2.72	5.16	0.06	0.00	0.00	0.00
Keratella serrulata	0.00	0.00	0.00	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.02	0.00
Lecane spp.	0.01	0.01	0.06	0.12	0.00	0.05	0.32	0.05	0.01	0.00	0.00	0.10	0.01	0.00	0.08	0.20
Lepadella ovalis	0.00	0.04	0.00	0.00	0.01	0.00	0.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Monostyla bulla	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Appendix Table A-26. Continued.

Taxon	Mean number/L															
	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Jan	Mar	Apr	May	Jun	Jul	Aug	Sep
Rotifera (continued)																
Mytlinia mucronatum	0.00	0.02	0.00	0.00	0.00	0.00	0.00	0.01	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Notholca acuminata	5.99	10.21	2.12	0.00	0.00	0.00	0.00	0.00	0.00	0.11	0.59	22.76	0.16	0.02	0.00	0.00
Philodina spp.	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.26	0.20
Platyias quadricornis	0.00	0.00	0.00	0.00	0.00	0.00	0.04	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Polyarthra vulgaris	34.32	19.68	10.90	15.88	10.88	51.47	96.54	0.71	0.04	0.04	17.38	26.63	5.26	5.92	24.60	37.30
Pompholyx sulcata	0.01	0.00	0.02	0.02	0.00	0.08	0.08	0.07	0.02	0.15	0.00	0.00	0.00	0.00	0.00	0.00
Synchaeta pectinata	4.17	0.66	11.37	49.22	32.05	1.09	0.19	0.72	0.47	1.94	15.72	15.29	7.97	25.96	66.72	44.21
Trichocerca spp.	0.00	0.19	0.26	3.62	4.65	4.36	23.14	0.06	0.00	0.00	0.05	0.05	0.22	2.01	2.66	6.82
Trichotria pocillum	0.01	0.00	0.03	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.03
Totals	129.80	94.35	48.36	85.93	55.39	65.94	130.69	2.99	0.84	3.22	54.03	144.55	19.40	37.41	104.20	111.43

Appendix Table A-27. Zooplankton densities at John Day Reservoir, Crow Butte backwater, March to September, 1995.

	Mean number/L						
Taxon	Mar	Apr	May	Jun	Jul	Aug	Sep
Cladocera							
Alona costata	0.00	0.00	0.07	0.00	0.36	0.00	0.00
Alona quadrangularis	0.00	0.00	0.00	0.18	0.00	0.00	0.00
Bosmina longirostris	16.02	114.10	115.13	7.38	1.11	0.00	9.20
Ceriodaphnia quadrangula	0.00	0.00	0.20	0.36	0.20	0.00	0.00
Chydorus sphaericus	6.01	1.94	0.92	0.42	0.00	0.00	0.46
Daphnia retrocurva	0.00	0.00	0.00	9.95	35.40	1.28	0.00
Daphnia thorata	1.00	2.64	26.04	6.79	0.83	0.00	0.00
Diaphanasoma birgeii	0.00	0.00	0.00	0.00	4.12	0.00	0.00
Kurzia latissima	0.00	0.00	2.38	0.00	0.00	0.00	0.00
Leptodora kindtii	0.00	0.00	0.07	0.18	0.93	0.00	0.50
Immature	3.00	3.03	6.44	0.71	3.78	0.00	0.00
Totals	26.03	121.71	151.25	25.97	46.73	1.28	10.16
Copepoda							
Cyclopoid copepodid	0.00	0.87	0.39	2.14	1.45	5.42	3.79
Diacyclops thomasi	0.00	6.07	2.44	1.90	3.65	1.32	1.36
Leptodiaptomus ashlandi	0.00	0.00	0.00	0.00	1.11	0.00	0.00
Nauplii	3.00	14.37	17.66	18.80	54.73	55.32	17.90
Totals	3.00	21.31	20.49	22.84	60.94	62.06	23.05
Rotifera							
Asplanchna girodi	2.00	23.17	18.59	2.32	0.00	0.36	43.92
Asplanchna sieboldi	8.01	0.00	0.21	0.00	0.00	0.00	0.00
Bdelloid	1.00	0.18	0.00	0.00	0.20	0.00	0.00
Brachionus angularis	0.00	0.00	0.00	0.54	5.97	17.90	4.90
Brachionus calyciflorus	21.03	9.41	0.55	0.12	0.00	0.00	1.12
Collotheca spp.	0.00	0.00	0.00	0.00	0.00	0.00	0.50
Colurella spp.	0.00	0.00	0.00	0.12	0.00	0.00	0.00
Conochiloides dossaurius	0.00	0.00	0.00	0.00	0.81	9.53	1.05
Conochilus unicornis	0.00	0.00	0.00	5.37	0.00	12.09	12.81
Euchlanis calpidia	0.00	0.27	0.00	0.12	0.15	0.00	0.00
Filinia longiseta	0.00	0.00	0.69	0.00	0.00	0.00	0.00
Gastropus hypotus	0.00	0.00	0.00	0.00	0.00	0.00	0.85
Gastropus stylifer	0.00	0.00	0.00	0.00	0.00	0.00	14.78
Kellicottia longispina	0.00	0.30	2.95	7.20	29.11	0.00	0.00
Keratella cochlearis	18.03	57.78	92.49	28.13	254.67	94.15	163.58
Keratella quadrata	4.01	4.78	2.54	0.00	0.00	0.00	0.00
Lecane spp.	0.00	0.00	0.00	0.12	0.84	1.70	0.96
Lepadella ovalis	0.00	0.00	0.00	0.12	0.00	0.00	0.00
Monostyla bulla	0.00	0.00	0.00	0.00	0.71	0.00	0.22
Notholca acuminata	3.00	0.67	1.93	0.00	0.09	0.00	0.00
Philodina spp.	0.00	0.00	0.00	0.00	0.00	0.21	0.46
Polyarthra vulgaris	13.02	68.85	23.24	34.67	91.06	293.26	288.82
Synchaeta pectinata	131.19	63.75	96.92	81.36	130.67	114.66	68.84
Trichocerca spp.	0.00	0.00	0.84	13.50	47.13	16.34	45.11
Trichotria pocillum	0.00	0.00	0.00	0.10	0.55	0.00	0.00
Totals	201.29	229.16	240.95	173.79	561.96	560.20	647.92

Appendix Table A-28.Zooplankton densities (organisms/L) at John Day Reservoir sampling stations, April 1994 to September 1995.

		Cladocerans		Copepods		Rotifers	
Station	Date	Mean[a]	SD[b]	Mean	SD	Mean	SD
Lower- and mid-reservoir stations							
LePage	4/11/94	[c]	[c]	0.5	0.3	57.7	3.8
Towal	4/11/94	0.0[d]	[e]	0.3	0.3	56.4	4.3
Arlington	4/11/94	[c]	[c]	0.5	0.1	31.7	3.1
LePage	5/10/94	0.3	0.1	2.9	1.1	144.6	41.7
Towal	5/10/94	[c]	[c]	2.6	0.7	131.9	15.6
Arlington	5/10/94	0.0[d]	[e]	1.5	0.5	102.1	3.5
LePage	6/7/94	2.1	0.7	12.9	3.2	252.0	28.3
Towal	6/7/94	0.5	0.2	4.5	2.3	144.9	6.0
Arlington	6/9/94	0.8	0.5	4.3	1.1	80.7	10.2
LePage	7/6/94	3.7	0.9	13.2	8.4	220.5	35.9
Towal	7/6/94	3.3	1.2	8.4	3.4	252.3	13.1
Arlington	7/6/94	6.6	1.1	10.5	1.5	348.6	28.8
LePage	8/2/94	23.5	1.4	14.4	2.6	13.9	1.8
Towal	8/2/94	38.9	8.7	18.7	2.7	20.7	2.7
Arlington	8/2/94	6.5	4.0	7.8	0.5	69.7	7.3
LePage	9/15/94	1.6	0.6	13.0	1.3	15.3	1.1
Towal	9/15/94	3.9	0.1	18.1	2.3	19.7	2.0
Arlington	9/15/94	6.3	2.0	11.5	0.8	56.2	11.2
LePage	10/4/94	2.7	0.9	32.4	2.3	21.5	4.6
Towal	10/4/94	3.1	1.9	30.7	2.8	27.1	4.0
Arlington	10/4/94	19.2	6.7	18.5	5.9	103.3	21.9
LePage	11/15/94	0.4	0.1	2.5	0.5	9.8	2.5
Towal	11/15/94	2.2	1.0	2.8	1.3	10.0	2.3
Arlington	11/15/94	0.4	0.2	1.0	0.1	8.2	2.1
LePage	1/10/95	0.2	0.0	0.6	0.3	4.7	0.4
Towal	1/10/95	0.1	0.0	0.5	0.3	3.3	1.2
Arlington	1/10/95	0.3	0.2	0.4	0.0	3.1	0.9
LePage	3/13/95	0.2	[e]	0.2	[e]	6.2	3.4
Towal	3/13/95	0.2	[e]	[c]	[c]	7.6	3.8
Arlington	3/13/95	[c]	[c]	0.6	0.2	8.5	0.2
LePage	4/21/95	0.2	[e]	0.7	[e]	27.3	8.6
Towal	4/21/95	[c]	[c]	1.7	1.9	56.6	28.5
Arlington	4/21/95	0.7	0.1	3.0	1.5	163.2	26.6
LePage	5/15/95	[c]	[c]	0.5	[e]	50.1	21.7

Appendix Table A-28. Continued.

Station	Date	Cladocerans Mean	Cladocerans SD	Copepods Mean	Copepods SD	Rotifers Mean	Rotifers SD
Lower- and mid-reservoir stations (continued)							
Towal	5/15/95	3.4	1.2	4.8	1.9	258.6	178.3
Arlington	5/15/95	[c]	[c]	[c]	[c]	141.9	101.6
LePage	6/12/95	1.2	1.3	2.2	2.2	22.0	25.9
Towal	6/12/95	1.8	0.6	2.8	2.1	37.4	7.1
Arlington	6/12/95	0.6	0.4	2.1	0.7	19.0	10.3
LePage	7/10/95	2.6	0.4	4.6	1.9	145.3	19.6
Towal	7/10/95	2.1	1.2	4.1	1.1	108.2	26.0
Arlington	7/10/95	1.5	0.6	4.0	0.4	108.7	16.9
LePage	8/10/95	93.4	59.0	140.2	80.5	155.5	88.6
Towal	8/10/95	32.5	22.2	84.2	42.0	147.6	67.8
Arlington	8/10/95	10.1	0.7	10.4	1.9	154.4	34.1
LePage	9/8/95	5.1	3.4	17.9	4.7	381.0	21.6
Towal	9/8/95	0.6	[e]	13.5	10.4	271.4	108.0
Arlington	9/8/95	2.3	0.9	6.3	2.5	601.8	90.6
Upper-reservoir and backwater stations							
Sand Is.	4/13/94	0.3	0.2	2.8	2.1	90.9	4.1
Long Walk dn.	4/13/94	0.2	[e]	1.1	0.6	58.7	4.3
Big Blalock Is.	4/13/94	[c]	[c]	1.1	0.3	69.0	10.4
Long Walk up.	4/13/94	0.3	0.1	1.2	0.5	83.3	9.2
Paterson Slough	4/13/94	0.3	0.2	2.5	0.6	47.6	6.3
Sand Is.	4/26/94	0.2	[e]	5.6	2.5	232.8	31.1
Long Walk dn.	4/26/94	0.3	0.0	3.7	1.9	146.6	10.9
Big Blalock Is.	4/26/94	0.4	0.0	6.1	1.9	281.1	15.5
Long Walk up.	4/26/94	0.1	[e]	6.5	2.8	150.8	4.2
Paterson Slough	4/26/94	0.3	0.0	4.6	0.4	137.5	35.8
Sand Is.	5/11/94	0.2	[e]	6.3	2.7	149.8	7.3
Long Walk dn.	5/11/94	0.3	[e]	8.8	1.9	93.9	7.3
Big Blalock Is.	5/11/94	0.2	[e]	4.9	0.7	151.4	6.2
Long Walk up.	5/11/94	0.1	[e]	9.3	1.4	119.5	9.0
Paterson Slough	5/11/94	0.1	[e]	5.6	0.4	121.9	10.2
Sand Is.	5/24/94	0.6	0.3	2.2	0.8	76.8	9.9
Long Walk dn.	5/24/94	0.6	0.8	3.5	1.3	38.7	6.0
Big Blalock Is.	5/24/94	0.3	[e]	2.8	1.5	90.5	28.1

Appendix Table A-28. Continued.

Station	Date	Cladocerans Mean	Cladocerans SD	Copepods Mean	Copepods SD	Rotifers Mean	Rotifers SD
Upper-reservoir and backwater stations (continued)							
Long Walk up.	5/24/94	0.1	[e]	3.6	0.8	47.6	13.5
Paterson Slough	5/24/94	0.6	0.1	2.3	1.6	53.2	3.7
Sand Is.	6/8/94	1.5	1.0	4.8	1.4	48.4	5.3
Long Walk dn.	6/8/94	0.6	0.1	5.7	5.8	22.5	11.4
Big Blalock Is.	6/8/94	1.5	0.4	5.3	3.0	69.4	14.4
Long Walk up.	6/8/94	0.2	[e]	8.0	2.0	51.5	10.4
Paterson Slough	6/8/94	3.6	1.0	5.5	1.1	35.9	8.9
Sand Is.	6/21/94	1.0	1.1	8.5	1.1	67.4	9.8
Long Walk dn.	6/21/94	0.1	[e]	3.6	1.5	34.9	10.5
Big Blalock Is.	6/21/94	0.8	0.7	6.8	0.5	54.2	7.5
Long Walk up.	6/21/94	0.7	0.9	7.5	1.4	46.0	5.6
Paterson Slough	6/21/94	0.8	0.4	2.8	2.1	59.8	11.9
Sand Is.	7/7/94	4.0	3.5	7.3	2.9	89.8	0.9
Long Walk dn.	7/7/94	5.1	2.2	4.2	2.2	45.8	11.1
Big Blalock Is.	7/7/94	2.5	1.5	3.5	1.1	82.2	6.6
Long Walk up.	7/7/94	2.0	2.0	4.9	1.9	50.3	4.9
Paterson Slough	7/7/94	3.2	2.3	5.7	4.6	95.9	4.4
Sand Is.	7/20/94	1.9	0.5	5.7	1.7	95.5	7.3
Long Walk dn.	7/20/94	2.4	2.1	6.3	1.7	117.9	9.3
Big Blalock Is.	7/20/94	1.5	1.1	5.0	2.0	96.6	7.3
Long Walk up.	7/20/94	0.9	0.1	3.7	2.7	98.6	5.2
Paterson Slough	7/20/94	1.6	0.8	3.6	1.5	86.8	13.4
Sand Is.	8/4/94	2.7	1.1	6.3	1.6	69.1	11.7
Long Walk dn.	8/4/94	2.1	0.6	11.0	2.6	74.5	4.8
Big Blalock Is.	8/4/94	4.8	0.8	6.0	1.0	54.9	3.5
Long Walk up.	8/4/94	2.1	1.0	5.9	2.3	58.3	1.1
Paterson Slough	8/4/94	1.6	0.9	3.3	0.4	27.5	6.1
Sand Is.	8/18/94	9.3	0.4	12.4	4.8	65.7	13.4
Long Walk dn.	8/18/94	2.9	2.2	7.6	2.0	30.1	0.8
Big Blalock Is.	8/18/94	8.7	0.4	11.6	3.0	78.2	5.5
Long Walk up.	8/18/94	5.6	2.0	7.6	2.6	54.3	4.5
Paterson Slough	8/18/94	2.8	1.5	7.5	2.3	41.3	2.9
Sand Is.	9/14/94	3.9	2.0	7.3	1.5	34.4	4.3
Long Walk dn.	9/14/94	5.5	0.4	7.6	1.6	31.8	3.2
Big Blalock Is.	9/14/94	1.6	0.0	9.7	5.2	76.4	7.4

Appendix Table A-28. Continued.

Station	Date	Cladocerans Mean	Cladocerans SD	Copepods Mean	Copepods SD	Rotifers Mean	Rotifers SD
Upper-reservoir and backwater stations (continued)							
Long Walk up.	9/14/94	5.9	2.2	11.5	1.7	97.0	13.4
Paterson Slough	9/14/94	11.8	2.8	10.2	2.0	90.2	8.2
Sand Is.	10/6/94	12.0	3.5	15.0	5.1	118.0	2.7
Long Walk dn.	10/6/94	15.3	2.8	14.9	0.2	75.4	2.0
Big Blalock Is.	10/6/94	23.3	1.5	12.5	2.7	137.1	11.4
Long Walk up.	10/6/94	19.9	1.0	16.3	3.2	139.5	13.1
Paterson Slough	10/6/94	24.1	8.5	8.8	4.4	183.4	4.4
Sand Is.	11/17/94	0.7	0.6	0.5	0.3	2.4	0.6
Long Walk dn.	11/17/94	0.6	0.2	0.4	0.2	2.2	0.8
Big Blalock Is.	11/17/94	0.4	0.3	0.4	0.2	2.0	0.4
Long Walk up.	11/17/94	0.4	0.1	0.9	0.3	4.8	0.7
Paterson Slough	11/17/94	0.5	0.2	0.6	0.2	3.6	1.1
Sand Is.	1/25/95	0.5	0.3	0.3	0.2	0.7	0.1
Long Walk dn.	1/25/95	0.1	[e]	0.3	0.3	0.8	0.2
Big Blalock Is.	1/25/95	0.7	0.6	0.5	0.0	1.0	0.4
Long Walk up.	1/25/95	0.3	0.3	[c]	[c]	0.2	0.3
Paterson Slough	1/25/95	5.1	3.2	1.2	0.3	1.7	0.8
Crow Butte	3/15/95	26.0	[e]	3.0	[e]	201.3	[e]
Sand Is.	3/15/95	0.4	0.1	0.7	0.4	3.8	2.4
Long Walk dn.	3/15/95	0.2	[e]	1.0	0.4	2.6	0.7
Big Blalock Is.	3/15/95	0.2	0.0	0.6	0.5	2.5	1.0
Long Walk up.	3/15/95	0.4	0.2	0.7	0.7	3.1	1.5
Paterson Slough	3/15/95	0.1	[e]	0.2	[e]	5.0	0.3
Crow Butte	4/4/95	26.3	7.2	8.2	0.5	153.9	6.2
Sand Is.	4/4/95	0.0[d]	[e]	0.3	0.1	4.5	2.6
Long Walk dn.	4/4/95	[c]	[c]	0.1	0.0	3.0	1.8
Big Blalock Is.	4/4/95	0.2	[e]	0.9	0.4	11.4	5.1
Long Walk up.	4/4/95	0.2	[e]	0.7	0.1	5.3	2.1
Paterson Slough	4/4/95	0.1	[e]	0.3	0.1	10.1	2.4
Crow Butte	4/18/95	217.1	43.5	34.5	8.4	304.4	32.7
Sand Is.	4/20/95	1.0	[e]	2.6	1.2	108.0	12.5
Long Walk dn.	4/20/95	0.8	0.3	0.9	0.5	65.2	18.0
Big Blalock Is.	4/20/95	[c]	[c]	0.9	0.5	121.2	9.3
Long Walk up.	4/20/95	[c]	[c]	2.6	0.4	108.8	4.6
Paterson Slough	4/20/95	[c]	[c]	1.3	0.8	103.9	10.8

Appendix Table A-28. Continued.

Station	Date	Cladocerans Mean	Cladocerans SD	Copepods Mean	Copepods SD	Rotifers Mean	Rotifers SD
Upper-reservoir and backwater stations (continued)							
Crow Butte	5/1/95	174.7	43.2	20.1	6.4	234.7	60.4
Paterson Slough	5/2/95	3.7	1.3	6.3	1.5	128.3	10.8
Sand Is.	5/3/95	3.0	1.4	2.0	0.2	228.3	21.1
Long Walk dn.	5/3/95	[c]	[c]	1.0	[e]	153.5	44.4
Big Blalock Is.	5/3/95	[c]	[c]	3.8	3.5	251.5	11.4
Long Walk up.	5/3/95	4.0	3.9	4.0	3.9	202.9	12.7
Crow Butte	5/17/95	98.7	57.2	12.1	6.2	73.2	49.2
Sand Is.	5/17/95	1.5	0.7	5.0	1.7	324.8	70.5
Long Walk dn.	5/17/95	1.0	[e]	6.5	2.2	133.1	21.3
Big Blalock Is.	5/17/95	3.1	5.3	5.5	3.2	167.2	73.2
Long Walk up.	5/17/95	0.6	[e]	3.2	1.2	111.9	27.2
Paterson Slough	5/17/95	2.2	3.5	2.3	2.9	118.8	70.7
Crow Butte	5/30/95	180.3	76.2	29.3	15.6	414.8	157.2
Sand Is.	5/31/95	0.9	0.5	0.9	0.2	57.9	12.5
Long Walk dn.	5/31/95	2.3	2.6	6.3	3.0	113.9	12.3
Big Blalock Is.	5/31/95	1.1	0.5	1.9	0.1	71.8	1.9
Long Walk up.	5/31/95	0.2	[e]	1.3	1.1	47.0	6.4
Paterson Slough	5/31/95	[c]	[c]	0.6	0.0	60.5	8.8
Sand Is.	6/14/95	0.1	[e]	0.6	0.9	4.3	0.8
Long Walk dn.	6/14/95	[c]	[c]	2.2	2.3	18.7	3.0
Big Blalock Is.	6/14/95	0.2	[e]	1.5	0.4	8.1	2.9
Long Walk up.	6/14/95	[c]	[c]	2.1	0.6	8.2	4.0
Paterson Slough	6/14/95	[c]	[c]	0.9	0.4	8.7	5.0
Crow Butte	6/15/95	34.0	15.3	19.2	12.8	63.0	22.1
Crow Butte	6/28/95	18.0	4.8	26.5	5.3	284.6	38.7
Sand Is.	6/29/95	1.4	0.7	2.2	0.6	28.5	3.8
Long Walk dn.	6/29/95	0.3	[e]	1.9	1.3	22.8	0.9
Big Blalock Is.	6/29/95	0.4	0.2	1.5	0.6	29.5	10.3
Long Walk up.	6/29/95	0.7	0.5	2.6	0.7	32.3	7.6
Paterson Slough	6/29/95	0.4	0.2	2.6	1.4	33.4	9.2
Sand Is.	7/12/95	0.1	[e]	1.6	0.4	18.4	7.8
Long Walk dn.	7/12/95	0.3	0.0	2.0	1.5	17.1	10.3
Big Blalock Is.	7/12/95	0.2	[e]	1.5	0.2	21.0	14.7
Long Walk up.	7/12/95	1.0	0.5	2.5	1.3	65.4	31.0
Paterson Slough	7/12/95	0.5	0.6	1.1	0.7	17.8	7.1

Appendix Table A-28. Continued.

Station	Date	Cladocerans Mean	Cladocerans SD	Copepods Mean	Copepods SD	Rotifers Mean	Rotifers SD
Upper-reservoir and backwater stations (continued)							
Crow Butte	7/13/95	64.2	42.3	96.3	112.0	926.9	980.3
Crow Butte	7/24/95	29.2	27.0	25.5	18.3	197.0	124.0
Sand Is.	7/25/95	0.6	0.7	1.9	1.4	34.0	9.9
Long Walk dn.	7/25/95	0.7	0.2	1.8	0.2	65.4	5.1
Big Blalock Is.	7/25/95	0.5	0.6	2.6	1.8	43.3	22.7
Long Walk up.	7/25/95	0.1	[e]	3.1	1.3	71.0	15.2
Paterson Slough	7/25/95	[c]	[c]	1.1	0.8	20.8	11.5
Crow Butte	8/8/95	1.5	[e]	108.8	44.1	864.6	170.5
Sand Is.	8/8/95	0.9	0.3	6.1	0.6	89.7	15.7
Long Walk dn.	8/8/95	1.2	0.6	4.7	0.7	137.1	13.5
Big Blalock Is.	8/8/95	1.6	0.2	3.6	0.4	81.4	9.9
Long Walk up.	8/8/95	1.2	1.7	4.1	2.6	112.8	7.4
Paterson Slough	8/8/95	3.8	2.9	3.4	1.5	91.2	42.5
Crow Butte	8/21/95	1.1	[e]	15.3	10.7	255.8	160.7
Sand Is.	8/22/95	[c]	[c]	0.6	[e]	102.5	7.3
Long Walk dn.	8/22/95	[c]	[c]	0.8	[e]	83.8	12.6
Big Blalock Is.	8/22/95	0.5	[e]	1.5	1.2	99.6	28.8
Long Walk up.	8/22/95	2.0	1.2	5.2	3.2	189.0	31.7
Paterson Slough	8/22/95	[c]	[c]	2.2	0.4	55.1	6.8
Crow Butte	9/5/95	3.4	1.9	18.6	10.5	609.7	288.2
Sand Is.	9/7/95	1.1	[e]	1.1	[e]	166.8	14.7
Long Walk dn.	9/7/95	1.1	[e]	3.1	2.4	86.6	15.2
Big Blalock Is.	9/7/95	0.9	[e]	0.9	[e]	131.0	15.5
Long Walk up.	9/7/95	[c]	[c]	3.3	2.8	147.2	11.1
Paterson Slough	9/7/95	1.2	[e]	[c]	[c]	147.7	23.7
Crow Butte	9/19/95	16.9	9.9	27.5	16.0	686.1	417.0
Sand Is.	9/19/95	0.9	0.7	5.7	1.7	104.6	5.9
Long Walk dn.	9/19/95	1.2	0.9	6.9	2.0	92.1	18.2
Big Blalock Is.	9/19/95	0.3	[e]	3.9	0.7	61.4	10.9
Long Walk up.	9/19/95	0.9	0.4	4.9	1.5	84.3	44.3
Paterson Slough	9/19/95	0.4	[e]	3.3	2.8	92.5	12.0

[a] Means of three samples for all date/station strata except Crow Butte on 3/15/95 (one sample).
[b] Standard deviation.
[c] Taxon not present.
[d] Mean density less than 0.1 organism/L.
[e] Taxon present in one sample only.

Appendix Table A-29.John Day Reservoir benthic invertebrate collections, April 1994 to September 1995.

Station	Depth (m)	Date	Samples (no.)	Taxa (no.)	Density (no./m^2)	Standard deviation
Long Walk Island (downstream)	1	4/12/94	4	10	1,500	1,785
Long Walk Island (downstream)	3	4/12/94	4	15	15,355	3,438
Long Walk Island (downstream)	5	4/12/94	4	12	11,240	1,701
Long Walk Island (upstream)	1	4/12/94	4	17	14,010	4,813
Long Walk Island (upstream)	3	4/12/94	4	10	10,175	5,141
Long Walk Island (upstream)	5	4/12/94	4	10	2,555	606
Paterson Slough	1	4/12/94	2	14	2,380	1,075
Paterson Slough	3	4/12/94	2	9	1,880	481
Paterson Slough	5	4/12/94	2	10	2,470	127
Big Blalock Island	1	4/13/94	2	11	9,940	170
Big Blalock Island	3	4/13/94	2	13	2,920	141
Big Blalock Island	5	4/13/94	2	12	3,070	1,626
Sand Island	1	4/13/94	2	10	7,690	721
Sand Island	3	4/13/94	2	13	4,020	651
Sand Island	5	4/13/94	2	7	2,390	2,107
Big Blalock Island	1	5/12/94	4	10	6,315	4,217
Big Blalock Island	3	5/12/94	4	8	1,575	1,407
Big Blalock Island	5	5/12/94	4	9	665	471
Long Walk Island (downstream)	1	5/12/94	4	12	3,005	3,943
Long Walk Island (downstream)	3	5/12/94	4	18	13,870	2,849
Long Walk Island (downstream)	5	5/12/94	4	18	10,410	3,612
Long Walk Island (upstream)	1	5/12/94	4	10	4,875	1,006
Long Walk Island (upstream)	3	5/12/94	4	10	15,430	10,936
Long Walk Island (upstream)	5	5/12/94	4	14	3,535	3,853
Paterson Slough	1	5/12/94	4	12	9,780	5,063
Paterson Slough	3	5/12/94	4	12	2,050	944
Paterson Slough	5	5/12/94	4	12	2,330	555
Sand Island	1	5/12/94	4	9	5,030	2,045
Sand Island	3	5/12/94	4	9	2,090	1,228
Sand Island	5	5/12/94	4	7	1,010	296
Long Walk Island (downstream)	1	6/8/94	4	13	2,800	1,300
Long Walk Island (downstream)	3	6/8/94	4	16	14,155	3,610
Long Walk Island (downstream)	5	6/8/94	4	15	12,405	2,885
Sand Island	1	6/8/94	4	11	5,275	514
Sand Island	3	6/8/94	4	10	6,430	1,898
Sand Island	5	6/8/94	4	10	3,215	2,065

Appendix Table A-29. Continued.

Station	Depth (m)	Date	Samples (no.)	Taxa (no.)	Density (no./m^2)	Standard deviation
Big Blalock Island	1	6/9/94	4	10	8,750	4,787
Big Blalock Island	3	6/9/94	4	9	1,460	1,389
Big Blalock Island	5	6/9/94	4	12	4,570	2,237
Long Walk Island (upstream)	1	6/9/94	4	16	17,895	2,625
Long Walk Island (upstream)	3	6/9/94	4	16	20,525	5,305
Long Walk Island (upstream)	5	6/9/94	4	11	3,395	1,782
Paterson Slough	1	6/9/94	4	12	2,810	1,176
Paterson Slough	3	6/9/94	4	9	685	610
Paterson Slough	5	6/9/94	4	9	990	208
Big Blalock Island	1	7/7/94	4	13	5,305	2,609
Big Blalock Island	3	7/7/94	4	10	4,545	2,104
Big Blalock Island	5	7/7/94	4	8	3,505	4,320
Long Walk Island (downstream)	1	7/7/94	4	12	1,385	320
Long Walk Island (downstream)	3	7/7/94	4	9	4,205	2,654
Long Walk Island (downstream)	5	7/7/94	4	11	2,965	1,637
Long Walk Island (upstream)	1	7/7/94	4	20	7,465	13,370
Long Walk Island (upstream)	3	7/7/94	4	14	11,430	7,017
Long Walk Island (upstream)	5	7/7/94	4	9	4,240	2,395
Paterson Slough	1	7/7/94	4	19	5,435	264
Paterson Slough	3	7/7/94	4	10	2,385	1,431
Paterson Slough	5	7/7/94	4	8	1,110	714
Sand Island	1	7/7/94	4	16	6,290	4,861
Sand Island	3	7/7/94	4	10	2,320	1,178
Sand Island	5	7/7/94	4	8	1,425	1,650
Big Blalock Island	1	8/3/94	4	8	3,740	2,448
Big Blalock Island	3	8/3/94	4	10	2,825	1,464
Big Blalock Island	5	8/3/94	4	6	2,215	2,619
Long Walk Island (downstream)	1	8/3/94	4	10	2,720	1,406
Long Walk Island (downstream)	3	8/3/94	4	12	11,625	12,016
Long Walk Island (downstream)	5	8/3/94	4	13	8,155	8,245
Long Walk Island (upstream)	1	8/3/94	4	14	23,645	6,280
Long Walk Island (upstream)	3	8/3/94	4	8	4,735	1,031
Long Walk Island (upstream)	5	8/3/94	4	11	4,745	3,370
Paterson Slough	1	8/3/94	4	11	6,850	7,100
Paterson Slough	3	8/3/94	4	14	3,190	2,707
Paterson Slough	5	8/3/94	4	3	475	214
Sand Island	1	8/3/94	4	13	5,340	388

Appendix Table A-29. Continued.

Station	Depth (m)	Date	Samples (no.)	Taxa (no.)	Density (no./m^2)	Standard deviation
Sand Island	3	8/3/94	4	12	11,055	13,497
Sand Island	5	8/3/94	4	9	1,690	2,094
Big Blalock Island	1	9/13/94	4	11	8,655	4,951
Big Blalock Island	3	9/13/94	4	13	5,615	3,389
Big Blalock Island	5	9/13/94	4	10	2,285	185
Long Walk Island (downstream)	1	9/13/94	4	12	3,380	830
Long Walk Island (downstream)	3	9/13/94	4	12	25,140	6,726
Long Walk Island (downstream)	5	9/13/94	4	11	24,410	5,778
Long Walk Island (upstream)	1	9/13/94	4	15	27,625	8,326
Long Walk Island (upstream)	3	9/13/94	4	15	22,290	7,596
Long Walk Island (upstream)	5	9/13/94	4	14	12,830	8,256
Paterson Slough	1	9/13/94	4	15	14,340	3,450
Paterson Slough	3	9/13/94	4	13	2,515	556
Paterson Slough	5	9/13/94	4	12	3,405	470
Sand Island	1	9/13/94	4	4	525	226
Sand Island	3	9/13/94	4	10	5,950	1,643
Sand Island	5	9/13/94	4	10	2,245	1,281
Big Blalock Island	1	10/5/94	2	12	4,780	2,461
Big Blalock Island	3	10/5/94	2	8	3,760	2,206
Big Blalock Island	5	10/5/94	2	4	120	113
Long Walk Island (downstream)	1	10/5/94	2	5	4,710	4,822
Long Walk Island (downstream)	3	10/5/94	2	9	12,980	3,705
Long Walk Island (downstream)	5	10/5/94	2	10	15,930	693
Long Walk Island (upstream)	1	10/5/94	2	9	7,170	6,208
Long Walk Island (upstream)	3	10/5/94	2	11	10,290	8,867
Long Walk Island (upstream)	5	10/5/94	2	11	6,000	7,128
Paterson Slough	1	10/5/94	2	11	11,640	396
Paterson Slough	3	10/5/94	2	8	2,590	2,645
Paterson Slough	5	10/5/94	2	11	2,460	85
Sand Island	1	10/5/94	2	3	230	71
Sand Island	3	10/5/94	2	12	1,980	537
Sand Island	5	10/5/94	2	11	2,120	226
Long Walk Island (downstream)	1	11/17/94	4	7	1,290	425
Long Walk Island (downstream)	3	11/17/94	4	9	11,190	2,102
Long Walk Island (downstream)	5	11/17/94	4	10	14,810	2,528
Sand Island	1	11/17/94	4	11	7,425	1,961
Sand Island	3	11/17/94	4	11	4,845	2,137

Appendix Table A-29. Continued.

Station	Depth (m)	Date	Samples (no.)	Taxa (no.)	Density (no./m^2)	Standard deviation
Sand Island	5	11/17/94	4	10	2,150	1,081
Big Blalock Island	1	11/18/94	4	14	11,750	10,689
Big Blalock Island	3	11/18/94	4	16	3,130	634
Big Blalock Island	5	11/18/94	4	11	2,095	664
Long Walk Island (upstream)	1	11/18/94	4	12	6,685	2,458
Long Walk Island (upstream)	3	11/18/94	4	11	13,650	3,738
Long Walk Island (upstream)	5	11/18/94	4	12	5,035	2,290
Paterson Slough	1	11/18/94	4	9	1,850	220
Paterson Slough	3	11/18/94	4	12	3,925	1,291
Paterson Slough	5	11/18/94	3	9	3,713	722
Big Blalock Island	1	1/24/95	3	11	8,167	5,876
Big Blalock Island	5	1/24/95	1	5	1,840	
Long Walk Island (downstream)	1	1/24/95	4	14	1,885	352
Long Walk Island (downstream)	3	1/24/95	4	12	12,380	8,443
Long Walk Island (downstream)	5	1/24/95	4	14	15,615	2,562
Long Walk Island (upstream)	1	1/24/95	4	11	13,260	6,117
Long Walk Island (upstream)	3	1/24/95	4	8	5,160	3,789
Long Walk Island (upstream)	5	1/24/95	4	15	4,690	3,975
Paterson Slough	1	1/24/95	4	9	10,280	3,369
Paterson Slough	3	1/24/95	4	11	2,730	706
Paterson Slough	5	1/24/95	4	11	1,675	1,637
Sand Island	1	1/24/95	4	12	8,305	902
Sand Island	3	1/24/95	3	11	3,895	2,708
Sand Island	5	1/24/95	4	8	1,345	682
Big Blalock Island	1	3/14/95	1	6	9,040	
Big Blalock Island	5	3/14/95	1	5	620	
Long Walk Island (downstream)	1	3/14/95	4	9	855	387
Long Walk Island (downstream)	3	3/14/95	4	9	3,530	2,693
Long Walk Island (downstream)	5	3/14/95	4	9	8,955	4,615
Long Walk Island (upstream)	1	3/14/95	4	5	2,505	1,146
Long Walk Island (upstream)	3	3/14/95	4	12	6,860	5,321
Long Walk Island (upstream)	5	3/14/95	4	10	6,630	6,415
Paterson Slough	1	3/14/95	4	15	4,900	2,776
Paterson Slough	3	3/14/95	4	12	4,840	2,334
Paterson Slough	5	3/14/95	4	8	1,465	872
Sand Island	1	3/14/95	4	7	480	668
Sand Island	3	3/14/95	4	10	2,100	1,297

Appendix Table A-29. Continued.

Station	Depth (m)	Date	Samples (no.)	Taxa (no.)	Density (no./m2)	Standard deviation
Sand Island	5	3/14/95	4	10	900	719
Crow Butte	3	4/18/95	1	10	6,680	
Crow Butte	5	4/18/95	3	13	11,387	1,399
Big Blalock Island	1	4/19/95	3	9	4,980	1,611
Big Blalock Island	3	4/19/95	2	8	1,790	410
Long Walk Island (downstream)	1	4/19/95	4	9	565	353
Long Walk Island (downstream)	3	4/19/95	2	13	9,000	8,004
Long Walk Island (upstream)	3	4/19/95	2	11	6,260	4,497
Long Walk Island (upstream)	5	4/19/95	4	14	3,705	4,101
Paterson Slough	1	4/19/95	4	8	1,200	402
Paterson Slough	3	4/19/95	4	7	810	571
Paterson Slough	5	4/19/95	4	9	1,160	604
Sand Island	1	4/19/95	4	5	655	727
Sand Island	3	4/19/95	4	9	650	194
Sand Island	5	4/19/95	4	10	1,240	1,088
Big Blalock Island	5	5/16/95	4	8	990	596
Long Walk Island (downstream)	1	5/16/95	4	8	685	104
Long Walk Island (downstream)	3	5/16/95	4	14	6,635	4,480
Long Walk Island (downstream)	5	5/16/95	4	15	5,975	3,111
Paterson Slough	1	5/16/95	4	8	1,255	531
Paterson Slough	3	5/16/95	4	11	1,275	390
Paterson Slough	5	5/16/95	4	12	955	588
Sand Island	1	5/16/95	4	5	250	99
Sand Island	3	5/16/95	4	11	1,450	1,405
Sand Island	5	5/16/95	4	10	1,135	541
Crow Butte	1	5/30/95	4	18	9,750	4,526
Crow Butte	3	5/30/95	2	13	7,200	3,988
Long Walk Island (upstream)	1	6/13/95	3	14	12,273	1,134
Long Walk Island (upstream)	3	6/13/95	3	10	10,893	3,442
Long Walk Island (upstream)	5	6/13/95	4	8	3,515	1,600
Paterson Slough	1	6/13/95	4	10	1,125	171
Paterson Slough	3	6/13/95	4	8	2,010	1,945
Paterson Slough	5	6/13/95	4	12	8,690	4,759
Sand Island	1	6/13/95	4	9	675	467
Sand Island	3	6/13/95	2	9	6,270	71
Crow Butte	1	6/15/95	3	19	32,480	4,787
Crow Butte	3	6/15/95	2	11	5,640	651

Appendix Table A-29. Continued.

Station	Depth (m)	Date	Samples (no.)	Taxa (no.)	Density (no./m2)	Standard deviation
Long Walk Island (downstream)	1	7/11/95	4	15	4,160	1,024
Long Walk Island (downstream)	3	7/11/95	4	15	7,075	5,667
Long Walk Island (downstream)	5	7/11/95	4	12	6,185	1,843
Long Walk Island (upstream)	1	7/11/95	2	9	2,840	537
Long Walk Island (upstream)	3	7/11/95	4	9	6,490	3,392
Long Walk Island (upstream)	5	7/11/95	4	6	2,235	2,285
Paterson Slough	1	7/11/95	4	14	2,885	714
Paterson Slough	3	7/11/95	4	13	3,985	3,131
Paterson Slough	5	7/11/95	4	12	4,120	2,403
Sand Island	1	7/11/95	4	13	1,950	452
Sand Island	3	7/11/95	4	8	265	200
Sand Island	5	7/11/95	4	4	130	62
Crow Butte	1	7/13/95	1	11	15,700	
Crow Butte	1	8/8/95	4	23	14,120	7,508
Crow Butte	3	8/8/95	2	11	5,770	4,342
Long Walk Island (downstream)	1	8/9/95	4	10	2,505	390
Long Walk Island (downstream)	3	8/9/95	4	14	8,425	6,229
Long Walk Island (downstream)	5	8/9/95	3	12	9,440	6,632
Long Walk Island (upstream)	1	8/9/95	4	8	2,855	1,357
Long Walk Island (upstream)	3	8/9/95	2	12	7,510	3,719
Paterson Slough	1	8/9/95	4	17	4,900	2,064
Paterson Slough	3	8/9/95	4	12	4,595	2,715
Paterson Slough	5	8/9/95	4	12	5,370	1,600
Sand Island	1	8/9/95	4	12	4,305	2,303
Sand Island	3	8/9/95	4	13	2,800	409
Sand Island	5	8/9/95	4	9	470	181
Long Walk Island (downstream)	1	9/6/95	4	11	3,260	1,581
Long Walk Island (downstream)	3	9/6/95	4	15	16,165	6,407
Long Walk Island (downstream)	5	9/6/95	4	13	8,120	2,648
Sand Island	3	9/6/95	2	11	5,510	71
Sand Island	5	9/6/95	4	9	3,375	896

Appendix Table A-30. Invertebrate taxa identified in John Day Reservoir benthic samples, April 1994 to September 1995.

Taxon	Identified
Phylum Annelida	
Hirudinea	x
Oligochaeta	x
Polychaeta	x
Nereidae	
Neanthes limnicola	x
Phylum Arthropoda	
Arachnida	
Ixodides	x
Prostigmata	x
Araneae	x
Crustacea	
Amphipoda	
Corophidae	
Corophium spp.	x
Corophium salmonis	x
Corophium spinicorne	x
Gammaridae	
Ramellogammarus spp.	x
Ramellogammarus oregonensis	x
Hyalellidae	
Hyalella azteca	x
Isopoda	x
Mysidacea	x
Cladocera	x
Leptodoridae	x
Leptodora kindtii	x
Copepoda	x
Calanoida	x
Cyclopoida	x
Harpacticoida	x
Ostracoda	x

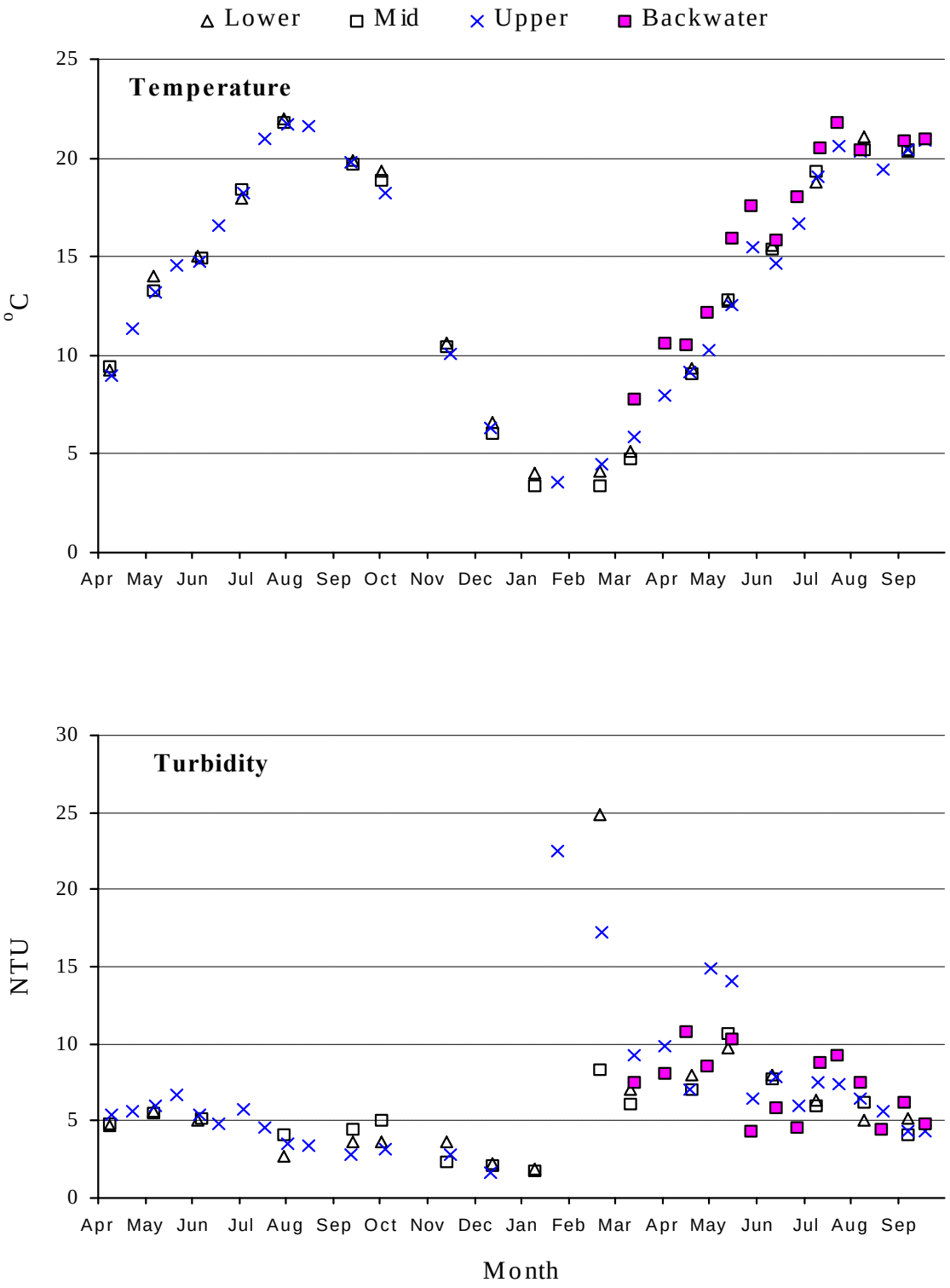
Appendix Table A-30. Continued.

Taxon	Identified	
Phylum Arthropoda (continued)		
Insecta	x	
Coleoptera	x	
Collembola	x	
Diptera		x
Ceratopogonidae	x	
Culicoides spp.	x	
Chironomidae	x	
Diamesinae	x	
Orthocladiinae	x	
Tanypodinae	x	
Psychodidae	x	
Simuliidae	x	
Tipulidae	x	
Ephemeroptera	x	
Caenidae	x	
Caenis spp.	x	
Ephemerellidae		
Ephemerella spp.	x	
Ephemeridae		
Hexagenia spp.	x	
Leptophlebiidae		
Choroterpes spp.	x	
Tricorythidae		
Tricorythodes spp.	x	
Hemiptera	x	
Homoptera	x	
Lepidoptera	x	
Odonata		
Anisoptera	x	
Gomphidae	x	
Octogomphus specularis	x	
Zygoptera	x	
Coenagrionidae	x	
Thysanoptera	x	
Trichoptera	x	

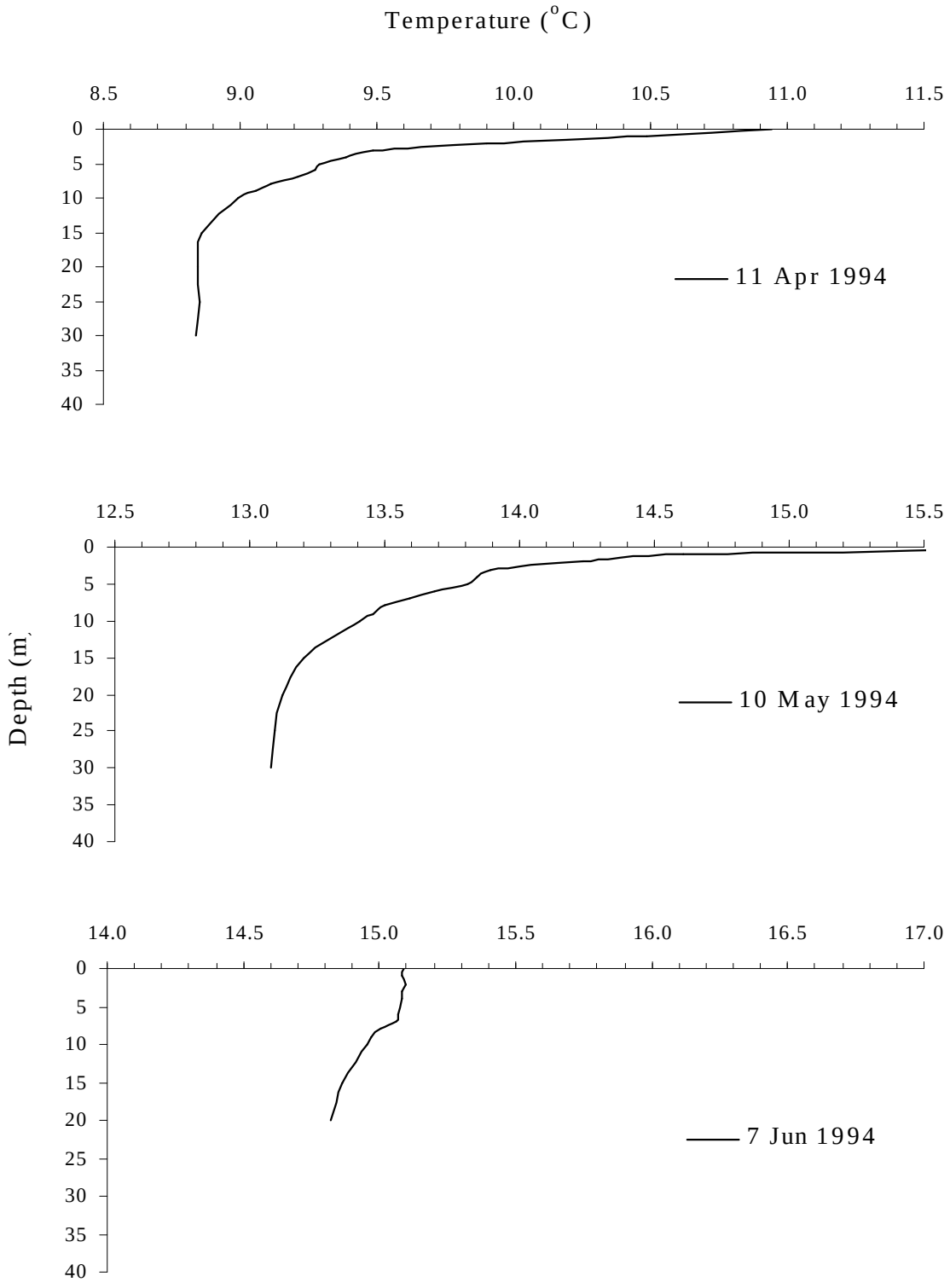
Appendix Table A-30. Continued.

Taxon	Identified
Phylum Cnidaria	
Hydrazoa	
Hydridae	
Hydra spp.	x
Phylum Mollusca	
Gastropoda	x
Archaeogastropoda	x
Pelecypoda	x
Phylum Nematoda	x
Phylum Nematomorpha	x
Phylum Nemertea	x
Phylum Platyhelminthes	
Turbellaria	x
Phylum Tardigrada	x
Total	62

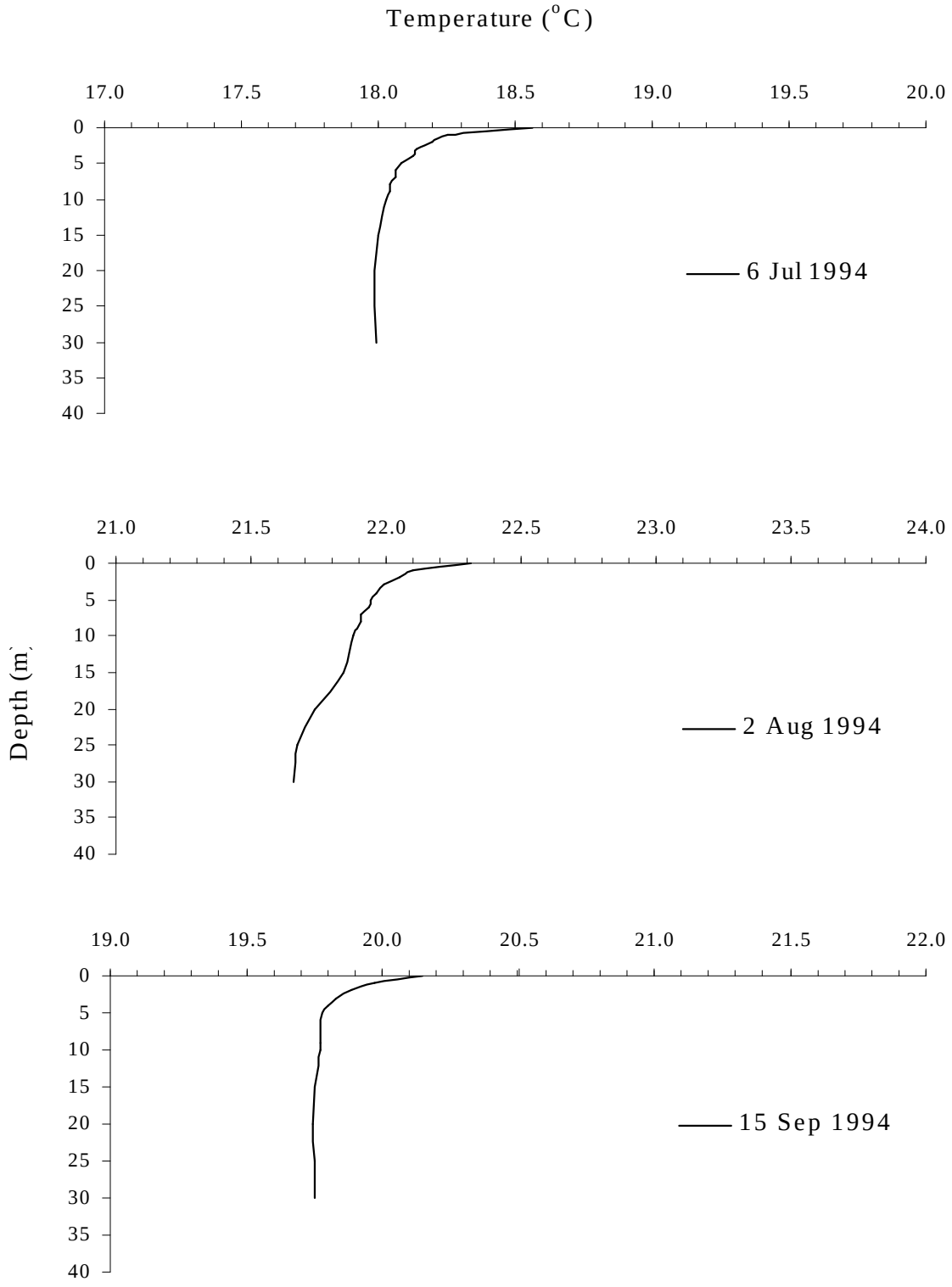
APPENDIX B



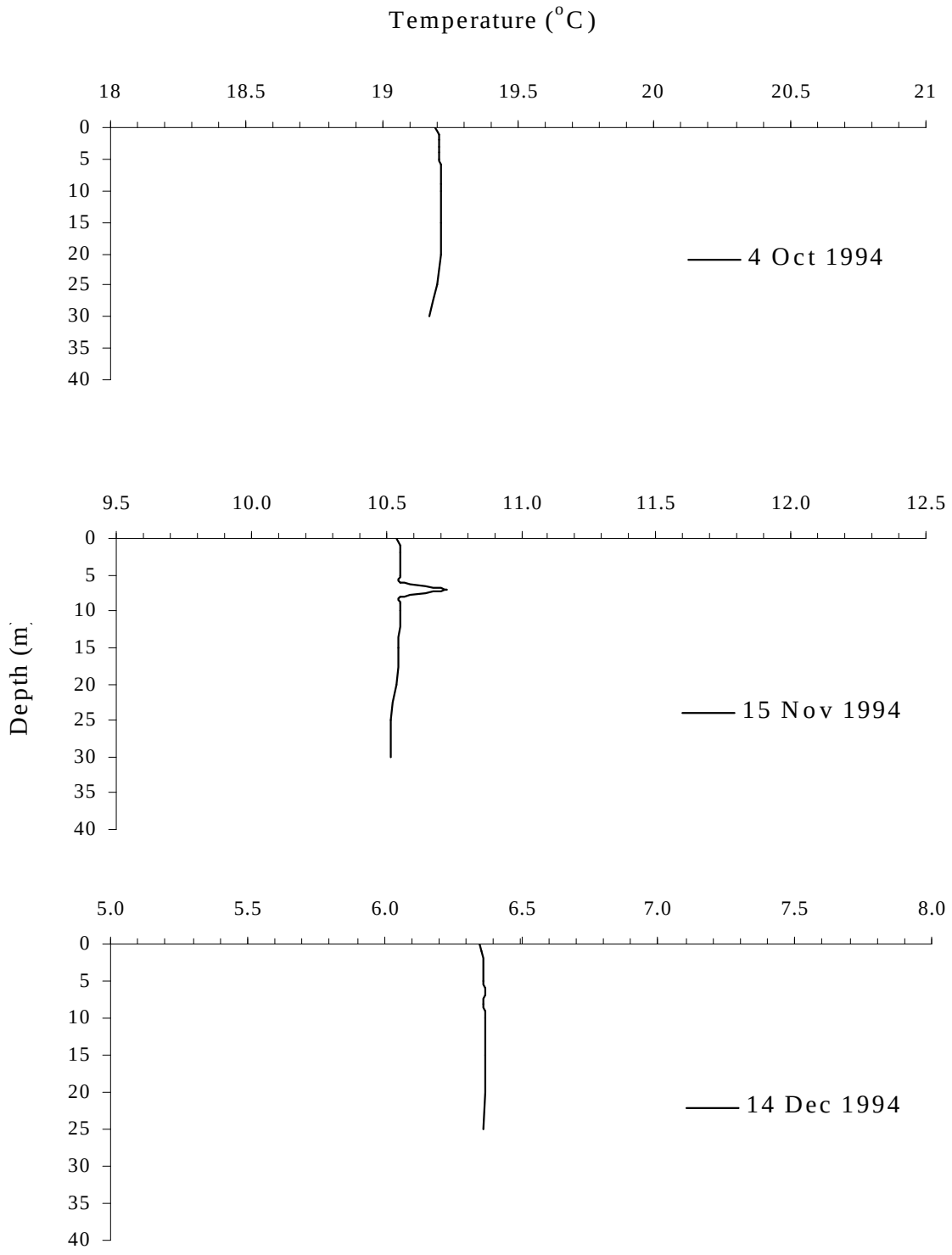
Appendix Figure B-1. Temporal and spatial variation of John Day Reservoir water temperature and turbidity, April 1994 to September 1995. Results from sampling at lower-, mid-, and upper-reservoir stations and at Crow Butte backwater.



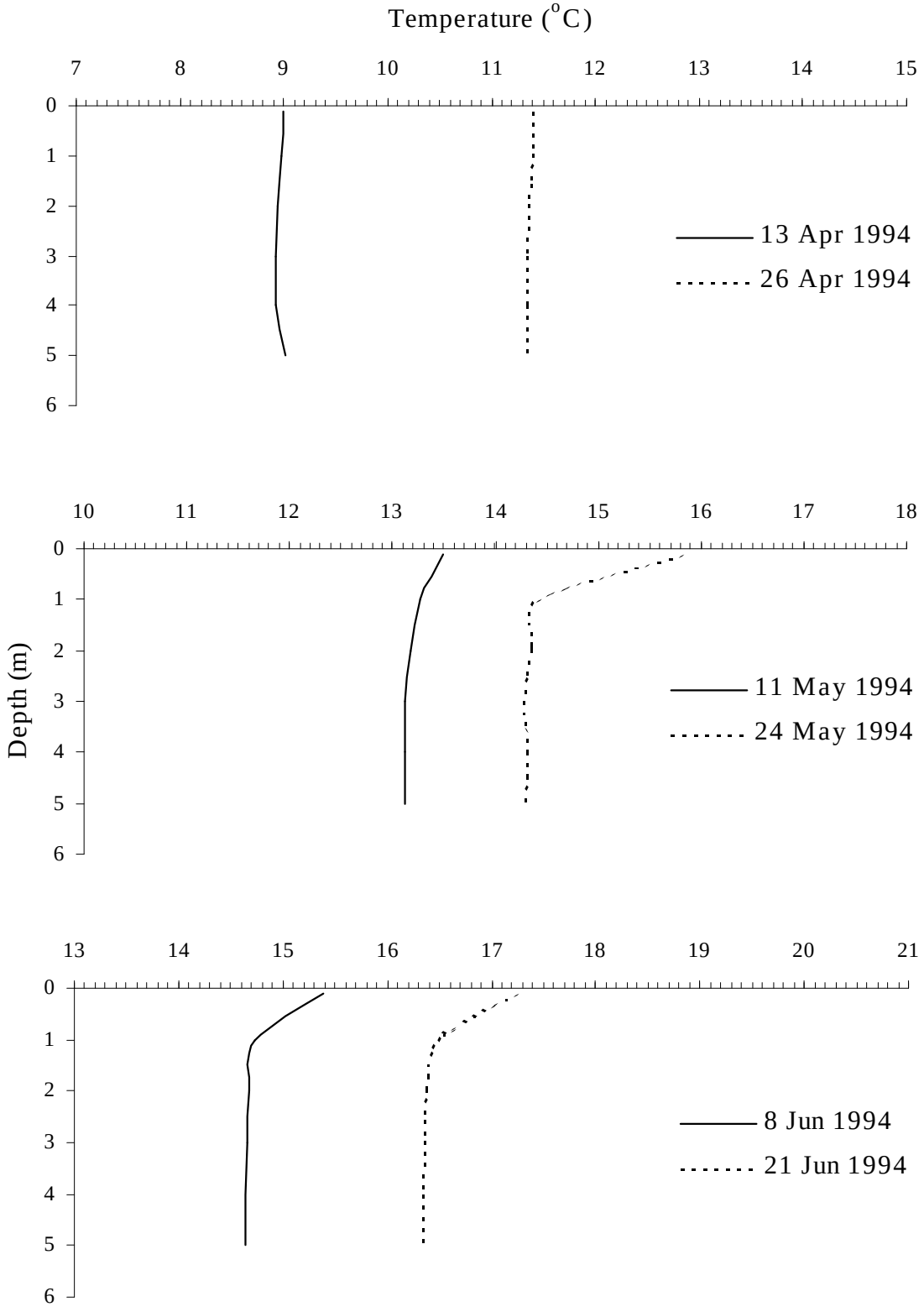
Appendix Figure B-2. John Day Reservoir surface-to-bottom temperature profiles. Combined data for lower- and mid-reservoir stations, April to December 1994.



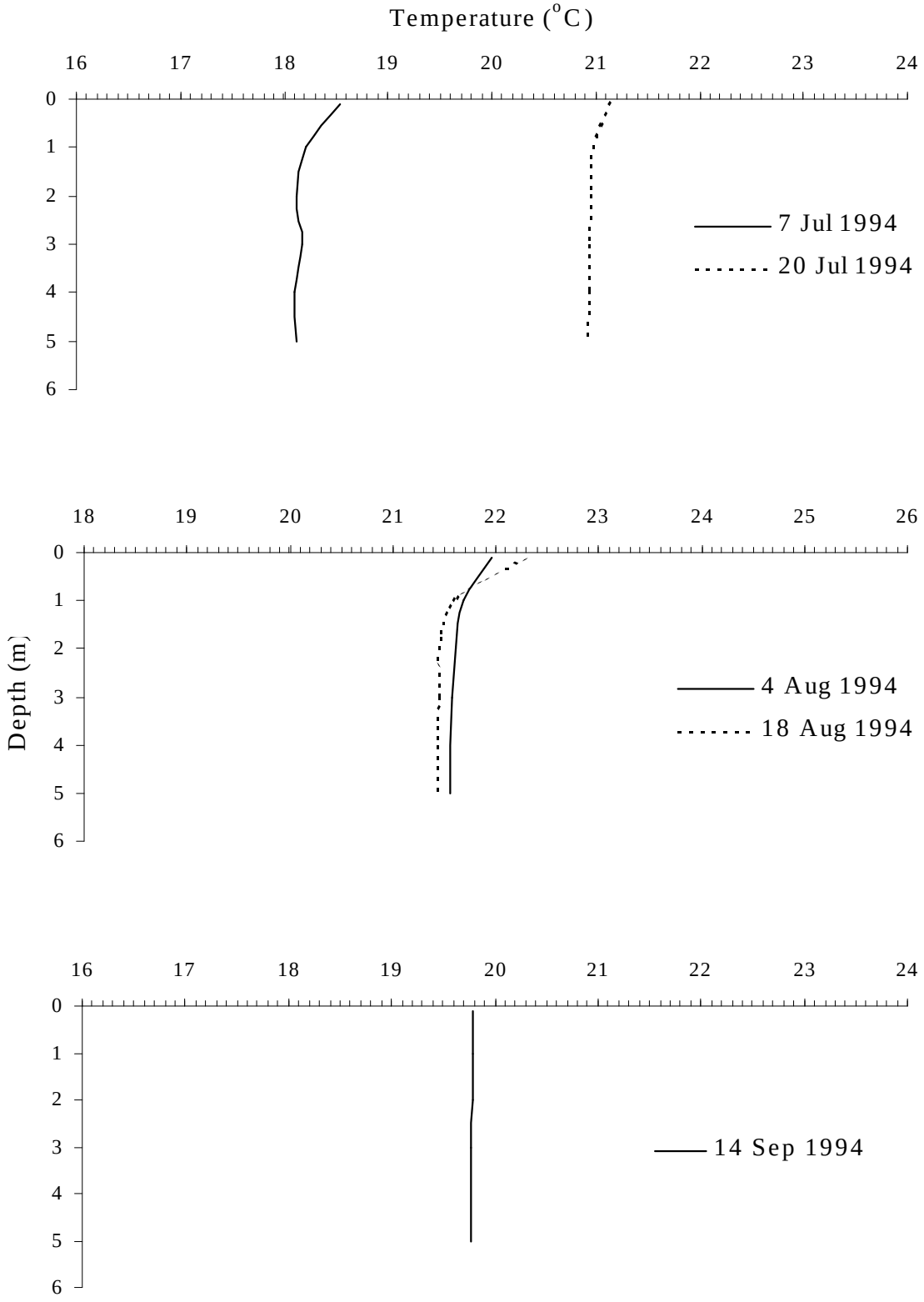
Appendix Figure B-2. Continued.



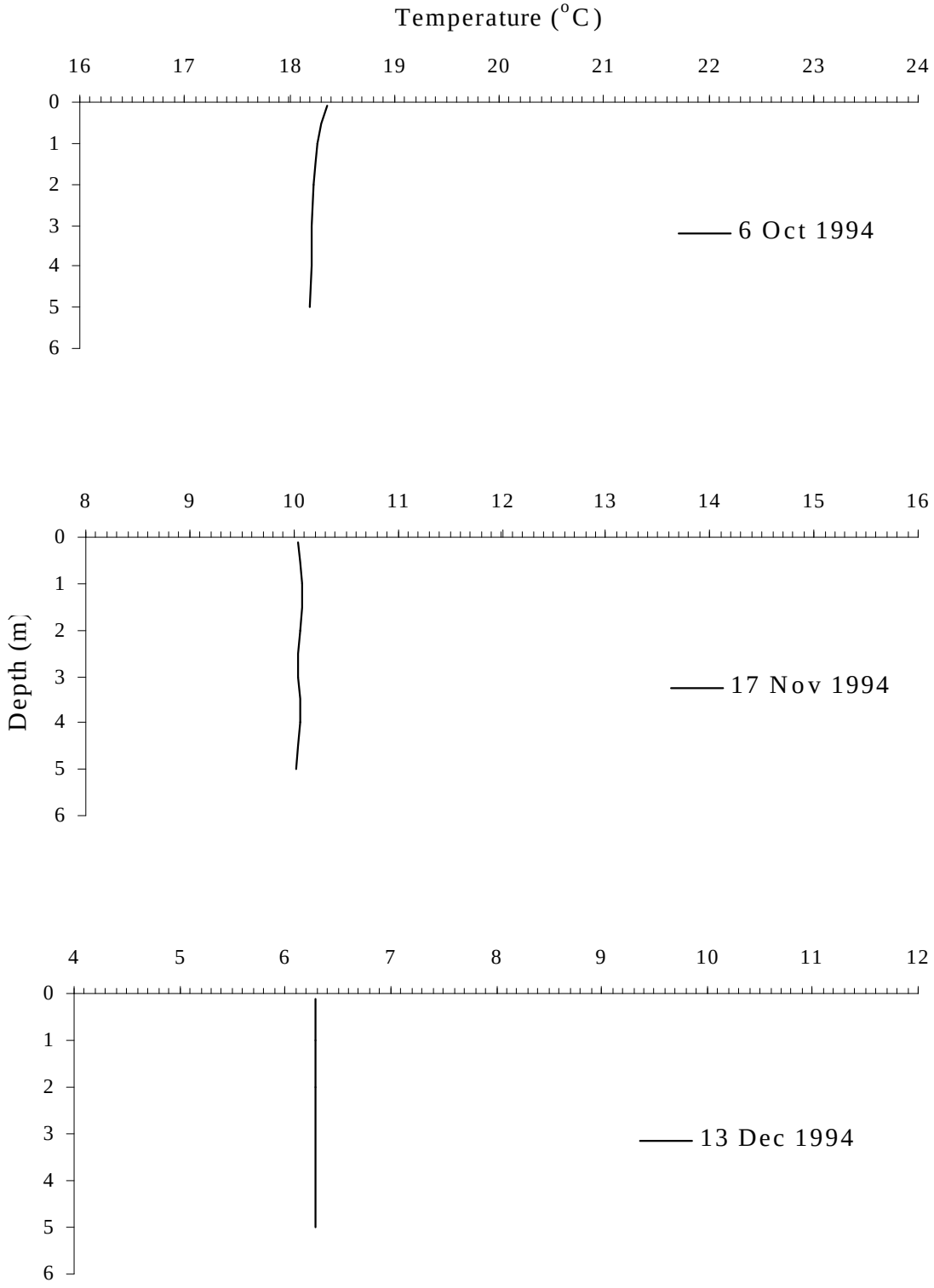
Appendix Figure B-2. Continued.



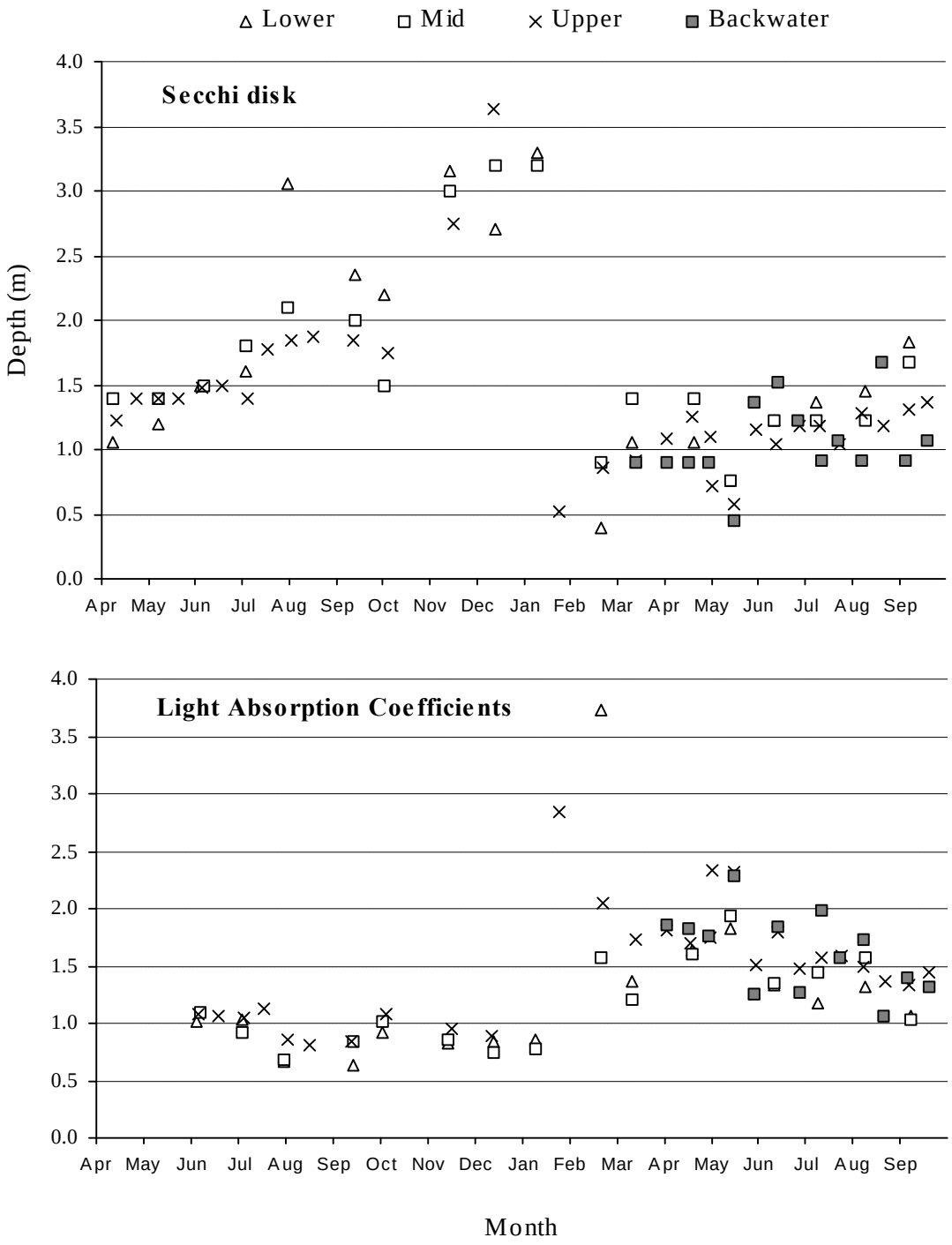
Appendix Figure B-3. John Day Reservoir surface-to-bottom temperature profiles. Combined data for five upper-reservoir stations, April to December 1994.



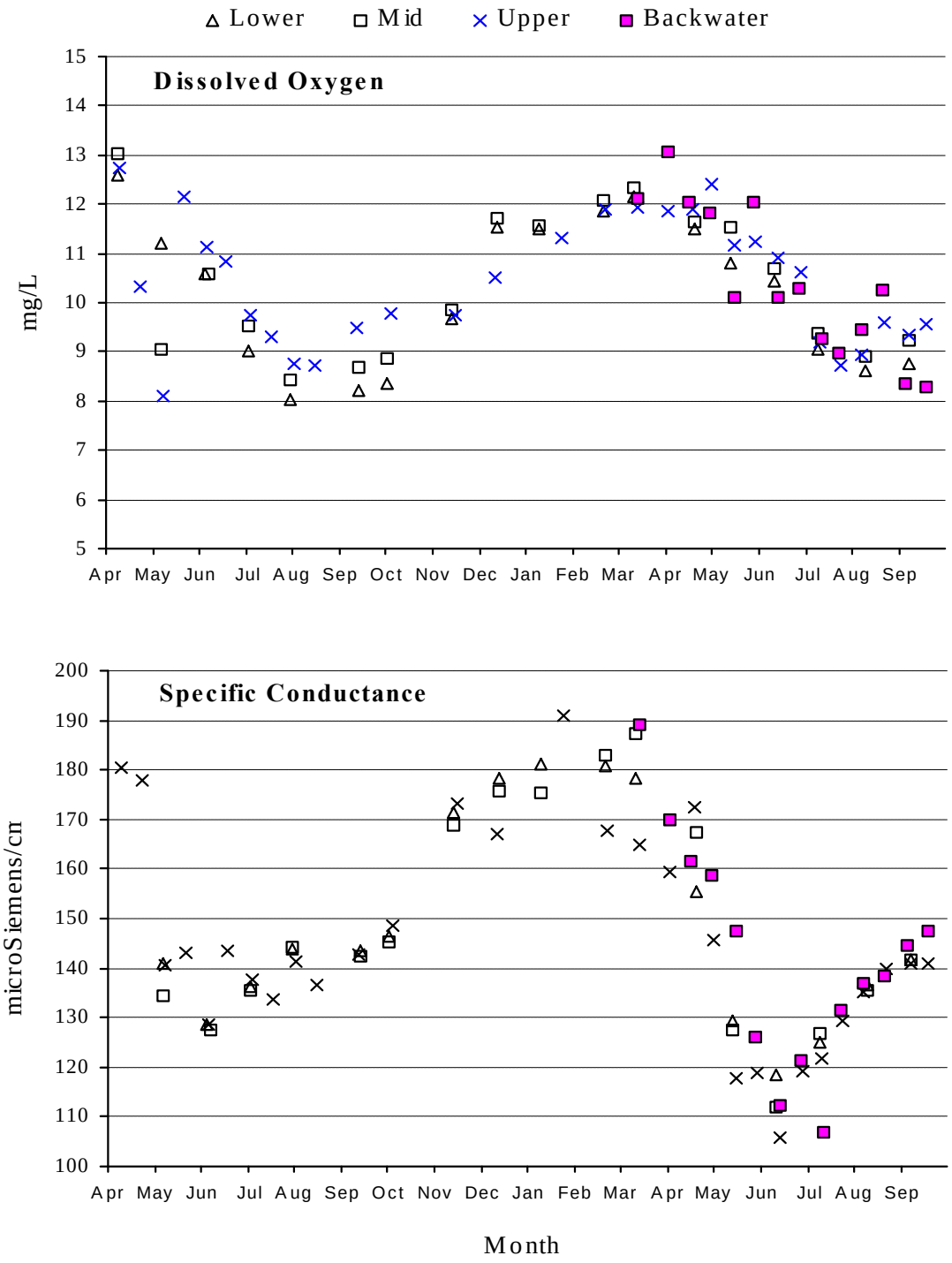
Appendix Figure B-3. Continued.



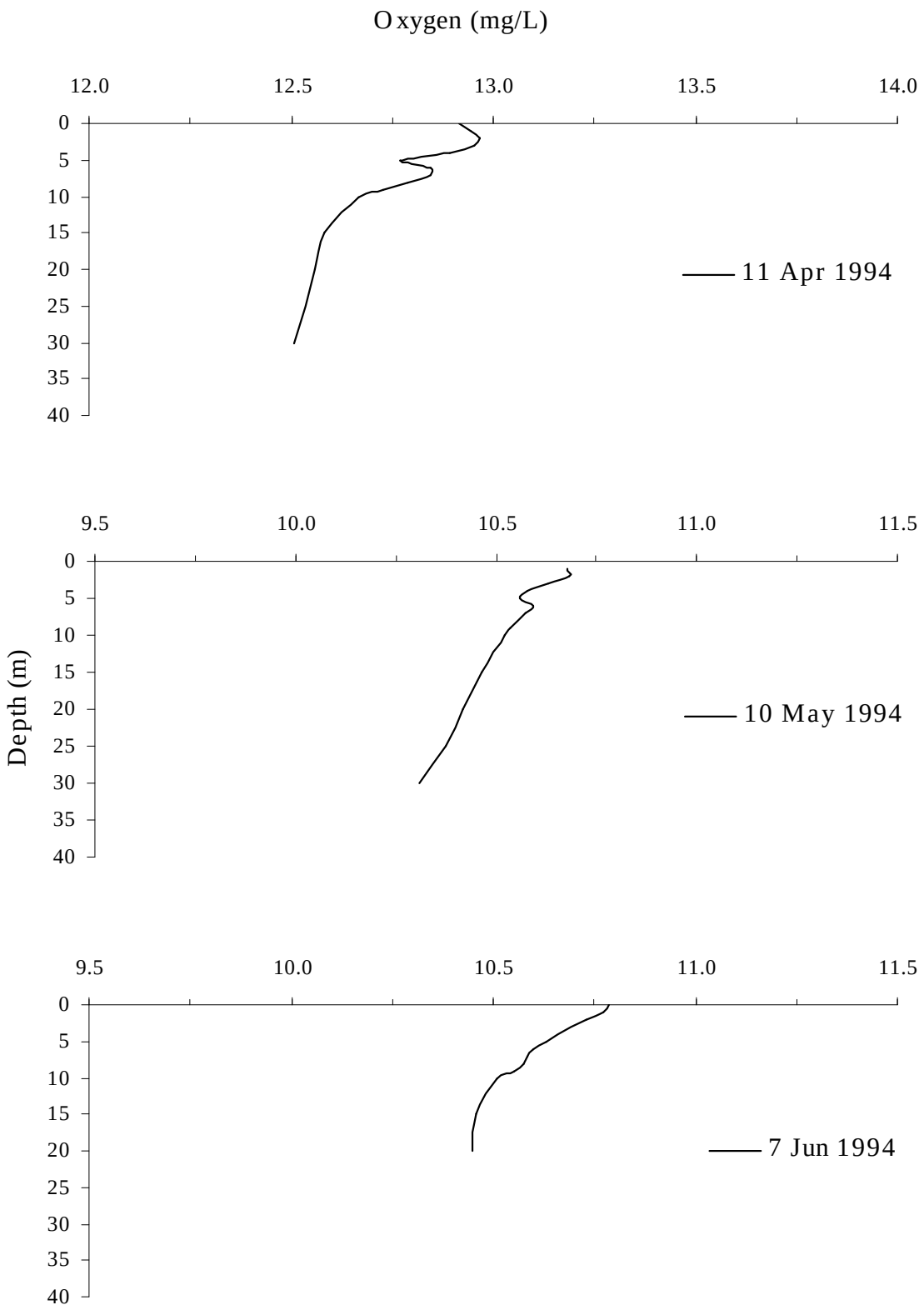
Appendix Figure B-3. Continued.



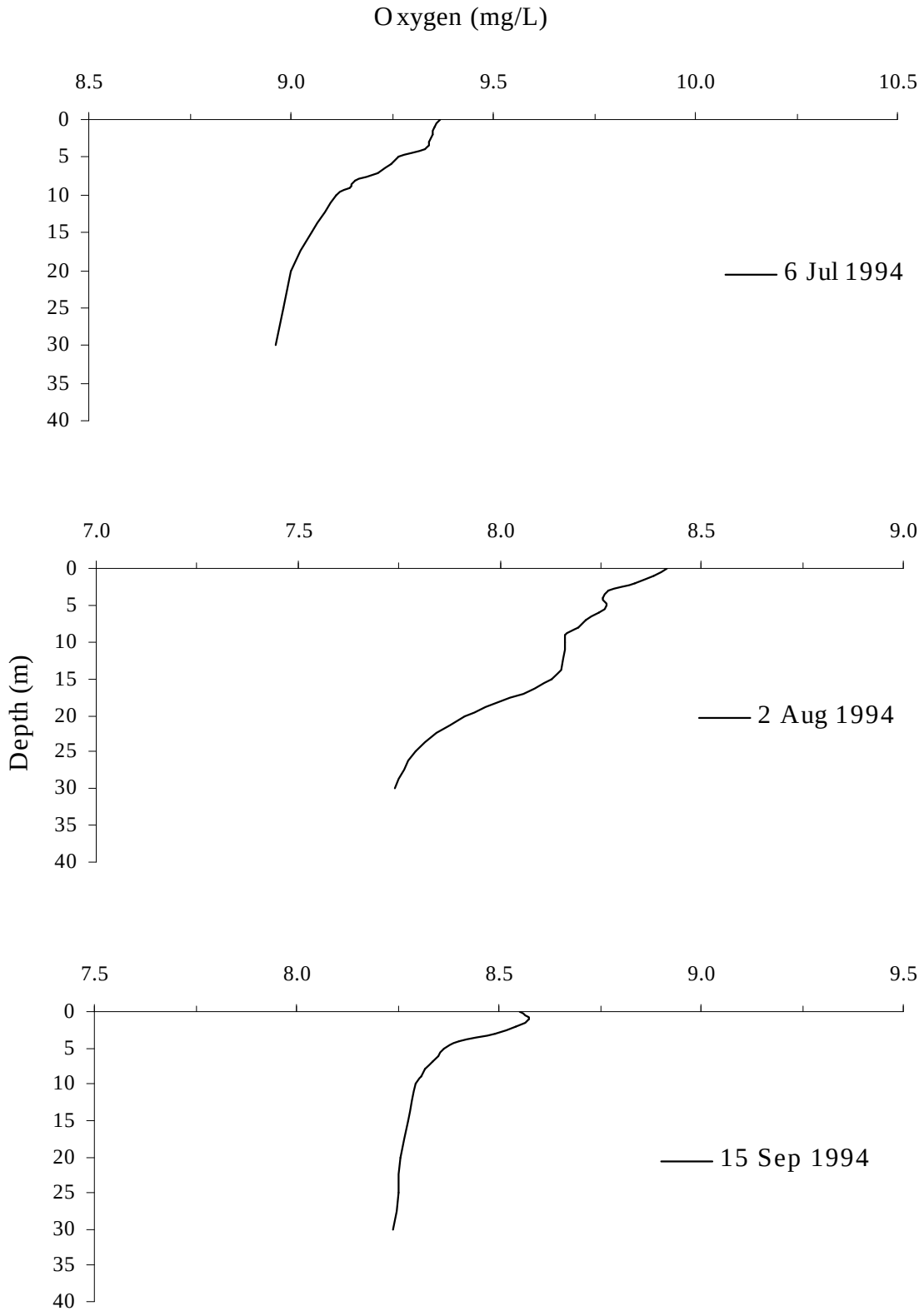
Appendix Figure B-4. Temporal and spatial variation of John Day Reservoir Secchi disk readings and vertical light absorption coefficients, April 1994 to September 1995. Results from sampling at lower-, mid-, and upper-reservoir stations and at Crow Butte backwater.



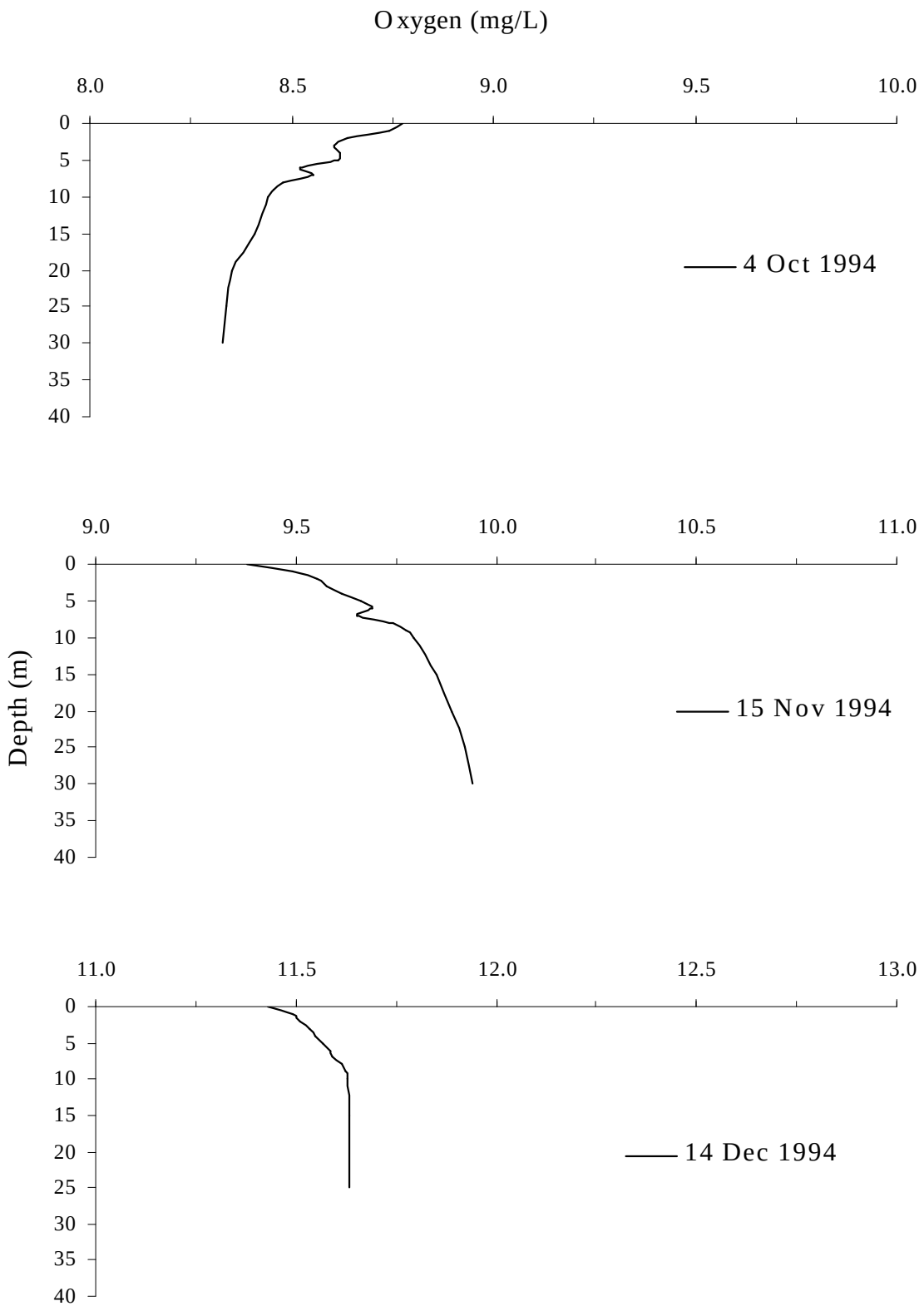
Appendix Figure B-5. Temporal and spatial variation of John Day Reservoir dissolved oxygen levels and specific conductance, April 1994 to September 1995. Results from sampling at lower-, mid-, and upper-reservoir stations and at Crow Butte backwater.



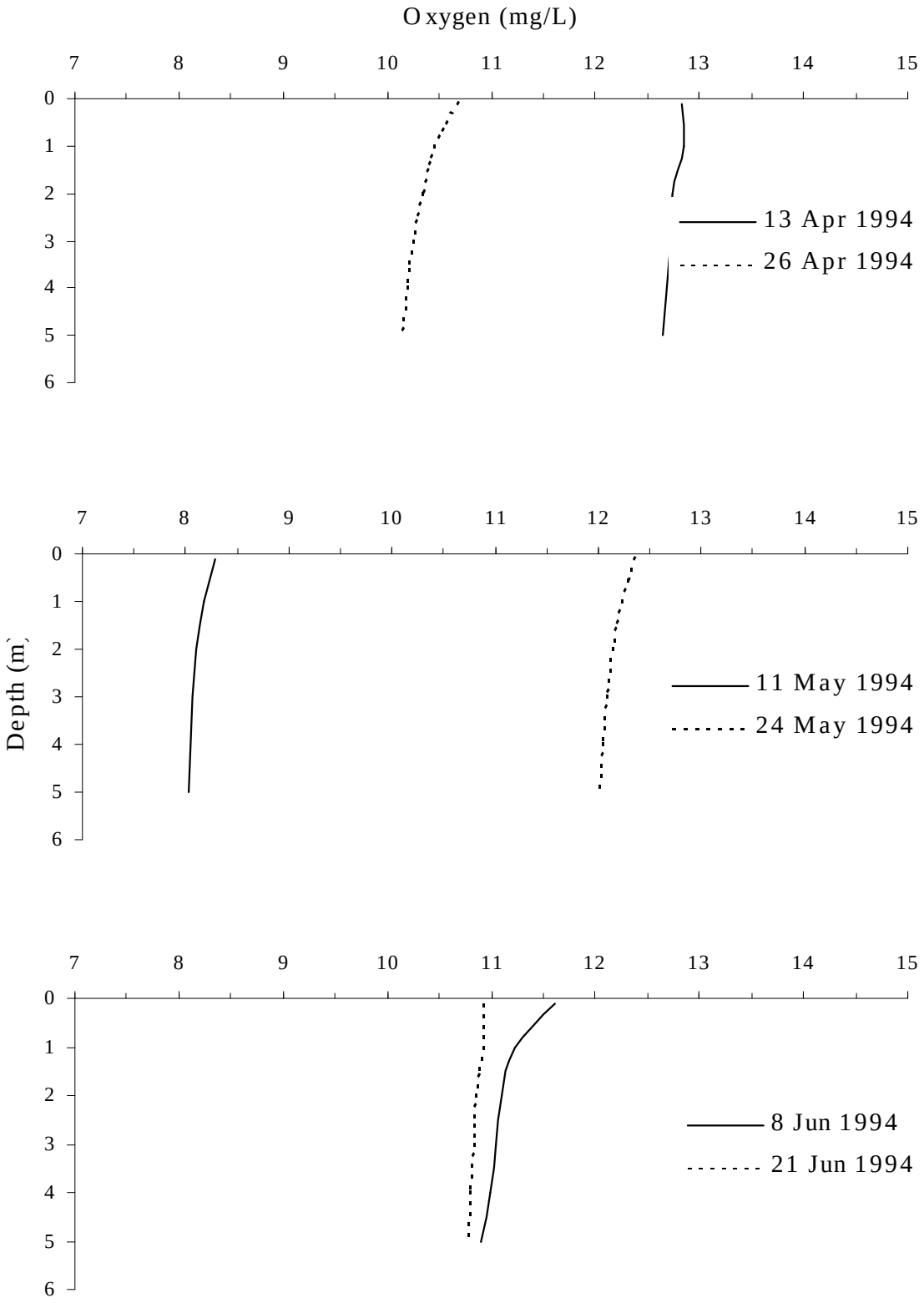
Appendix Figure B-6. John Day Reservoir surface-to-bottom oxygen profiles. Combined data for lower- and mid-reservoir stations, April to December 1994.



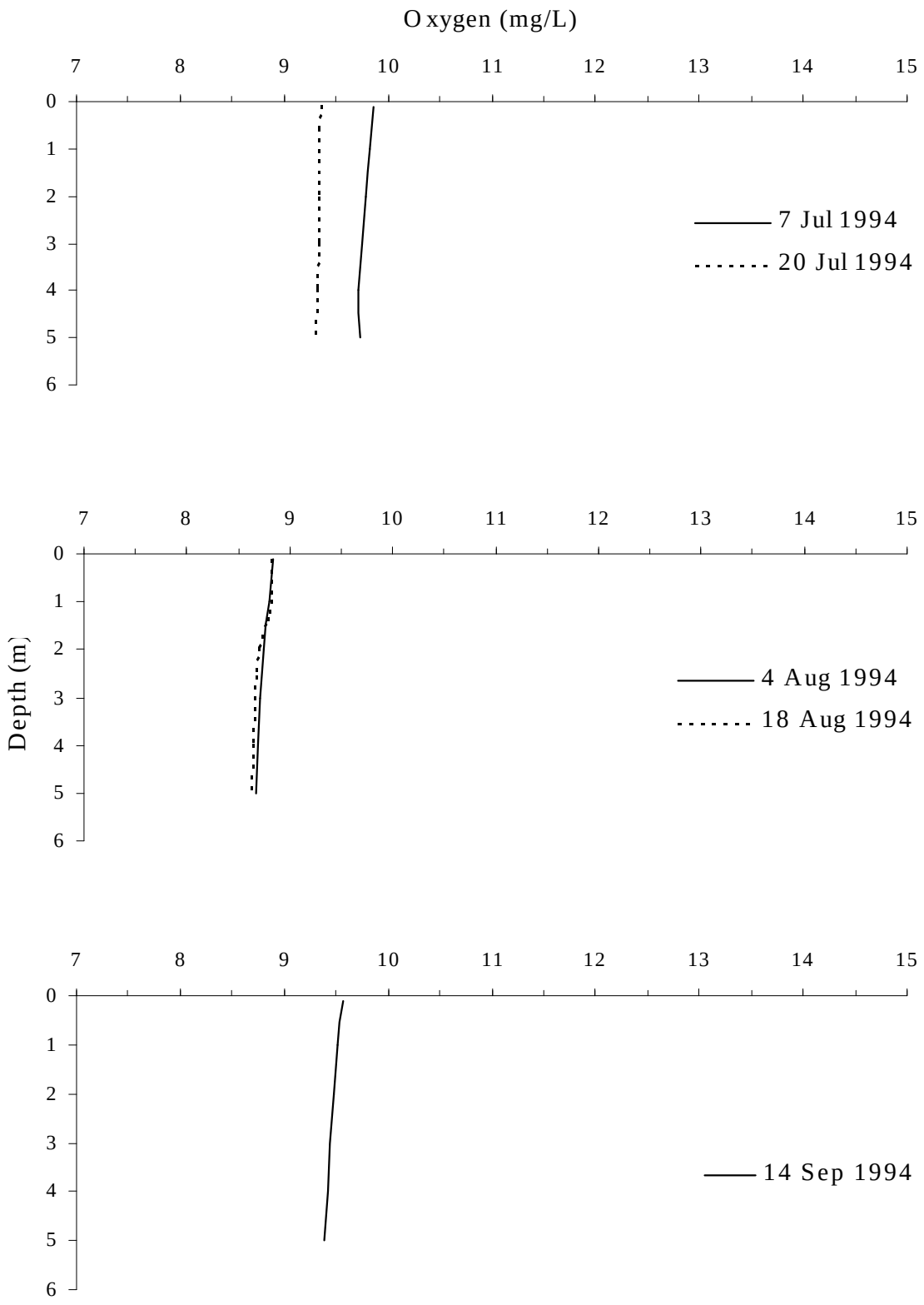
Appendix Figure B-6. Continued.



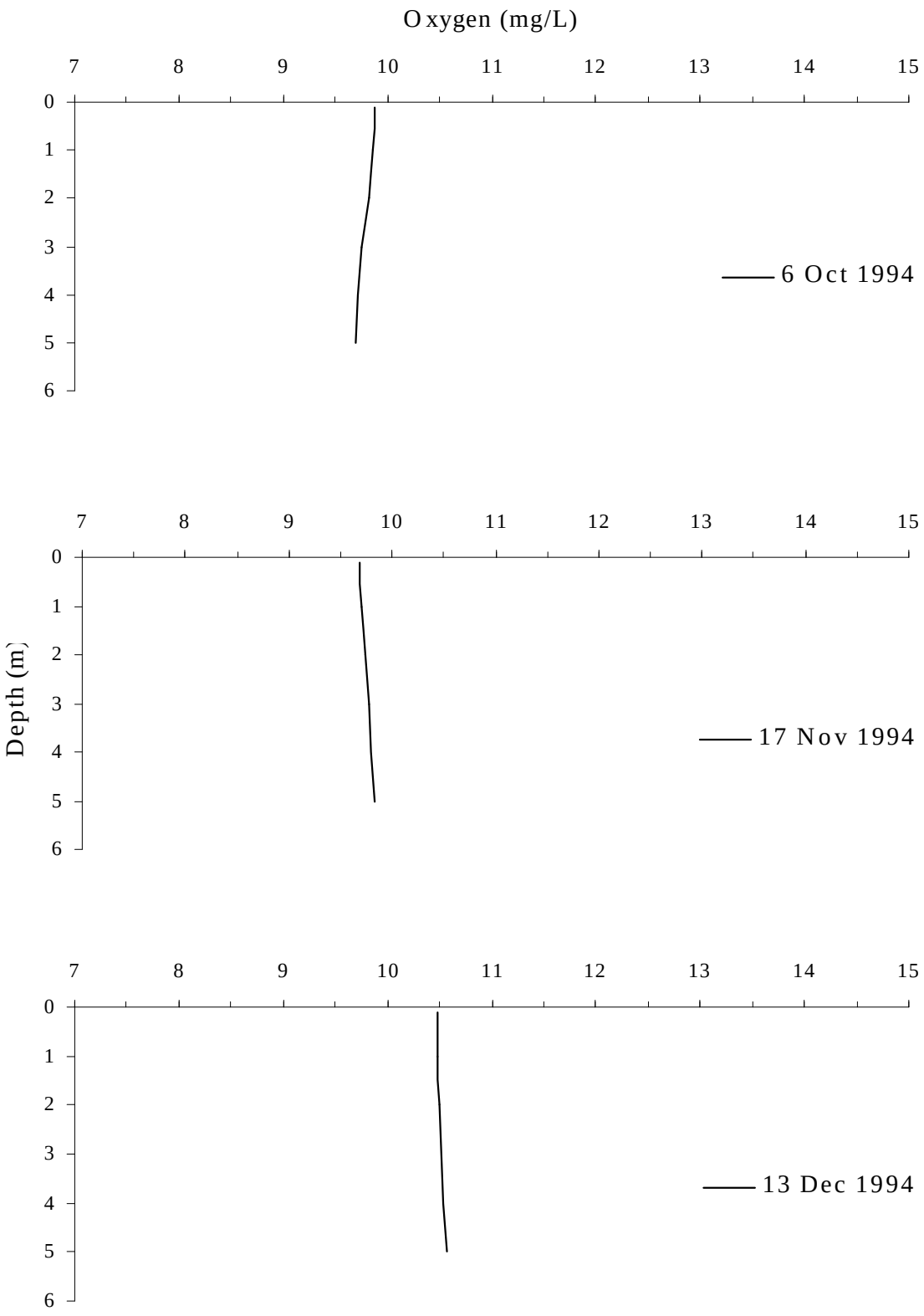
Appendix Figure B-6. Continued.



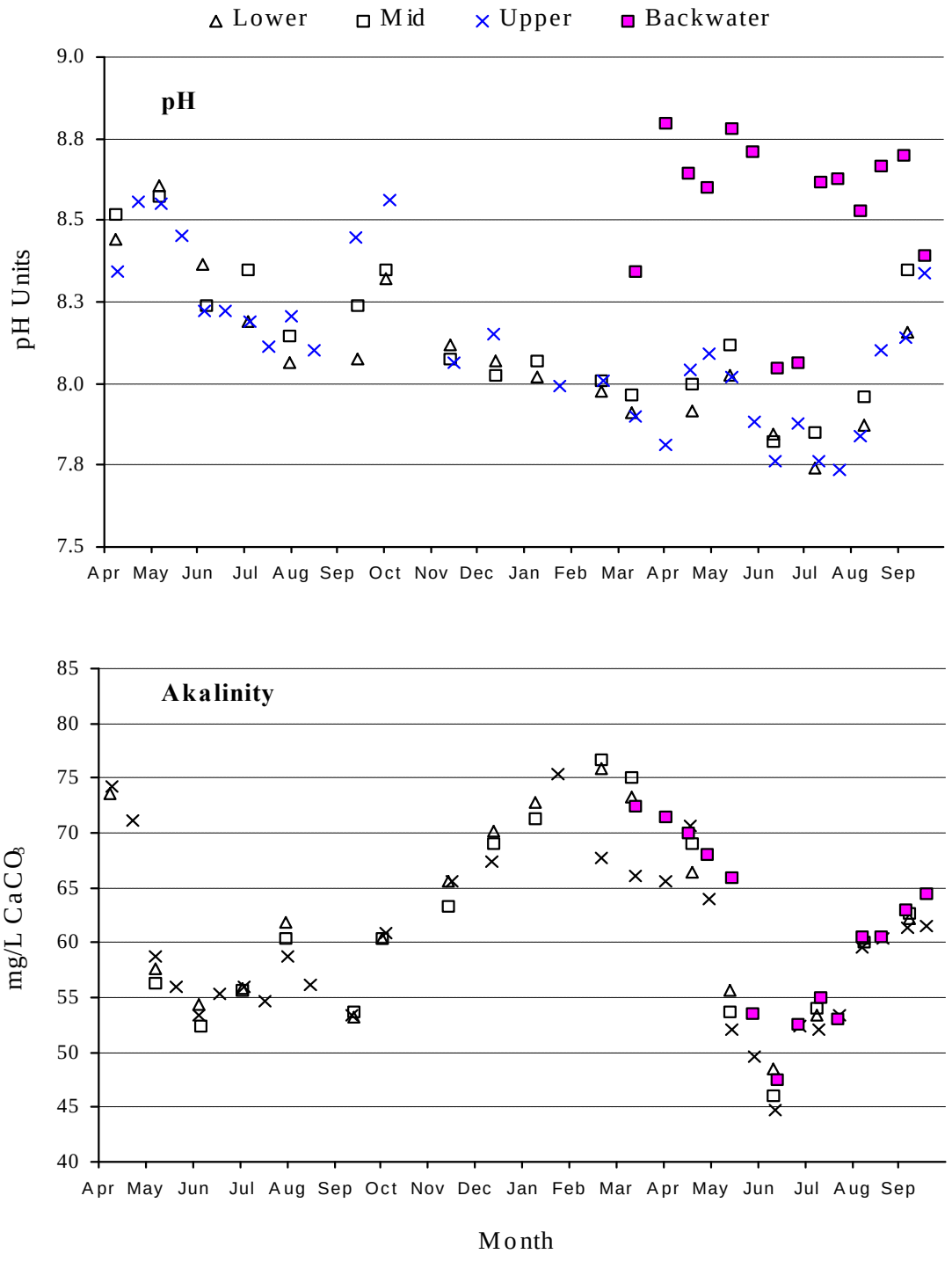
Appendix Figure B-7. John Day Reservoir surface-to-bottom oxygen profiles. Combined data for five upper-reservoir stations, April to December 1994.



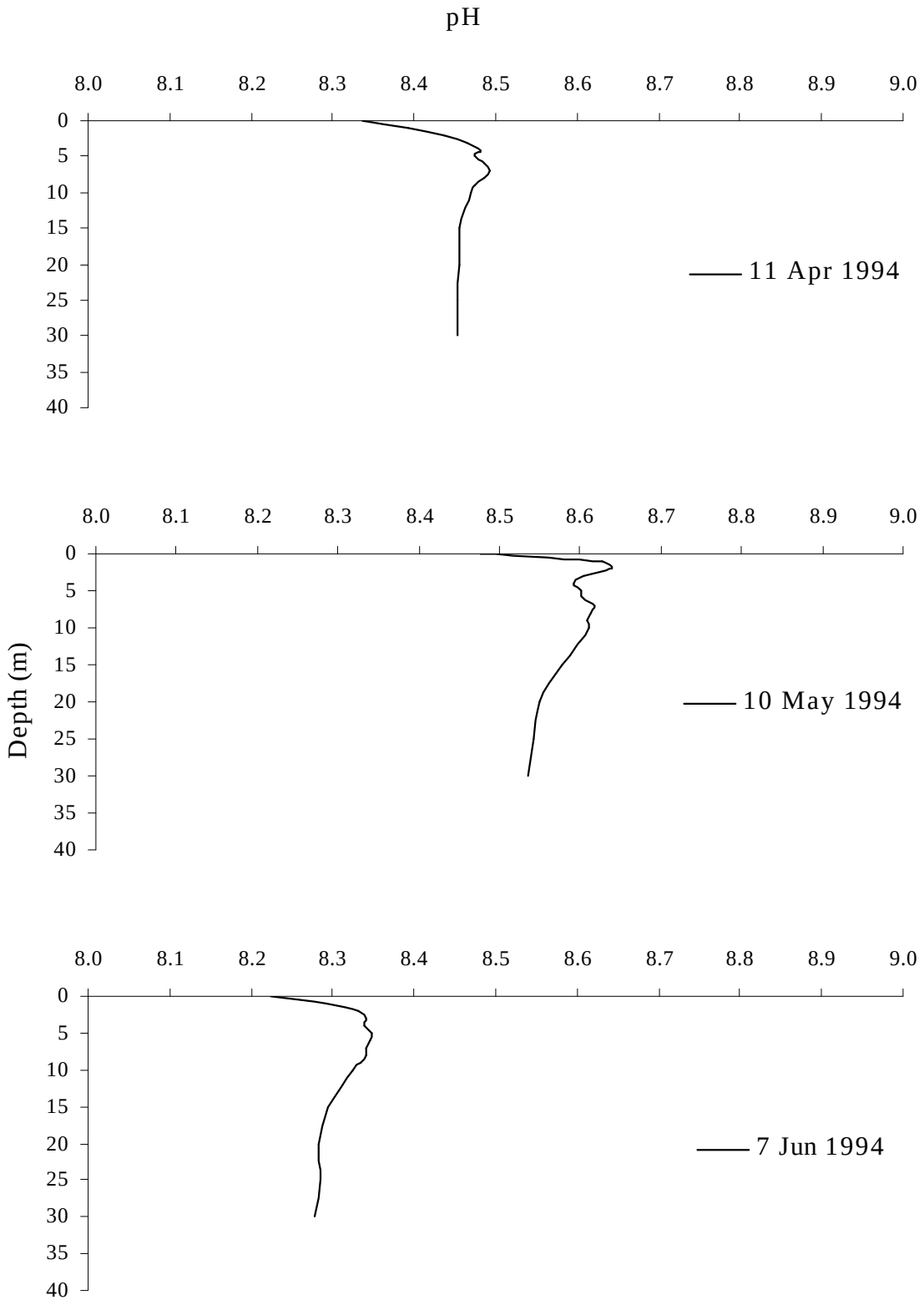
Appendix Figure B-7. Continued.



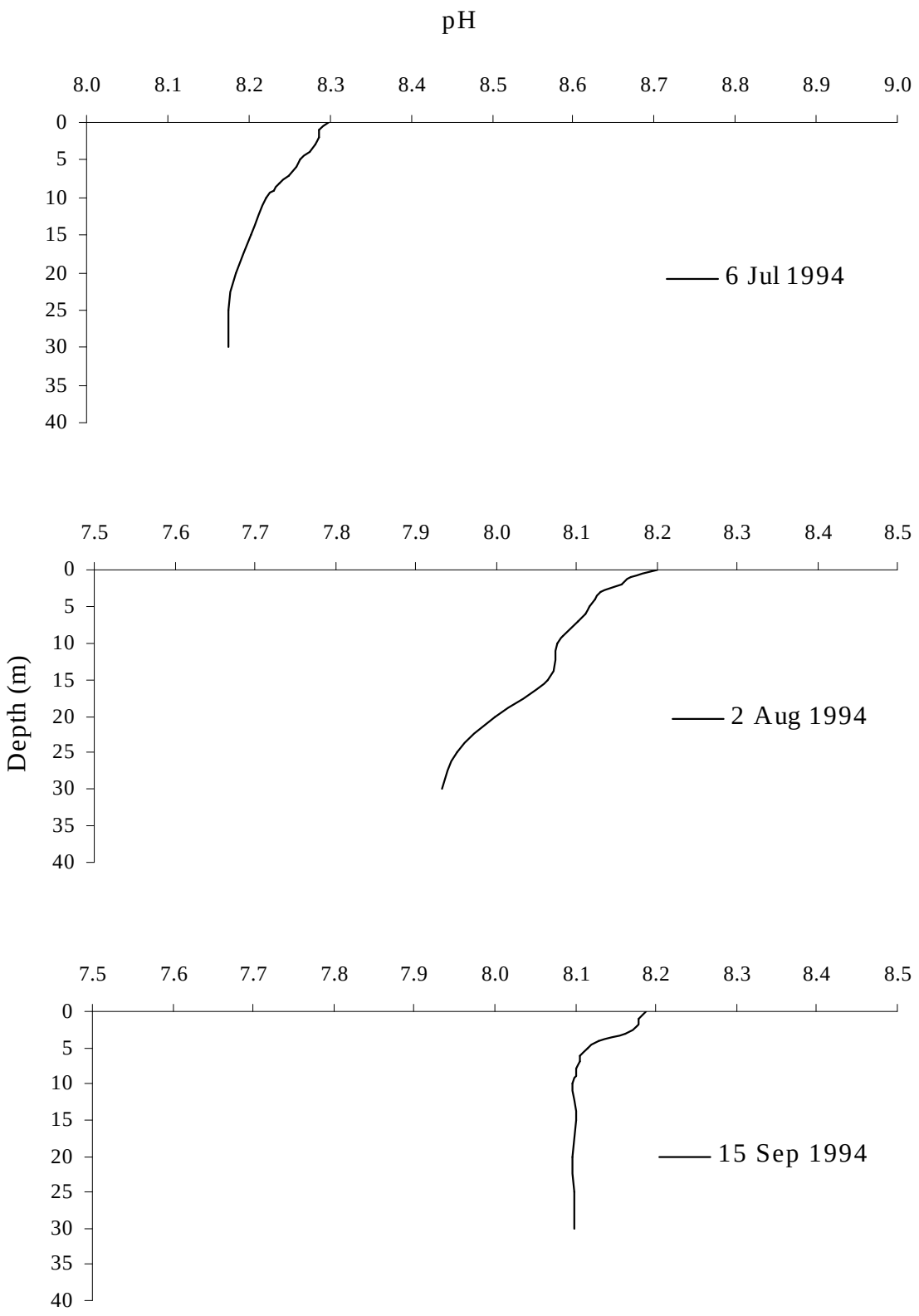
Appendix Figure B-7. Continued.



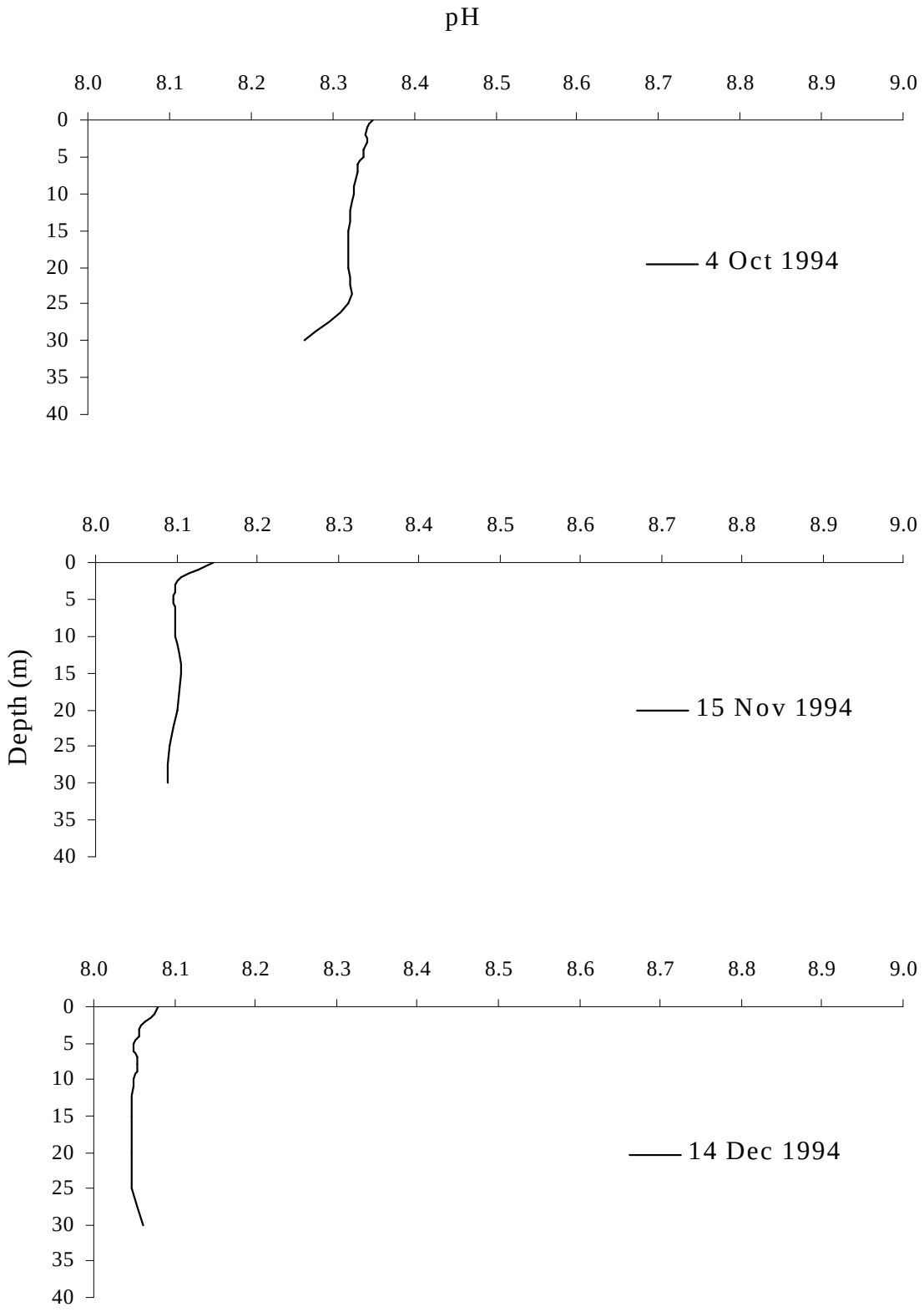
Appendix Figure B-8. Temporal and spatial variation of John Day Reservoir pH and alkalinity, April 1994 to September 1995. Results from sampling at lower-, mid-, and upper-reservoir stations and at Crow Butte backwater.



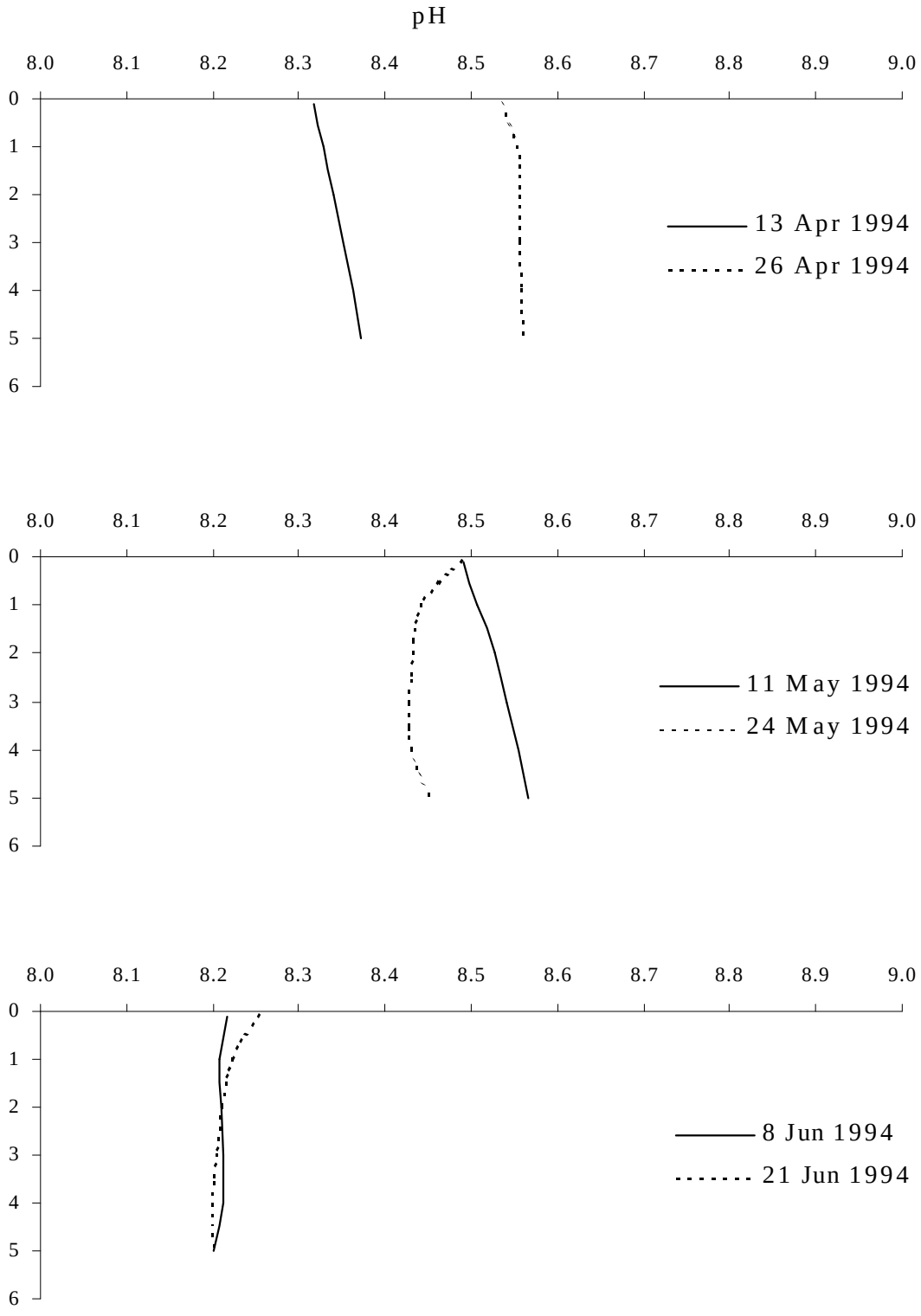
Appendix Figure B-9. John Day Reservoir surface-to-bottom pH profiles. Combined data for lower- and mid-reservoir stations, April to December 1994.



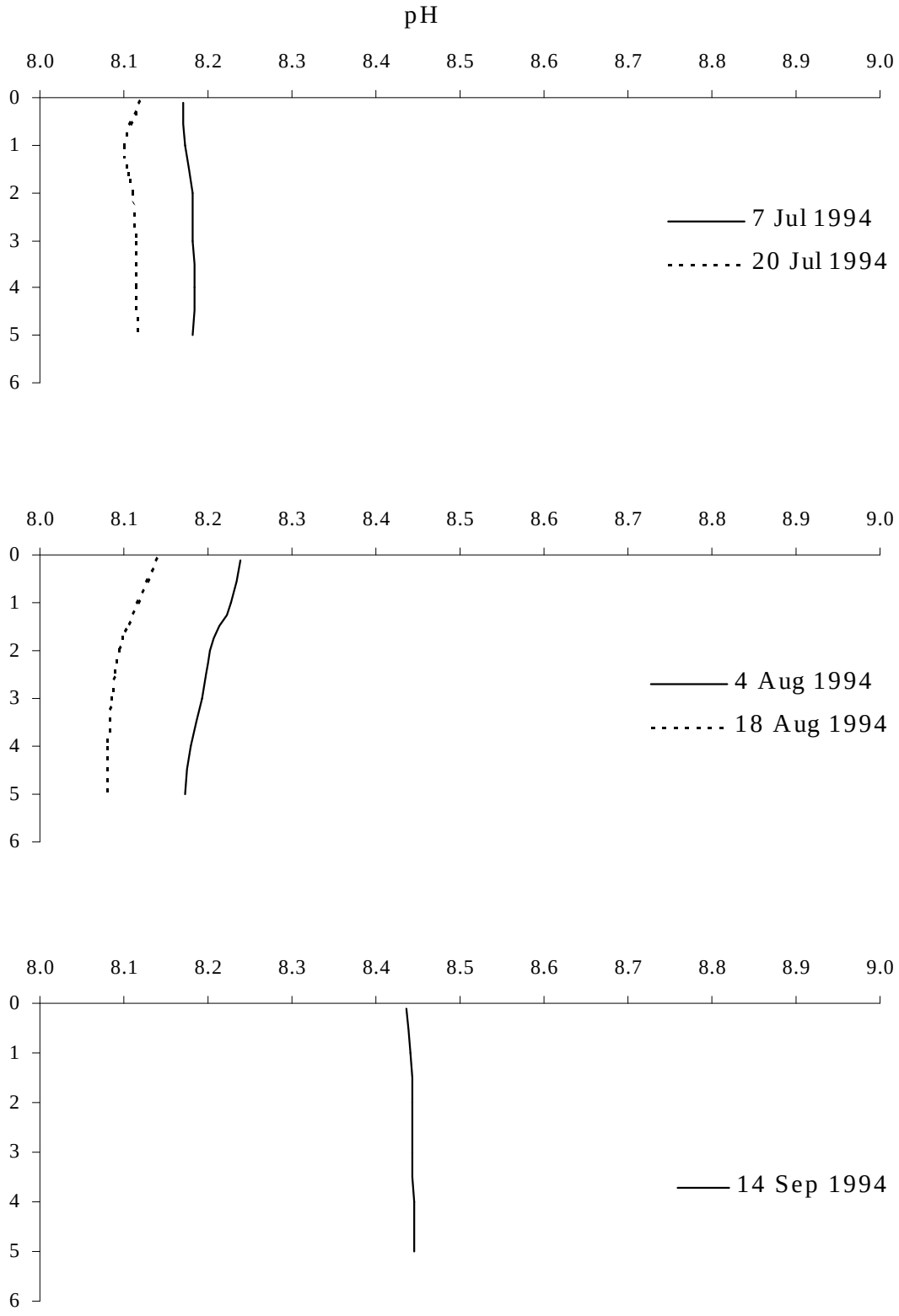
Appendix Figure B-9. Continued.



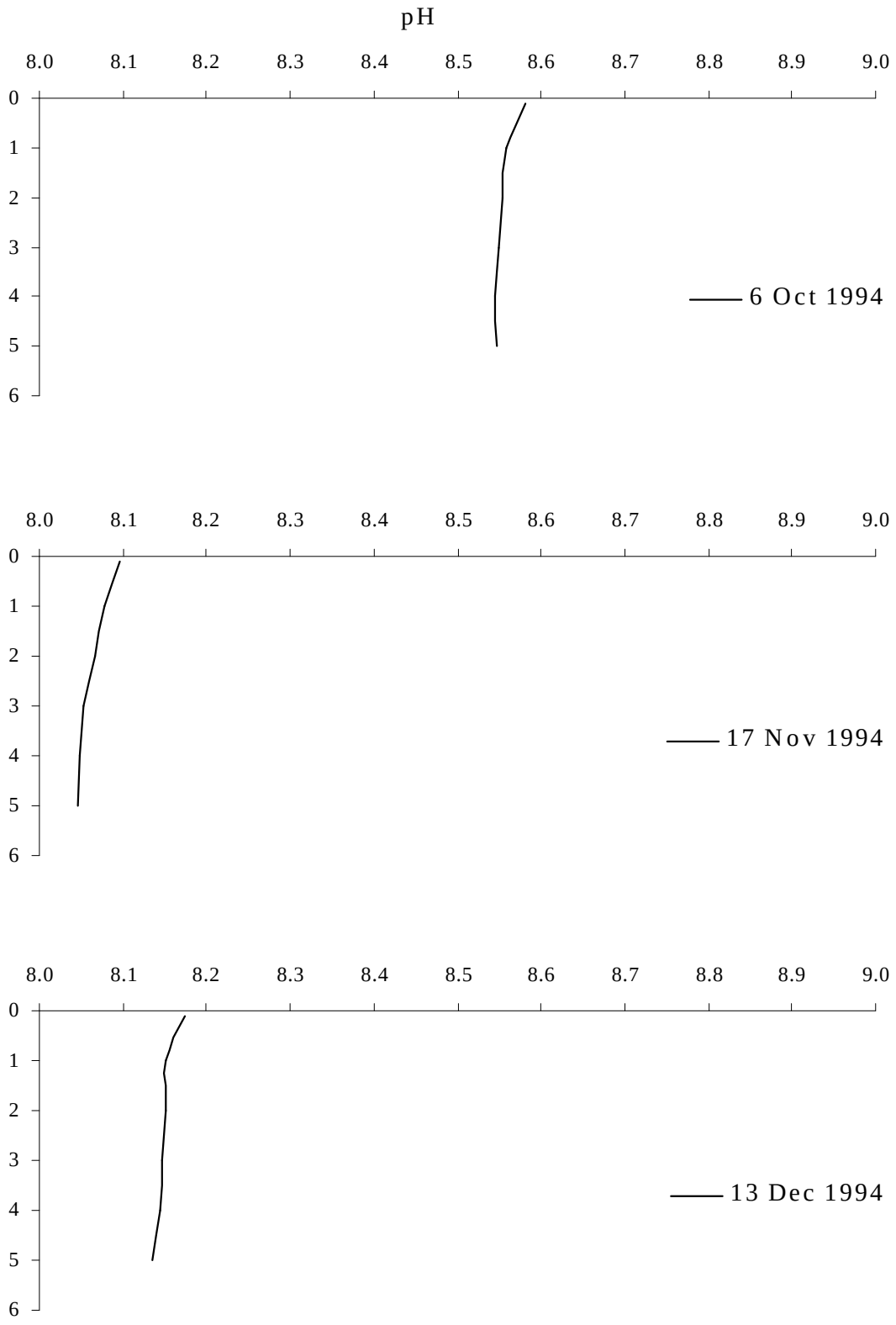
Appendix Figure B-9. Continued.



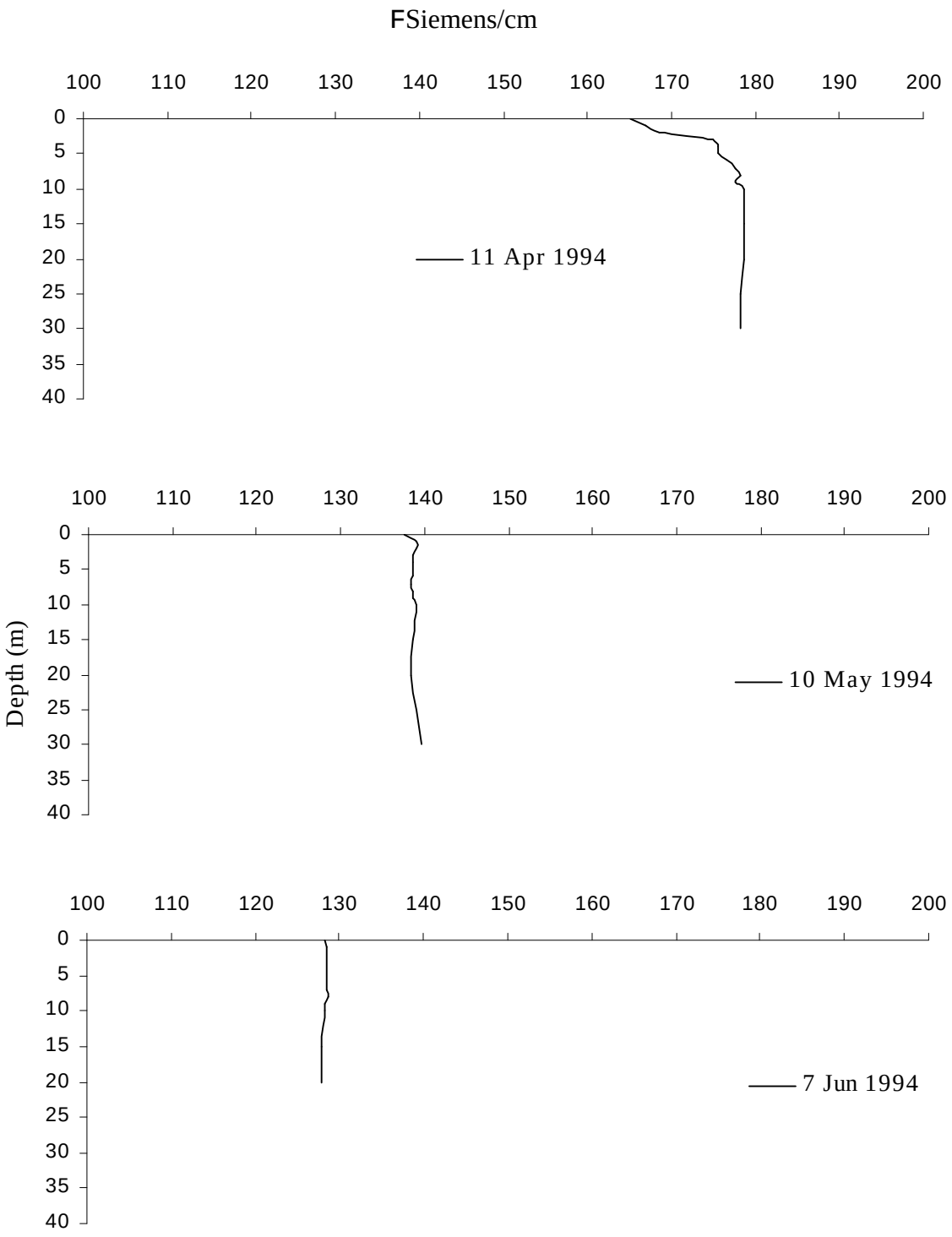
Appendix Figure B-10. John Day Reservoir surface-to-bottom pH profiles. Combined data for upper-reservoir stations, April to December 1994.



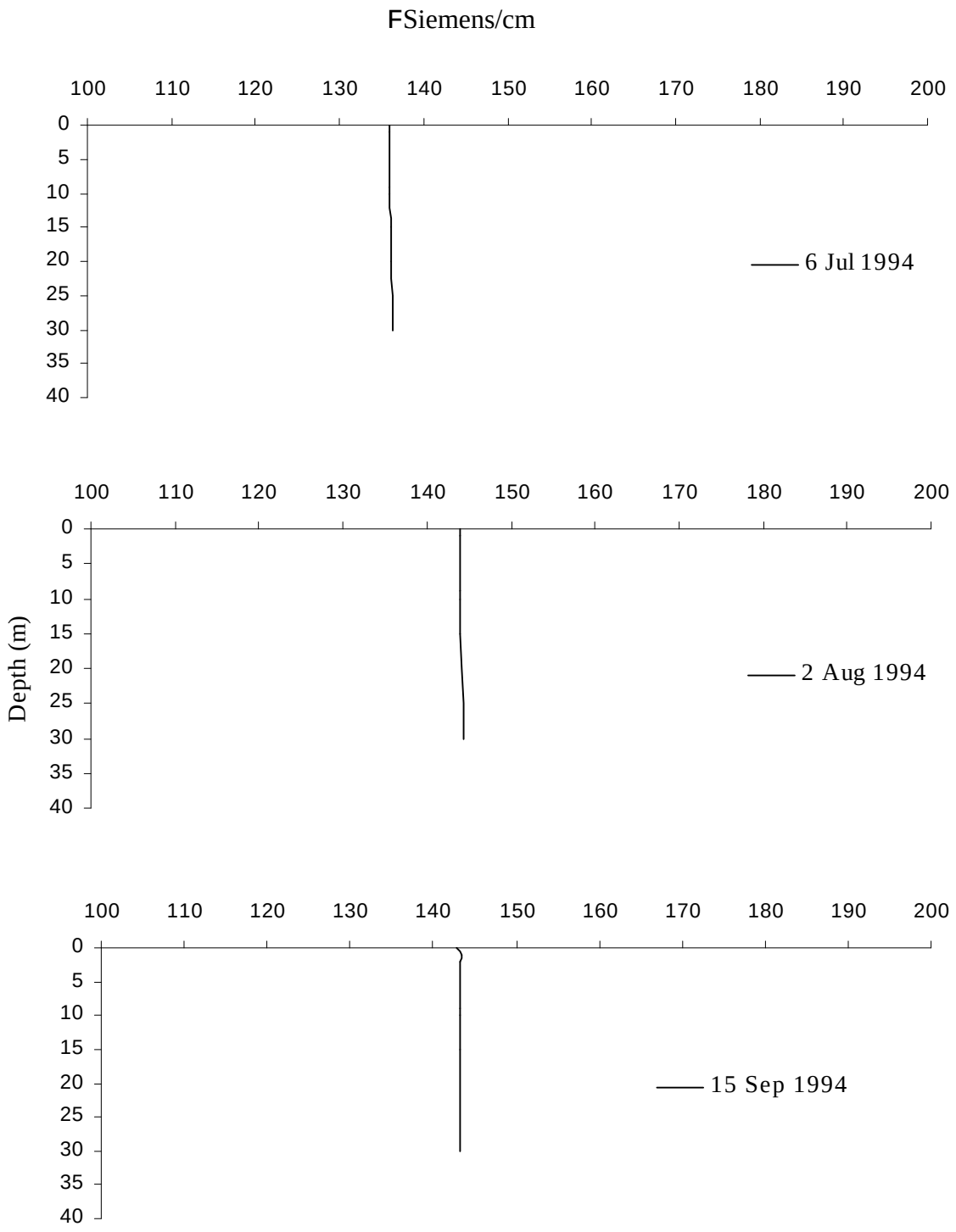
Appendix Figure B-10. Continued.



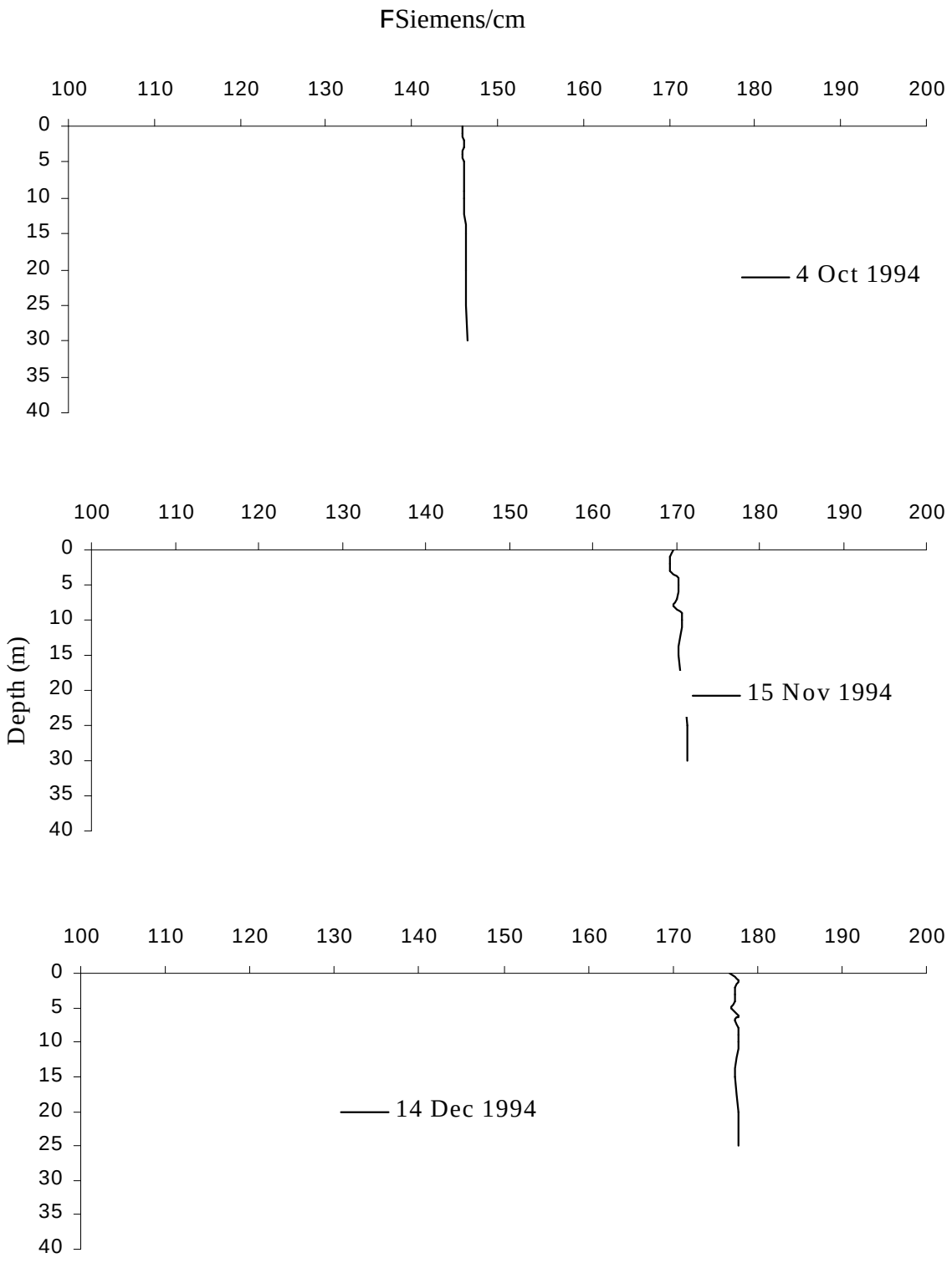
Appendix Figure B-10. Continued.



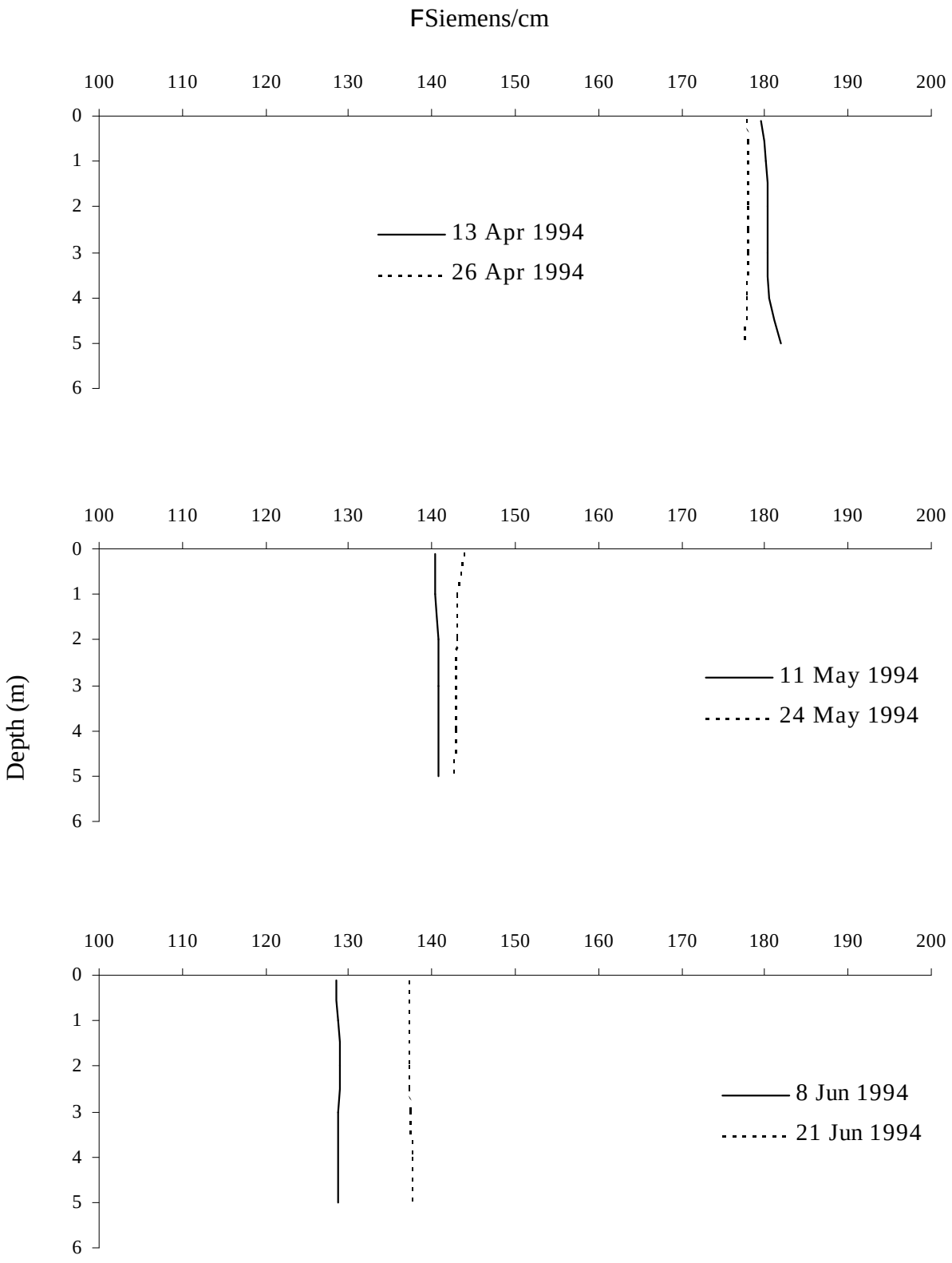
Appendix Figure B-11. John Day Reservoir specific conductance profiles. Combined data for lower- and mid-reservoir stations, April to December 1994.



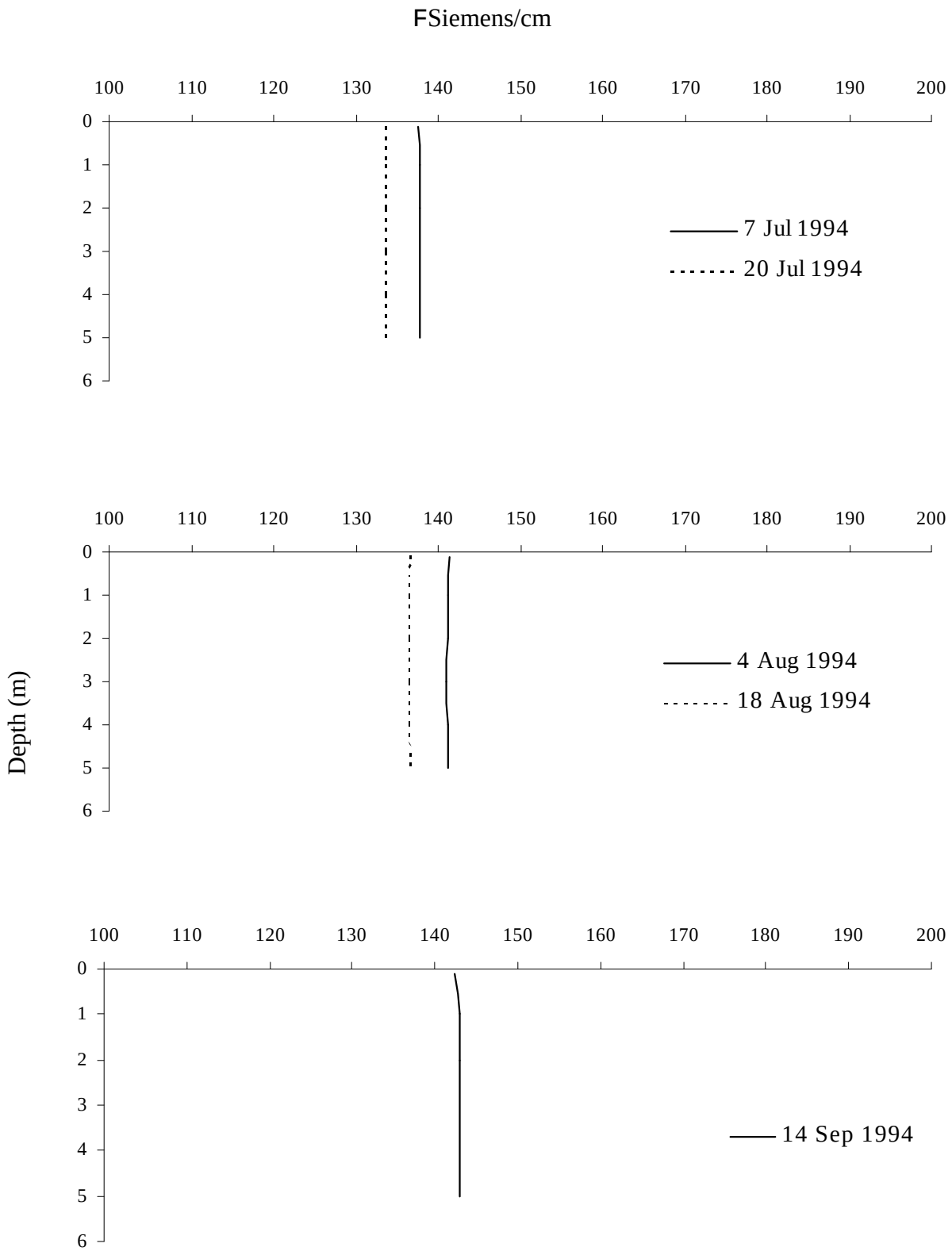
Appendix Figure B-11. Continued.



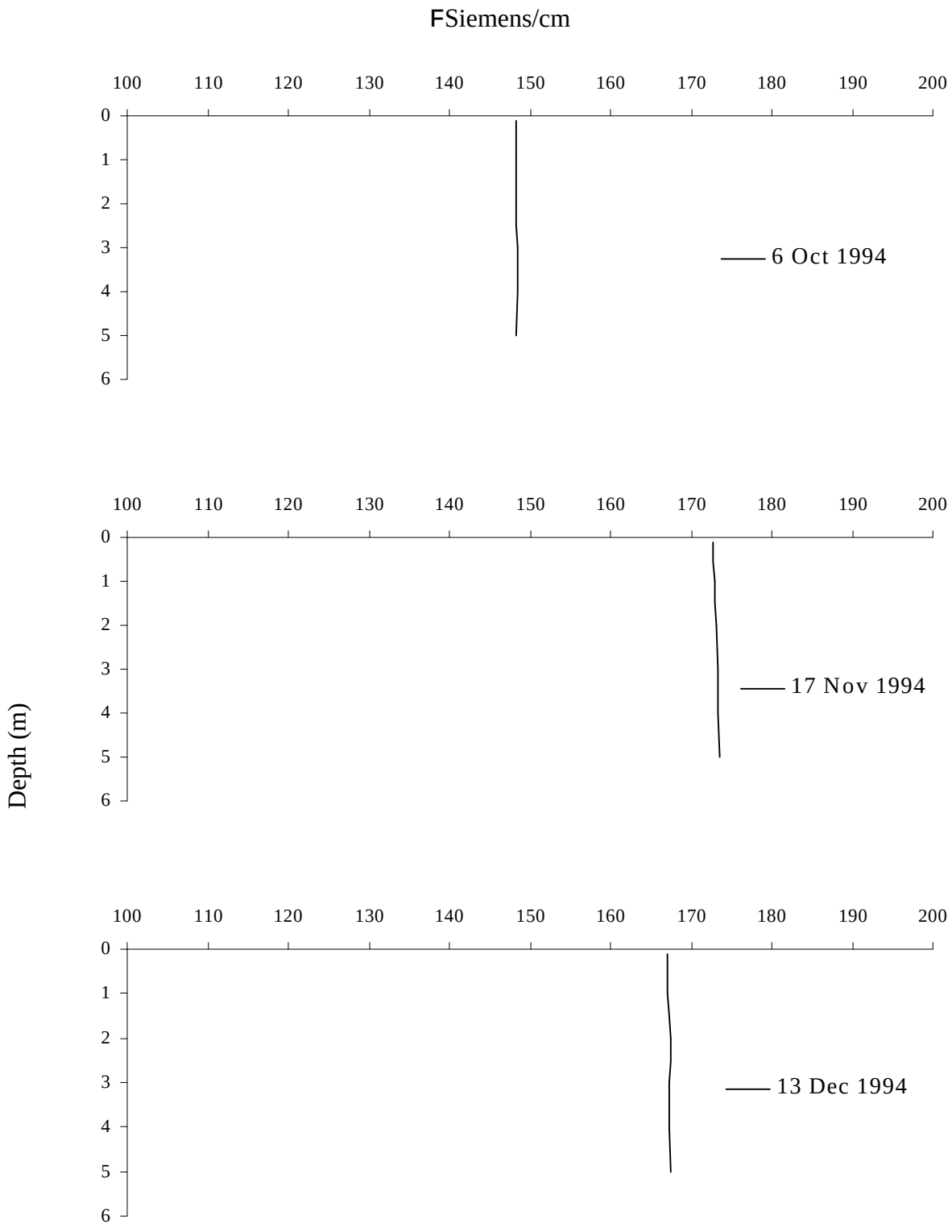
Appendix Figure B-11. Continued.



Appendix Figure B-12. John Day Reservoir specific conductance profiles. Combined data for upper-reservoir stations, April to December 1994.



Appendix Figure B-12. Continued.



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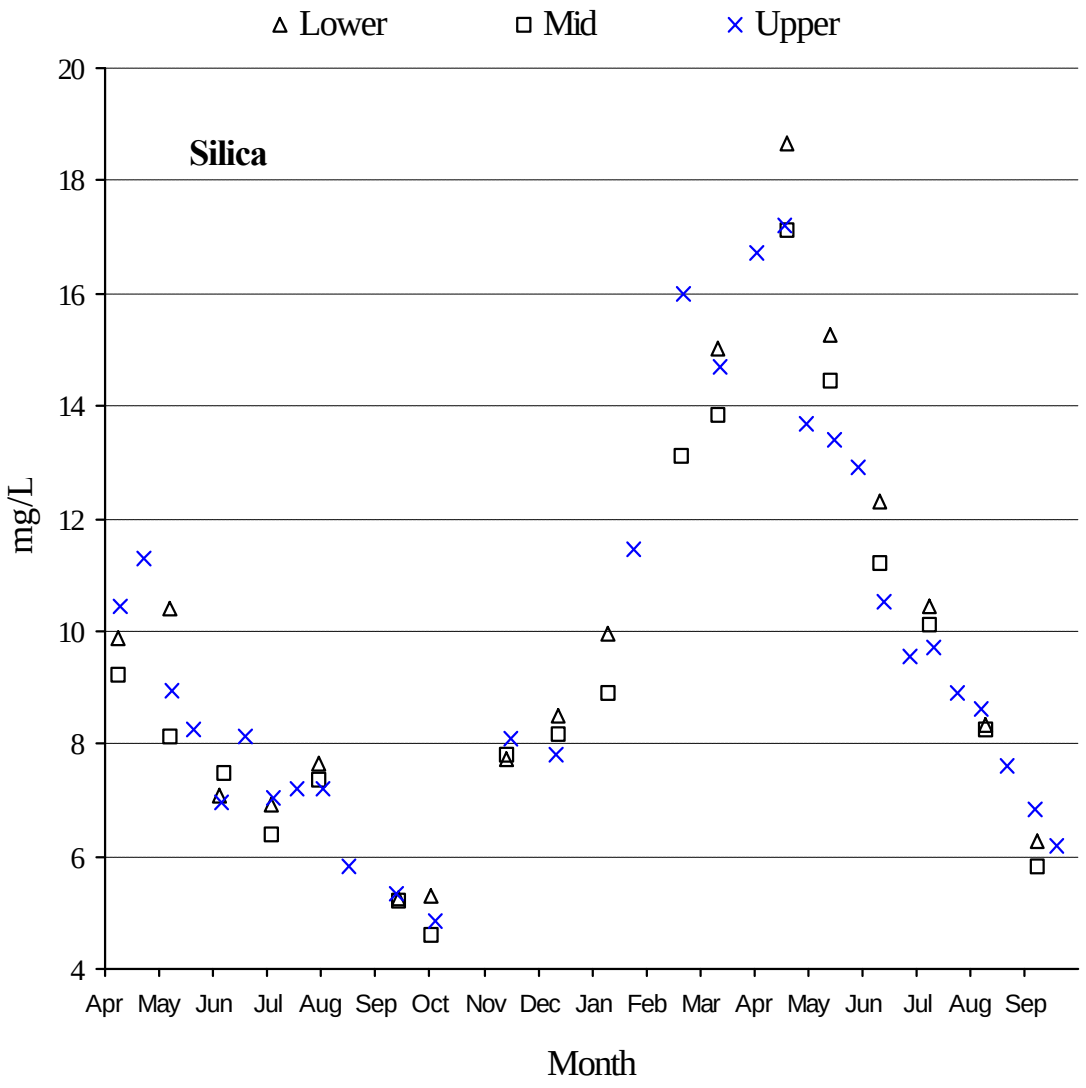
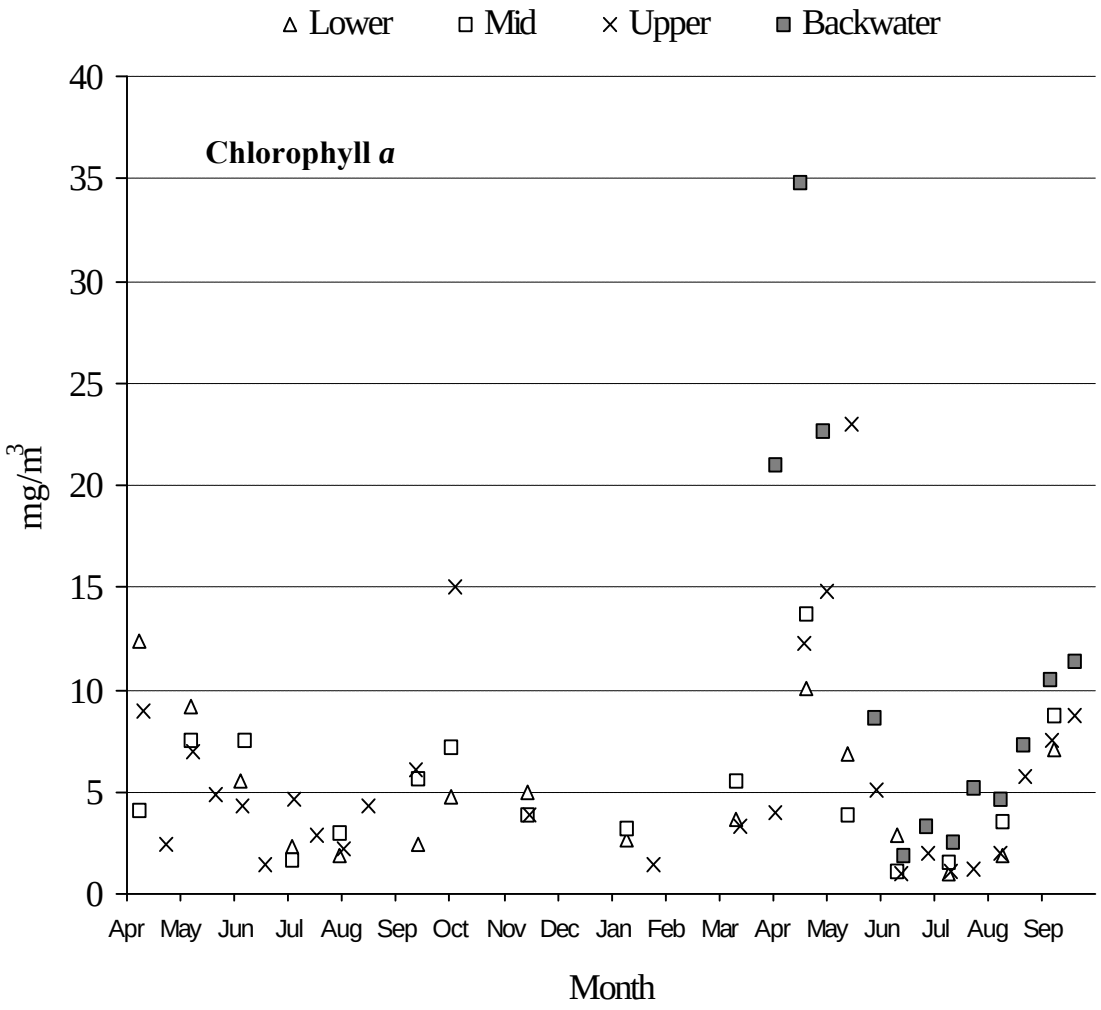
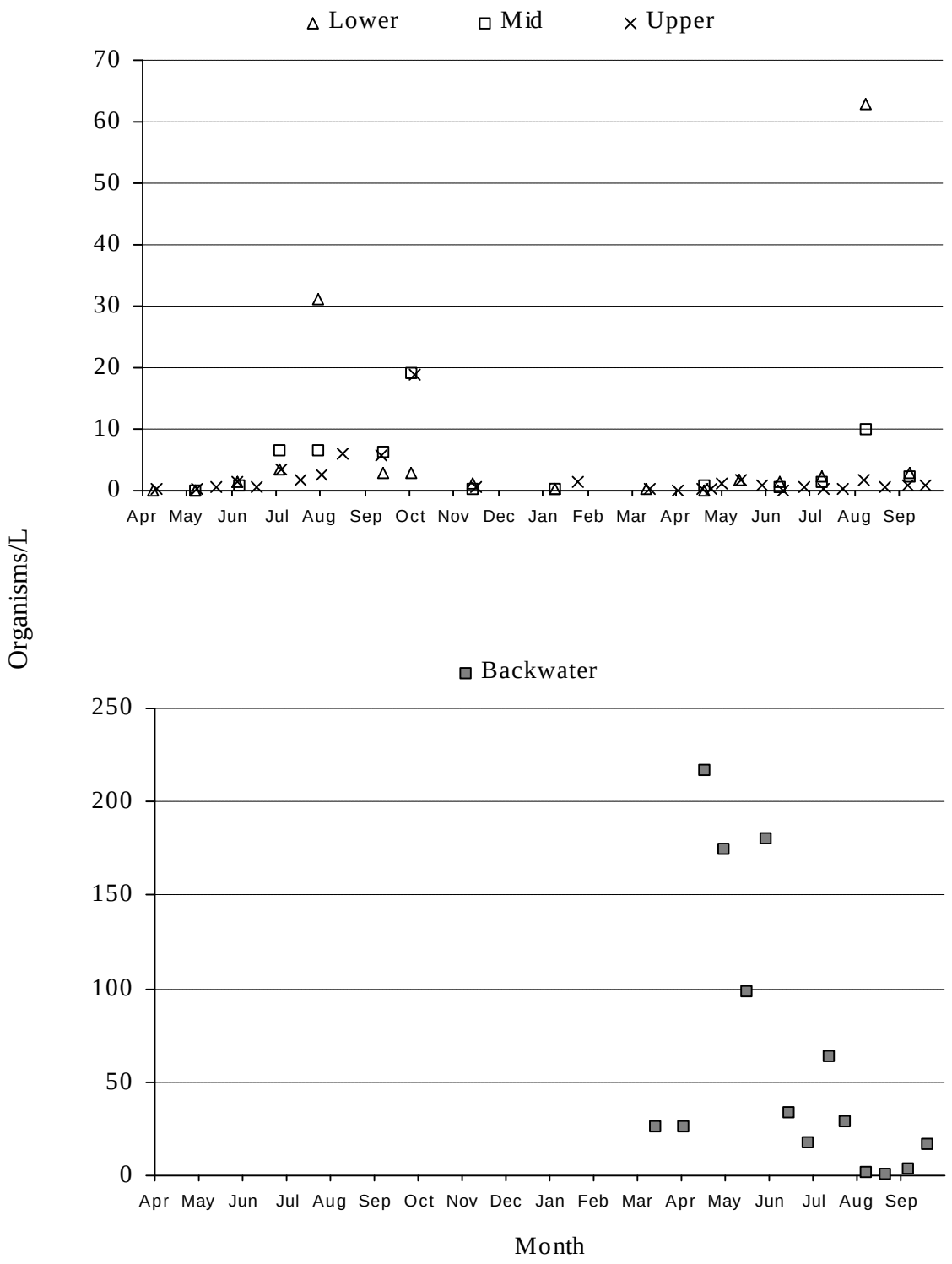


Appendix Figure B-13. Temporal and spatial variation of John Day Reservoir silica content, April 1994 to September 1995. Results from sampling at lower-, mid-, and upper-reservoir stations and at Crow Butte backwater.



Appendix Figure B-14. Temporal and spatial variation of John Day Reservoir chlorophyll *a* content, April 1994 to September 1995. Results from sampling at lower-, mid-, and upper-reservoir stations and at Crow Butte backwater.



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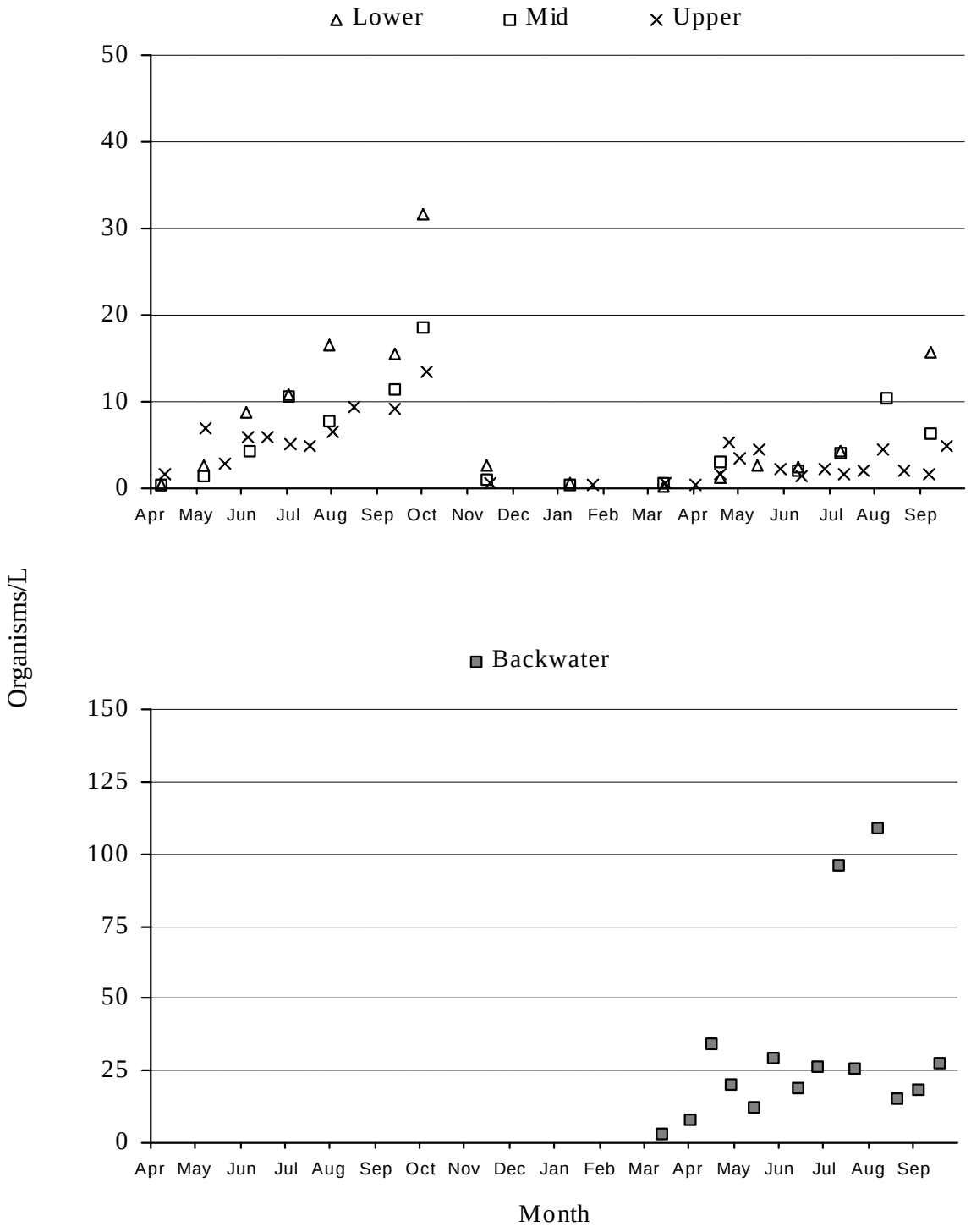

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Attachment C: *Potential Effects of Drawdown on Habitat and Associated Biota in John Day Reservoir, Columbia River.*

Potential Effects of Drawdown on Habitat and Associated Biota in John Day Reservoir, Columbia River

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ABSTRACT

Potential biological effects of water level drawdowns to 165-ft and 220-ft msl in John Day Reservoir were projected from expected habitat conditions and species requirements. I reviewed available information on the planktonic and benthic macroinvertebrate communities and some of the more abundant fishes in the reservoir to determine factors that could alter community structure under a drawdown regimen. GIS, using bathymetric and substrata data collected by the USGS and incorporated into a database, projected habitat changes under drawdown. The proposed drawdown to 165-ft MSL would alter the habitat considerably. The riverine conditions would result in higher water velocities that would radically change the substrata to larger sized material. Attached communities would comprise the bulk of the food production and the fish community would return to the historic conditions of native fishes. Minnow and sucker species would dominate the fish community and lacustrine species, many of which are introduced, would be largely eliminated, except for small populations in backwaters. Species like northern pikeminnow (Ptychocheilus oregonensis) and white sturgeon (Acipenser transmontanus) could increase in abundance while species like walleye (Stizostedion vitreum) and yellow perch (Perca flavescens) would decline. Similar changes are projected to occur under a drawdown of 220-ft msl in the upper 55 miles while in the lower 15 miles, lacustrine habitats, similar to those currently existing, would occur. I believe the fish community would differ considerably between the riverine and lacustrine sections. Minnow and sucker species also would dominate the fish community in the upper 55 miles while those in the downstream 15 miles would be more similar to current community structure of numerous introduced fishes. If drawdown were

implemented, possible studies to monitor changes in the biota should focus largely on community structure and predator-prey dynamics.

ACKNOWLEDGEMENTS

I thank the US Army Corps of Engineers, Portland, Oregon for funding this project. A myriad of people was extremely helpful to me in conducting this study. Messrs. Michael Parsley, Thomas Poe, and James Petersen, all from the USGS laboratory at Cook, Washington provided information on John Day Reservoir. Mike Parsley contributed greatly by providing GIS data, Tom Poe provided copies of previous reports, and James Petersen provided insight into predator – prey relationships and potential changes in the fish community. Josh Hayes, University of Washington, provided estimates of water particle travel time. I also thank Lyle Gilbreath, National Marine Fisheries Service, for providing limnological data. Thanks are extended to Mr. Troy Merrill, Ms. Nancy Wright, and Ms. Eva Strand for GIS work. Ms. Strand's efforts were especially valuable in "fine tuning" the printed GIS maps. Mr. Mark Smith and Mr. Chuck Willis, US Army Corps of Engineers, provided valuable contract support. Additional thanks are extended to Mike Parsley, Craig Barfoot, and Tom Poe for critical review and helpful comments on an earlier draft.

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INTRODUCTION

Construction of Bonneville Dam in 1933 and Grand Coulee Dam in 1939 initiated conversion of the Columbia River system to a series of reservoirs. The resulting reservoirs created a variety of water-related development opportunities including navigation, flood control, electrical power generation, and recreation. These projects resulted in substantial changes in the natural habitat and biota from riverine to river-lacustrine. In 1968, John Day Dam was completed, which created the second largest impoundment on the Columbia River.

Habitat created by the construction of John Day Dam is dramatically different from the original riverbed and floodplain. The converted river has the main channel area plus a number of backwaters and wetlands. These "created" aquatic habitats provide a diversity of opportunities for aquatic organisms not available in the original river channel. In addition to an altered aquatic habitat, natural patterns of spring flows from upstream snowmelt have been modified into a more stable hydrograph with reduced water velocities. Altered habitat conditions have favored development of shallow water biota and have placed additional stresses on downstream migrating salmonid fishes (*Oncorhynchus* spp.). The associated changes in the fish community and increased mortality of downstream migrating salmonid smolts have spurred interest in returning reservoirs to "river like" conditions. One such proposal is to draft Columbia River reservoirs to lower elevations during the salmonid smolt out-migration.

Little information is available on effects of drawdown on Western reservoirs. Weitkamp and Sullivan (1993) reported 18 risks associated with the proposed drawdown of John Day Reservoir to minimum operating pool. Those risks associated with the John

Day ecosystem included liabilities to resident and anadromous fishes. A test drawdown of Lower Granite Reservoir, Snake River, conducted in March 1992 provides limited information on possible effects of a John Day drawdown (Wik et al. 1993). Short-term effects were invertebrate desiccation, limited fish entrainment, and fish stranding. The Lower Granite drawdown, however, was limited to 1 month (March) and coincided with a time of restricted biological activity as a result of low flows in early spring and low water temperatures. Drawdown at another time of year may have much different biological effects. The purpose of this report is to assess effects of the possible permanent drawdown on the biota and habitat in John Day Reservoir. Specific objectives of this study were to:

OBJECTIVES

1. Review existing information on shallow water habitat, fishes, and habitat associations in John Day Reservoir;
2. Analyze effects of drawdown to 220 and 165 ft MSL elevation on fishes and associated habitat in John Day Reservoir; and
3. Develop recommendations for future studies to provide necessary data and monitor effects of drawdown.

STUDY AREA

Construction of John Day Dam on the Columbia River created the second largest reservoir (20,235 ha) in the Columbia River system. The reservoir extends from river mile (RM) 216 to RM 291 (river kilometer, Rkm, 348-469) and forms the north-south

border between Oregon and Washington (Figure 1).

John Day Reservoir attains full pool at 268 ft and minimum pool at 257 ft. Other elevations of significance are the minimum irrigation pool level at 262.5 ft and spillway crest at 210 ft. No specific project authorization currently exists for irrigation (Anonymous 1994).

John Day Ecosystem

Water Quality

Prior to 1994, limited water quality information was available on John Day Reservoir. Because of the sequence of reservoirs on the Columbia River, water quality in John Day Reservoir is largely related to upstream inflows. Gilbreath et al. (1999) conducted a comprehensive limnological survey in John Day Reservoir. Their data for 1994 and 1995 demonstrate the overall water quality characteristics of John Day Reservoir.

Water temperature.- Water temperatures in John Day Reservoir during April 1994 averaged about 10° C and increased through August with a mean temperature of 22° C (Figure 2). Water temperatures declined slowly from August through October 1994 and then declined rapidly from October through January 1995 when the lowest temperatures were recorded (4° C). Widest daily range (3.2° C) in water temperature occurred in July and ranged from 17.9 to 21.2° C. Water temperatures through September 1995 exhibited a similar pattern as in 1994. Water temperatures were similar between RM 219 (Rkm 353) and RM 279 (Rkm 449). Mean water temperatures for the period of record averaged about 14° C in the lower to mid reservoir area (RM 219-243; Rkm 353-

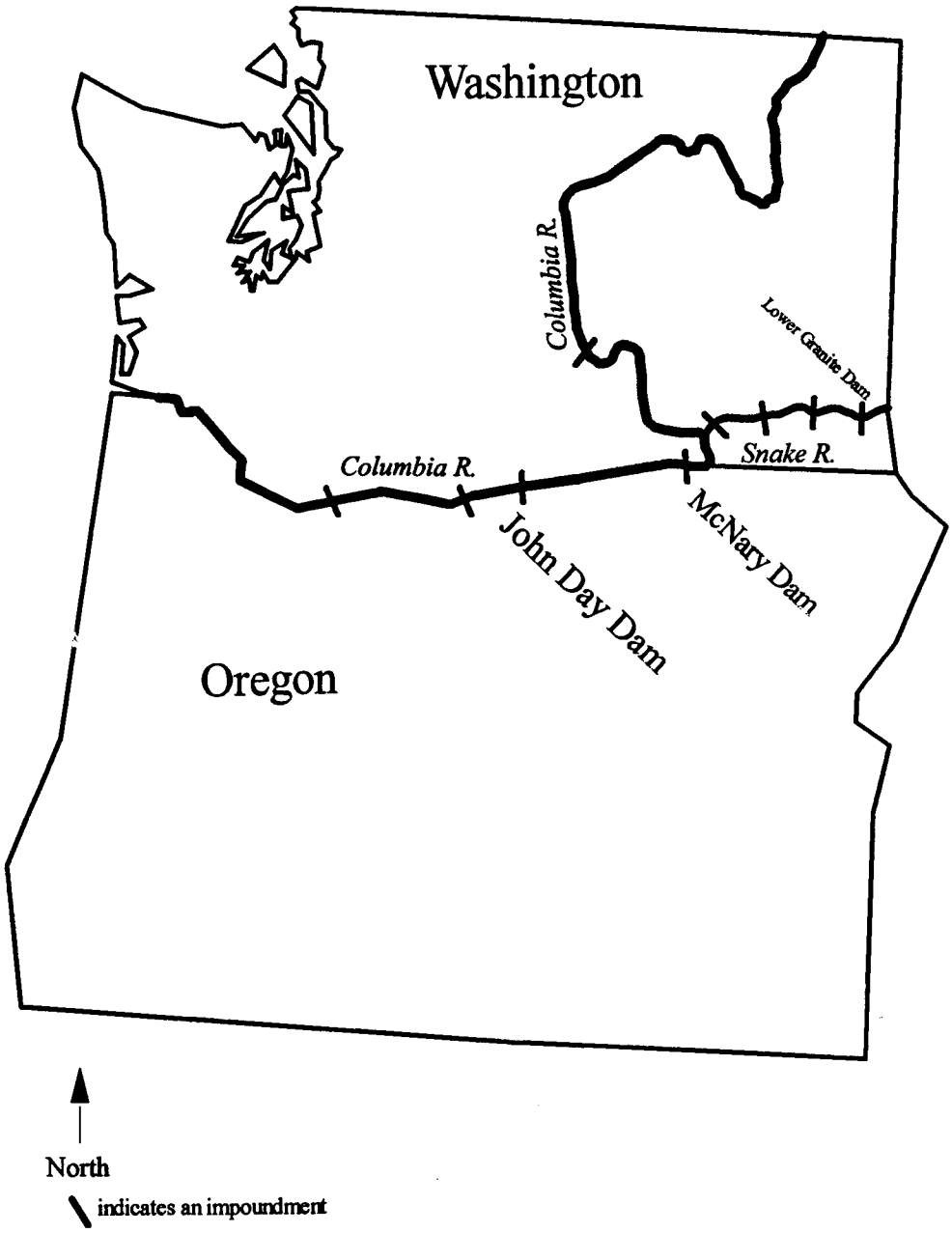


Figure 1. Location of John Day Reservoir and dams on the lower Columbia and Snake rivers.

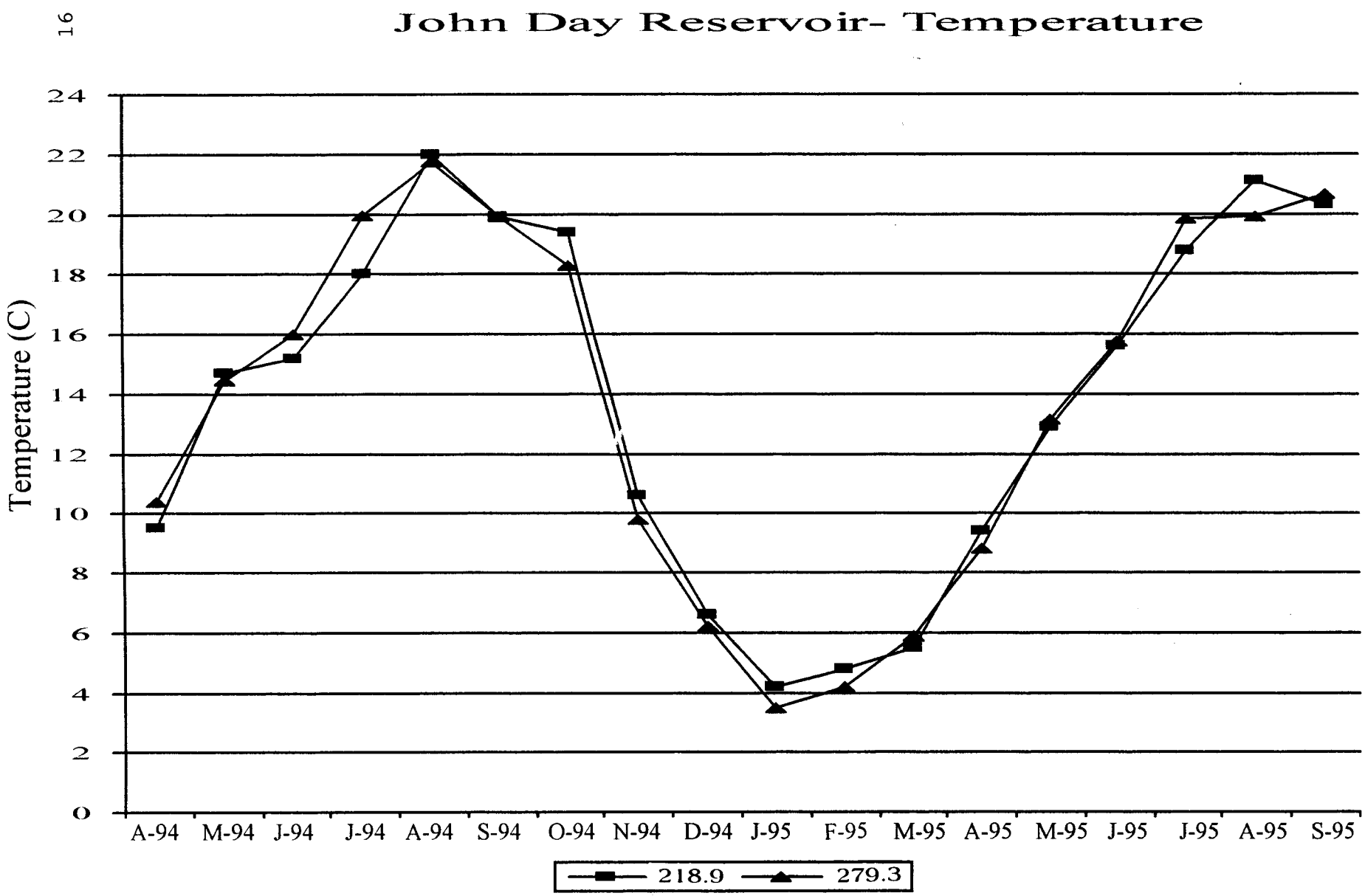


Figure 2. Monthly mean water temperatures measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

391), about 14.5° C in the upper reservoir (RM 275-279; Rkm 443-449), and were highest in the backwater (RM 264; Rkm 424; Figure 3). Mean water temperatures during April through July, the time of the majority smolt out-migration range from about 9 to 18°C.

Dissolved oxygen.- Mean monthly dissolved oxygen (DO) concentrations ranged from about 10 mg/L to 11 mg/L throughout John Day Reservoir (Figure 4). The highest dissolved oxygen was measured in the backwater at RM 264 (Rkm 424). Dissolved oxygen concentrations were highest in March and April at RMs 219 and 279 (Rkm 353 and 449), and then it declined slightly through December (Figure 5). The widest monthly variation in dissolved oxygen (8 to 12.3 mg/L) occurred in May 1994.

Conductivity.- Mean monthly water conductivity was highest in April, lowest in June, and gradually increased through January and ranged from about 130 to 185 μS/cm (Figure 6). The widest monthly variation in conductivity was recorded in June 1994 and ranged from 128 to168 μS/cm. Conductivity in the backwater at RM 264 (Rkm 424) was the lowest in both annual mean and range (Figure 7). Annual means were slightly lower (5 μS/cm) in the upper reservoir than in the lower reservoir.

Alkalinity.- Mean monthly alkalinity ranged from about 75 mg/L as $CaCO_3$ in February and March 1995 to <50 mg/L in June 1995 (Figure 8). Variation in alkalinity within each month was lowest in September and October and this pattern of variation was similar in other months. Slight differences in alkalinity occurred between RM 219 and 279 (Rkm 352 and 449), with higher values at the downstream station. Mean annual alkalinity averaged about 2-3 mg/L as $CaCO_3$ lower in the upper reservoir stations than those in the lower to mid-reservoir stations, with those in the backwater, RM 264 (Rkm 424) intermediate (Figure 9).

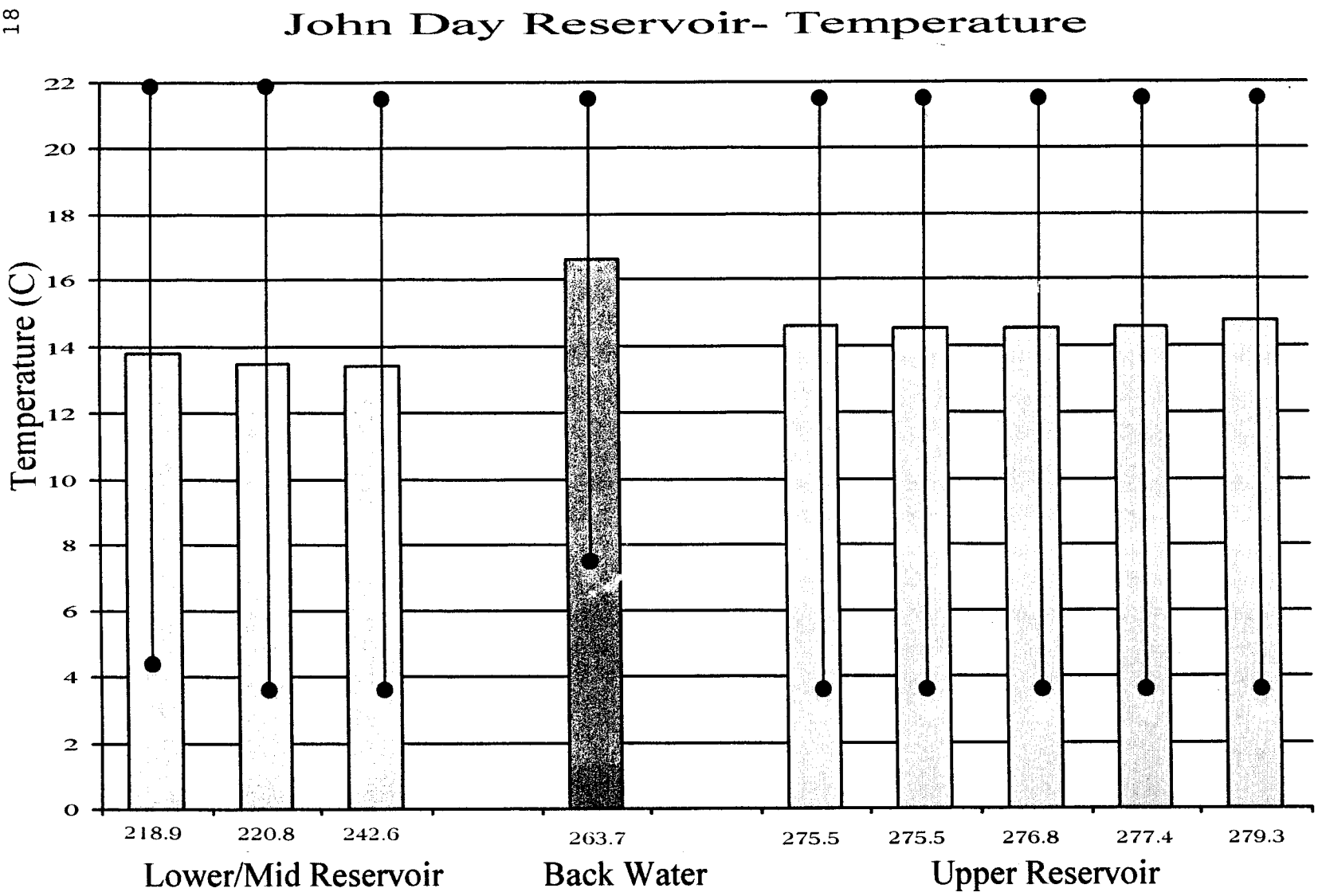


Figure 3. Annual mean and range of water temperatures measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

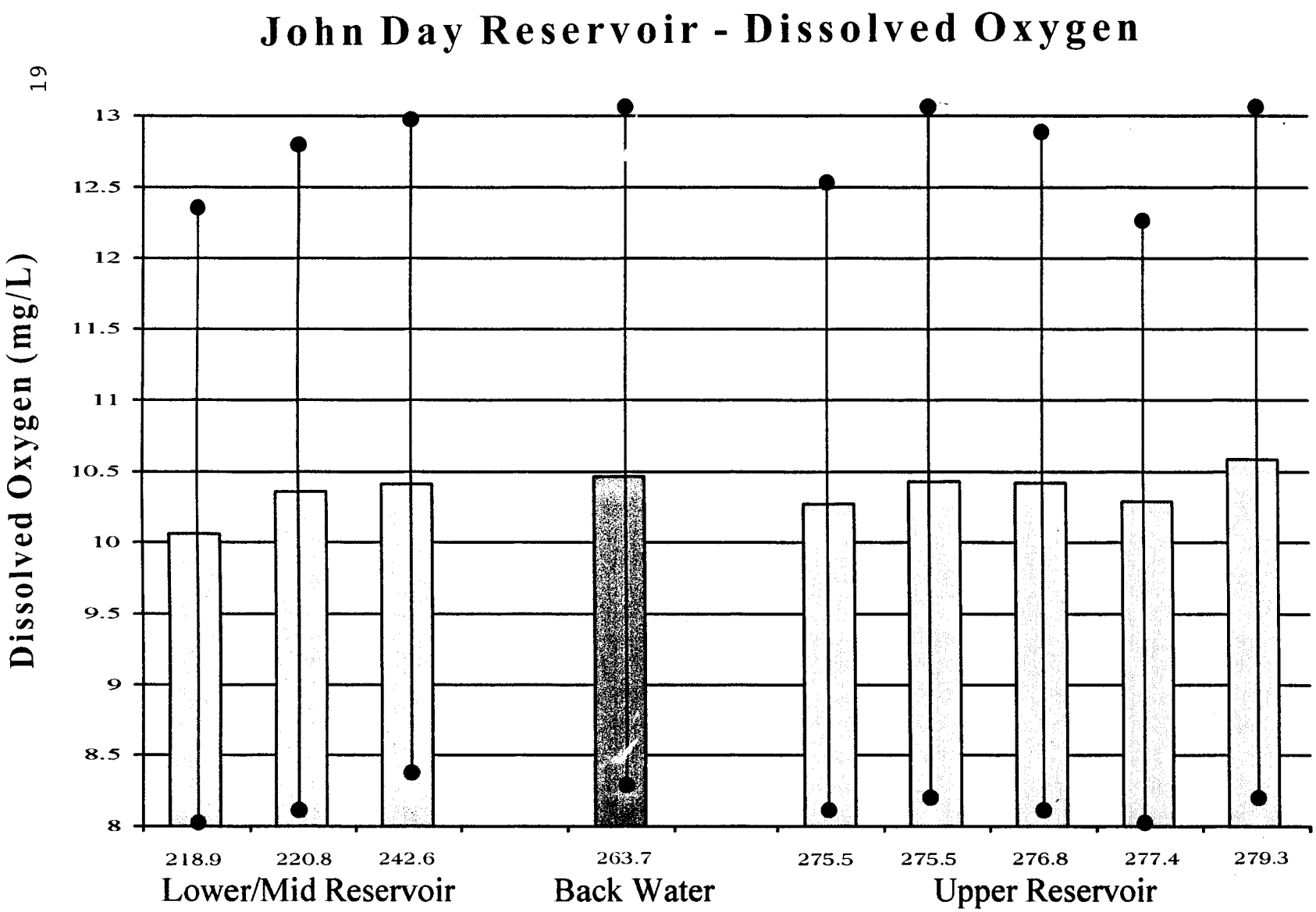


Figure 4. Annual mean and range of dissolved oxygen measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

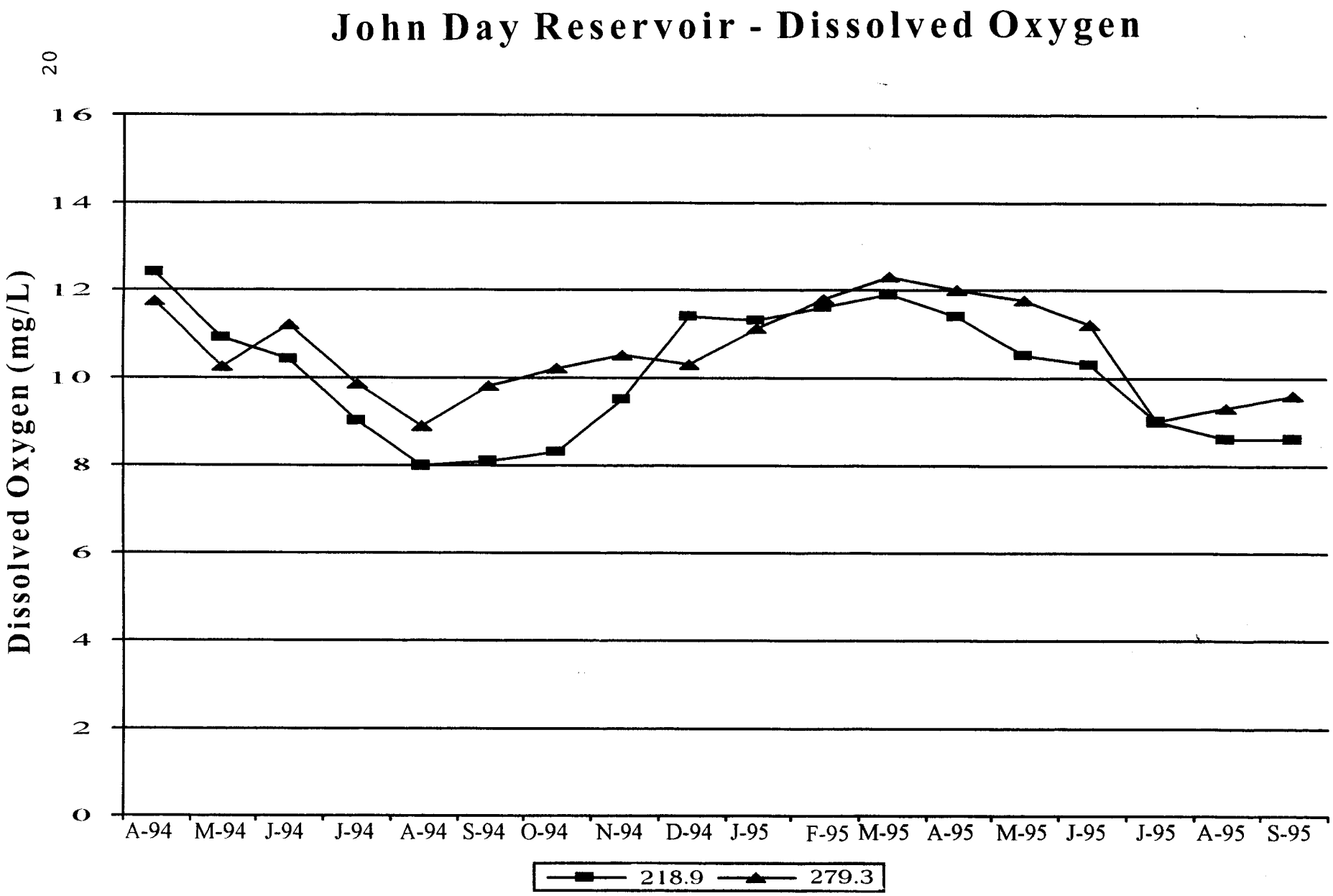
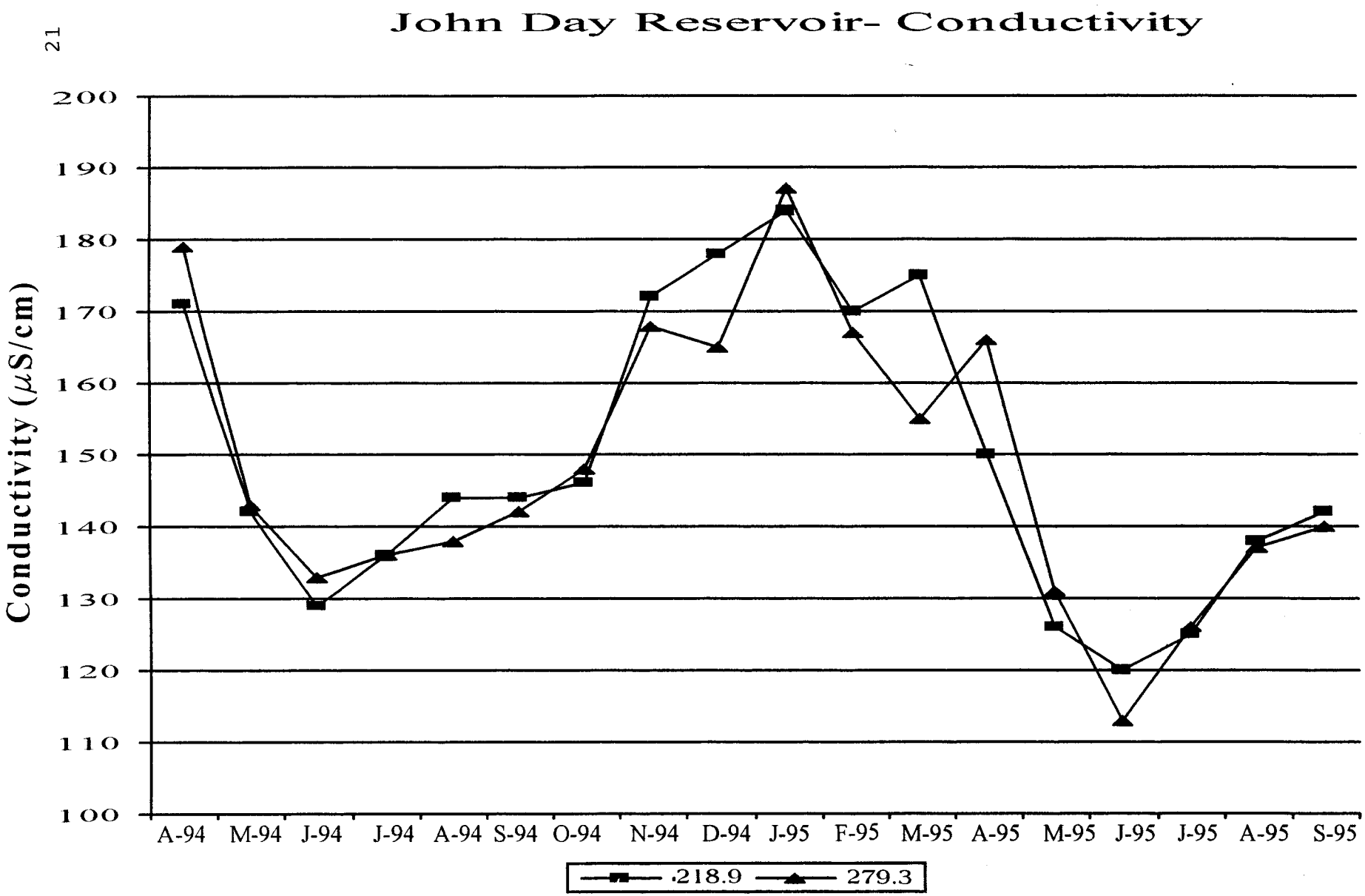


Figure 5. Monthly mean dissolved oxygen measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day reservoir, Columbia River.



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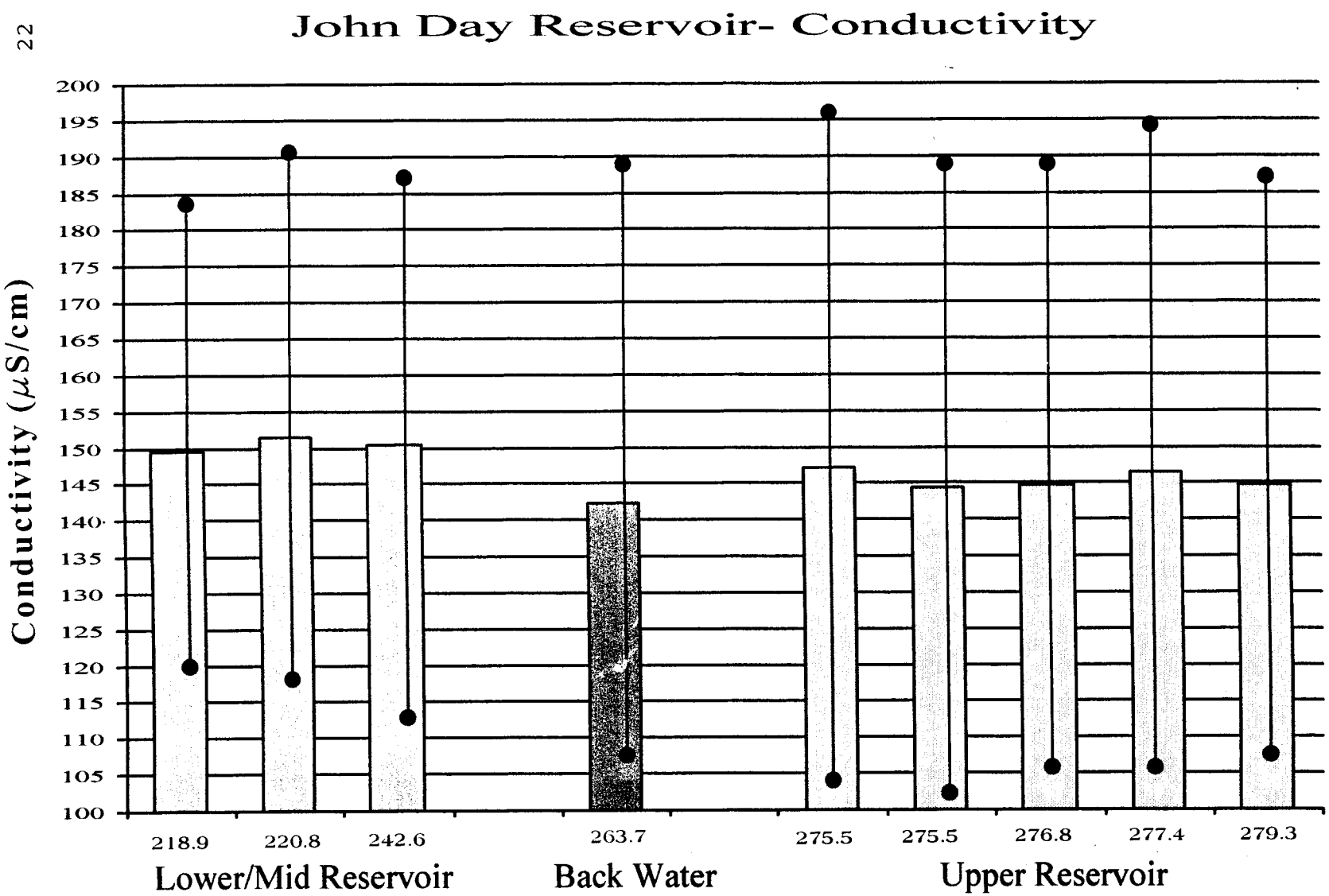


Figure 7. Annual mean and range of conductivity measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 from March 1995 to September 1995.

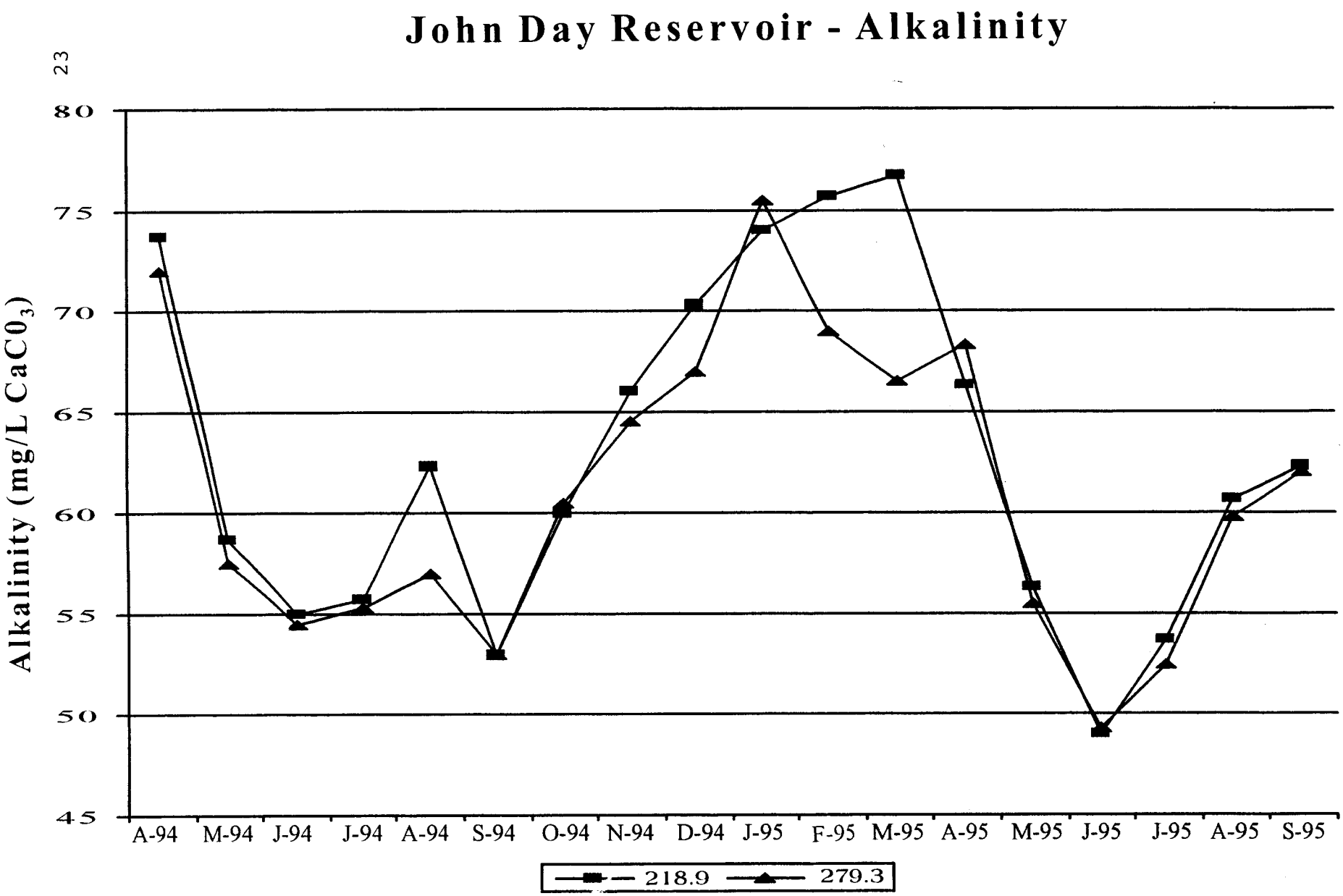


Figure 8. Monthly mean alkalinity measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

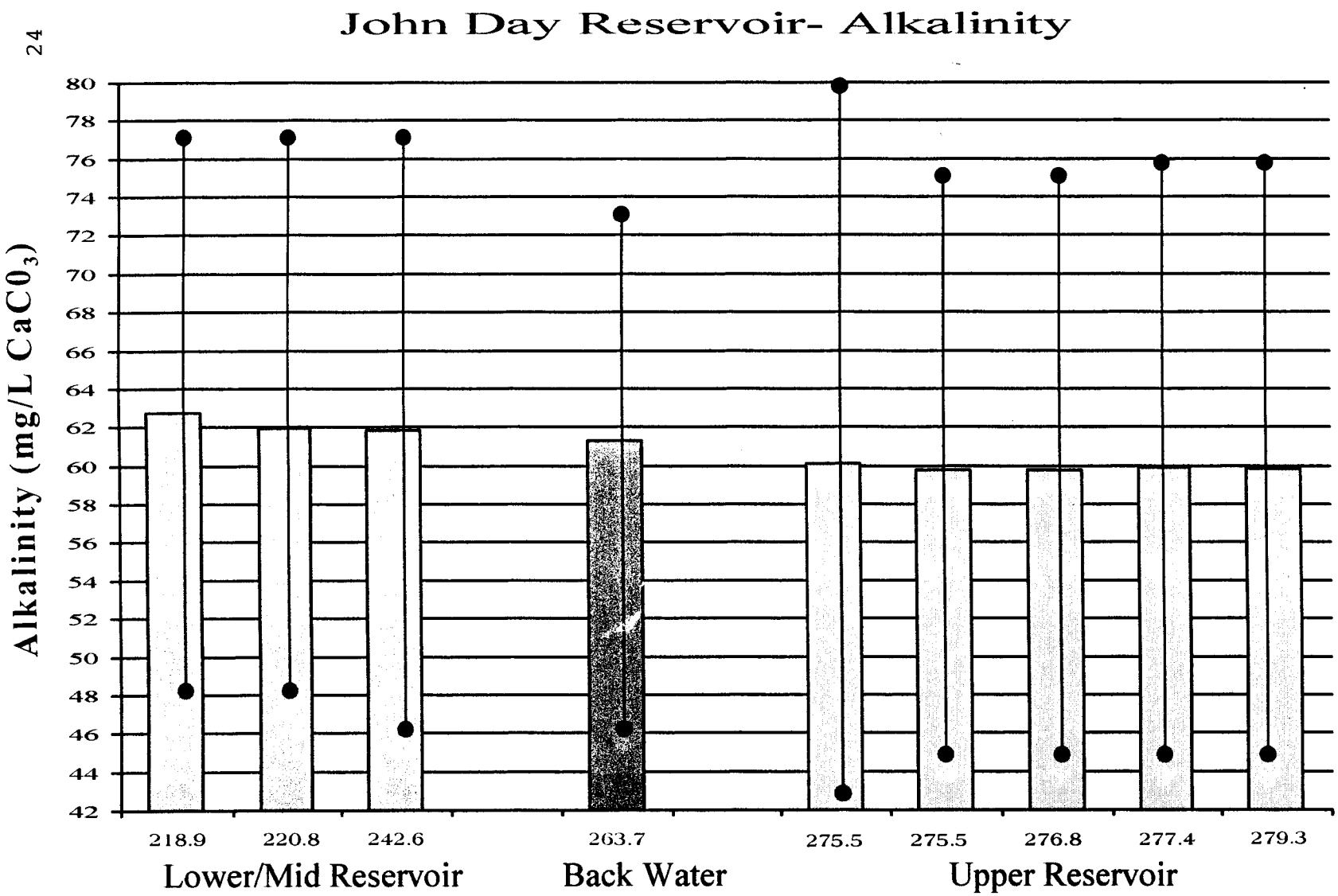


Figure 9. Annual mean and range of alkalinity measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

Phosphorus.- Mean concentrations of phosphorus ranged from about 40 to 45 μg/L in the lower to mid reservoir and were more highly variable in the upper reservoir, ranging from 33 to 47 μg/L (Figure 10). Concentrations of phosphorus averaged about 43 μg/L in the backwater at RM 264 (Rkm 424); this was an intermediate concentration relative to selected stations in both the upper and lower reservoir. Throughout the sampling period, concentrations were highest in the winter and lowest during the period from July through August (Figure 11). Monthly changes in phosphorus concentrations were similar, although concentrations were generally higher at RM 219 (Rkm 352) than RM 249 (Rkm 449).

Nitrogen.- Mean nitrogen levels generally decreased from April 1994, remained level in September 1994, and increased from October 1994 through April 1995 (Figure 12). In general, nitrogen concentrations were higher at RM 219 (Rkm 352) than at RM 279 (Rkm 449). All concentrations of nitrate measured were generally <1.0 mg/l, and the overall highest annual mean concentration was in the backwater at RM 264 (Rkm 424; Figure 13). Nitrogen concentrations were similar between upper reservoir and lower reservoir sampling stations.

Turbidity.- Mean monthly turbidity peaked in February 1995 near 40 NTU and decreased throughout the remaining months of sampling in 1995 (Figure 14). Mean turbidity throughout the period of April to December 1994 was low (~5 NTU) although higher (5-11 NTU) during the period from April through September 1995. Turbidity was generally similar among upper reservoir and lower reservoir stations and the backwater at RM 264 (Rkm 424; Figure 15). The widest range in turbidity was measured at RM 218.9 (Rkm 352) and RM 275.5.

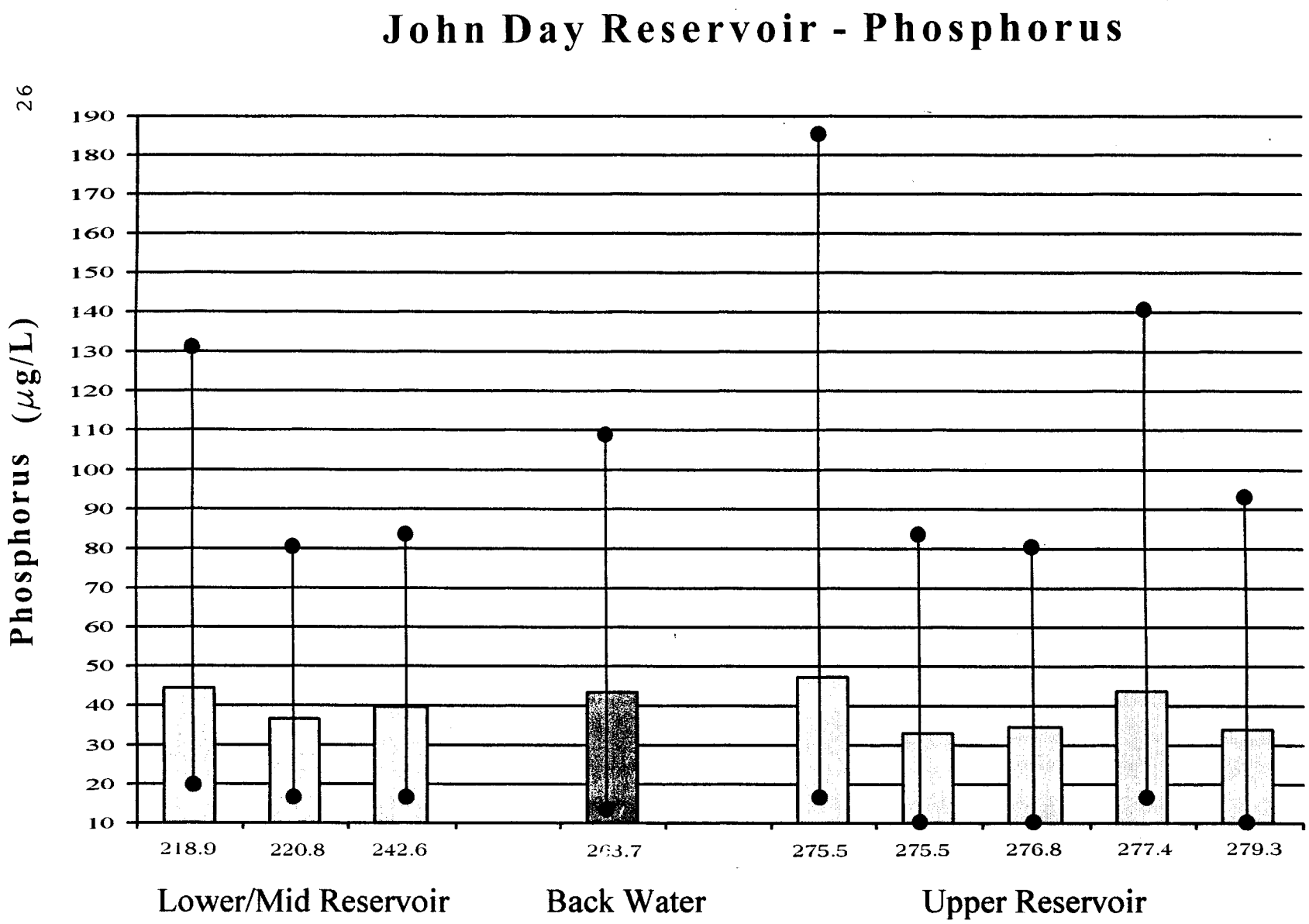
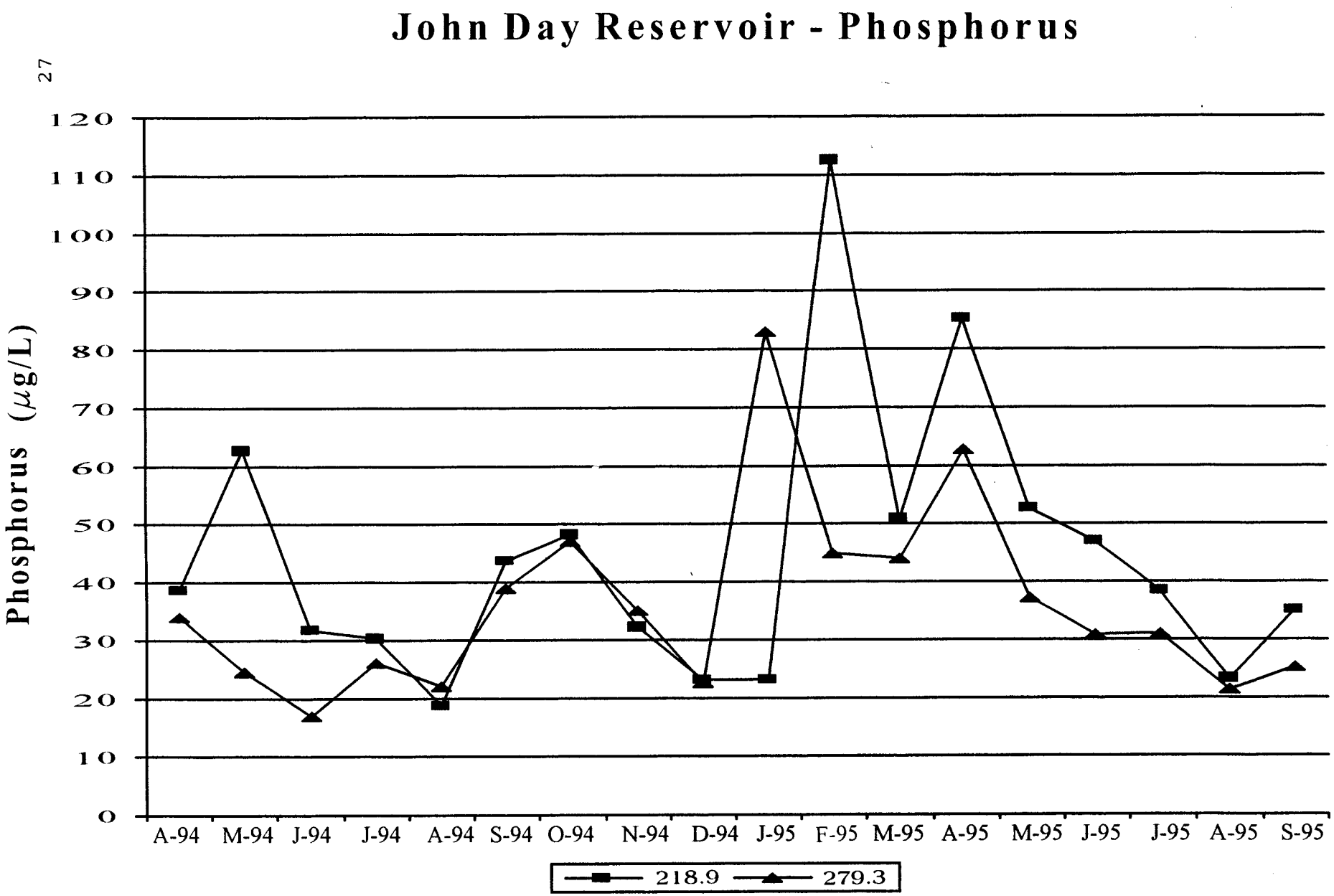


Figure 10. Annual mean and range of phosphorus measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.



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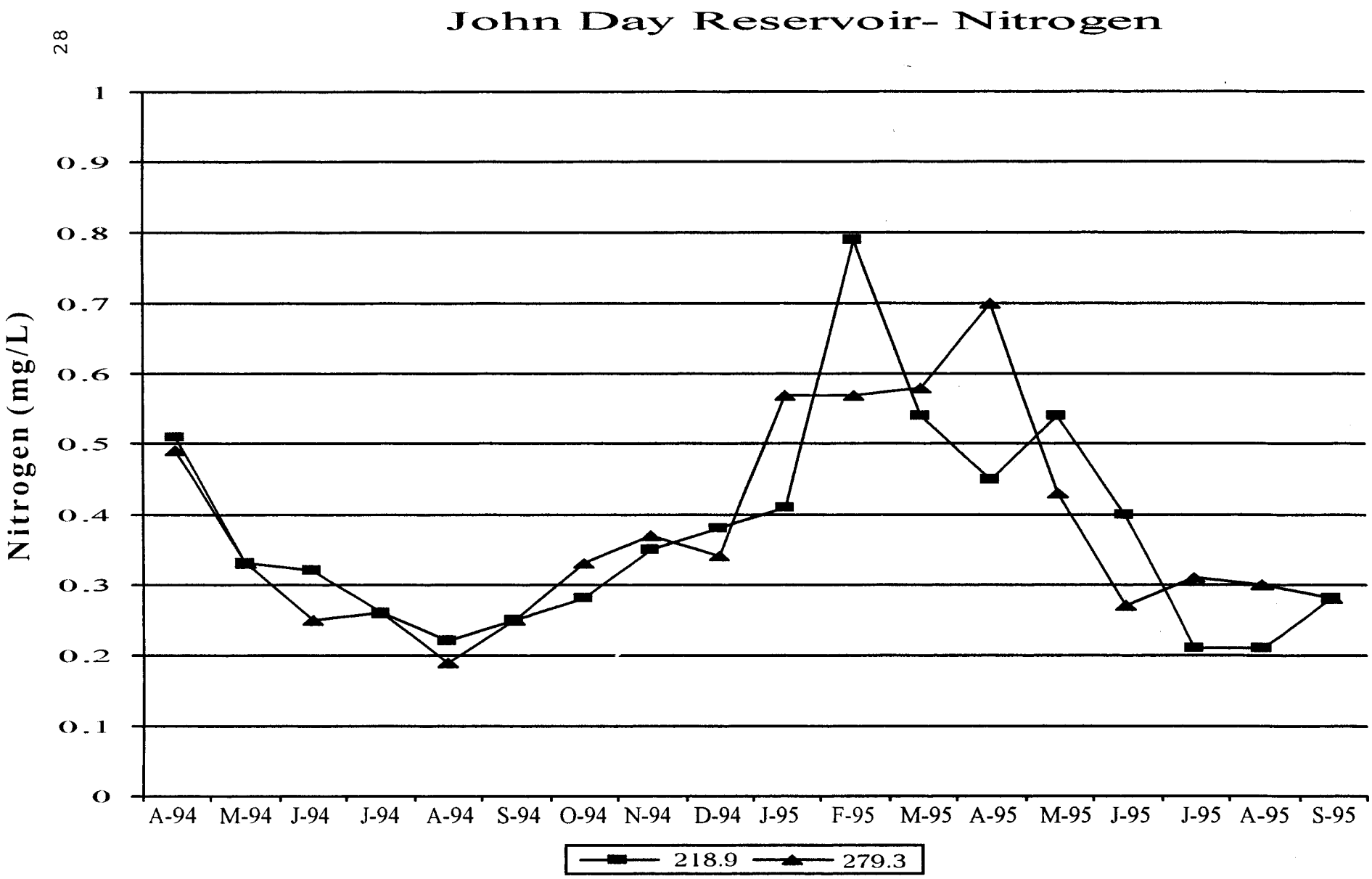

Figure 12. Monthly mean nitrogen measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

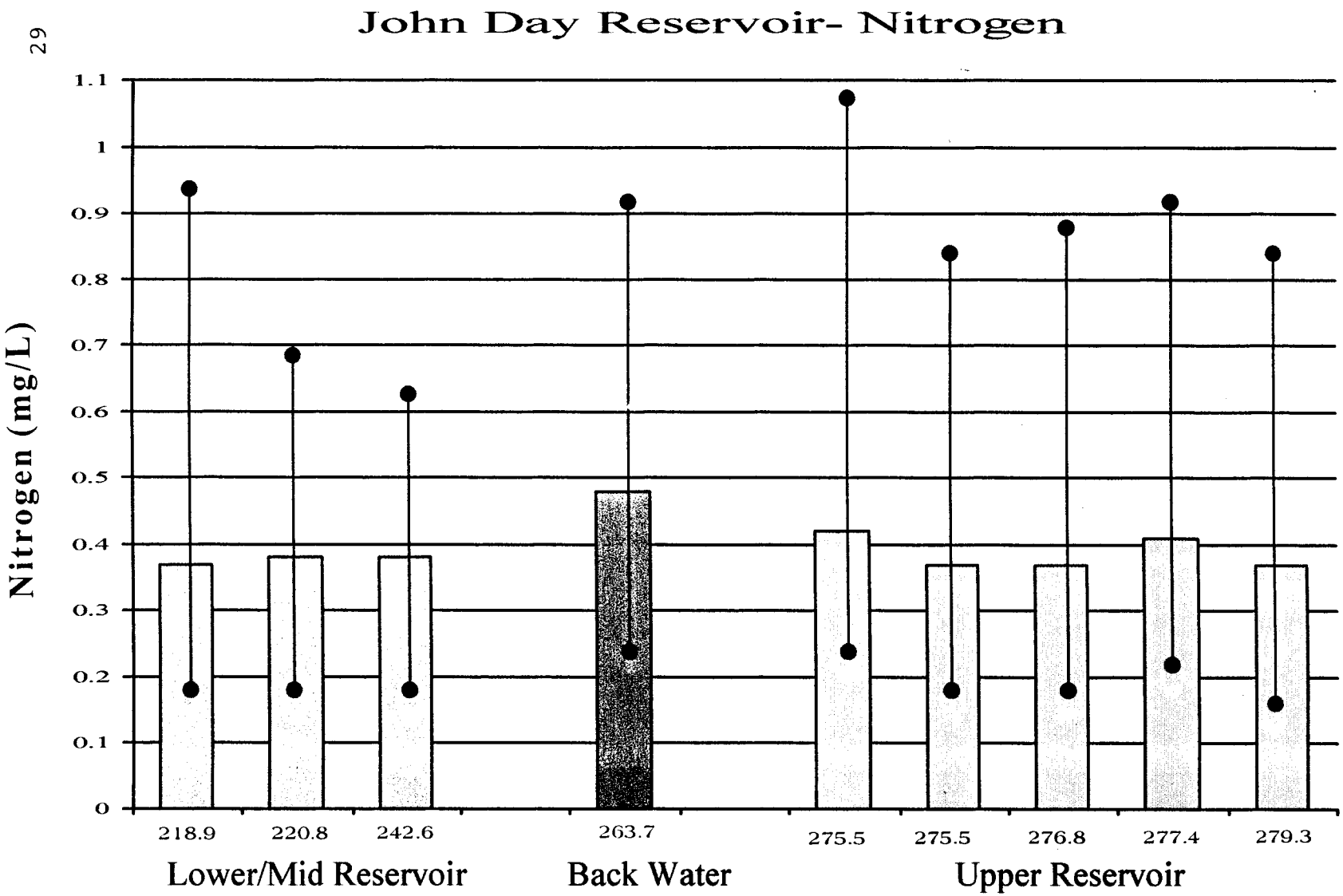

Figure 13. Annual mean and range of nitrogen measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

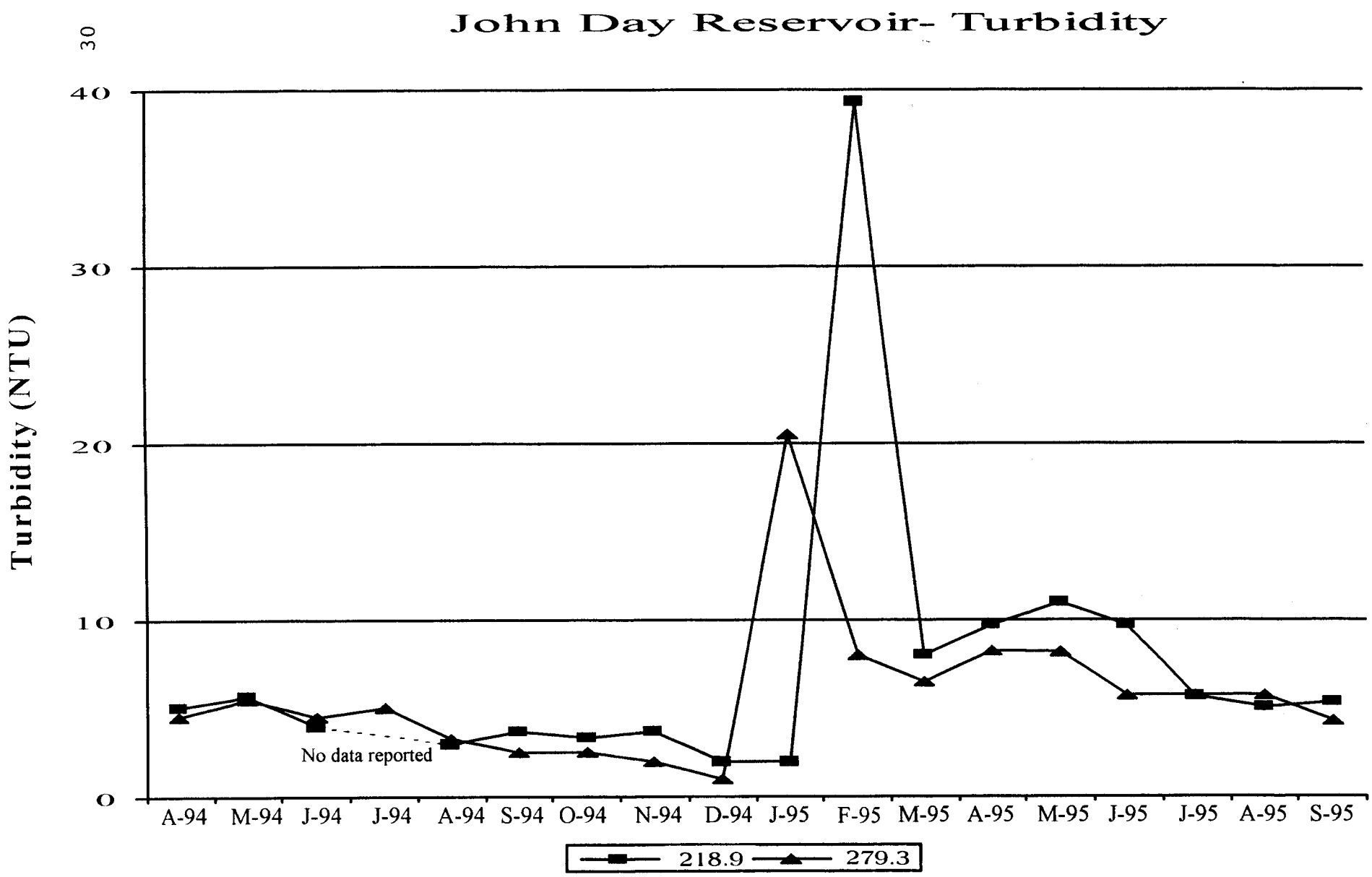

Figure 14. Monthly mean turbidity measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

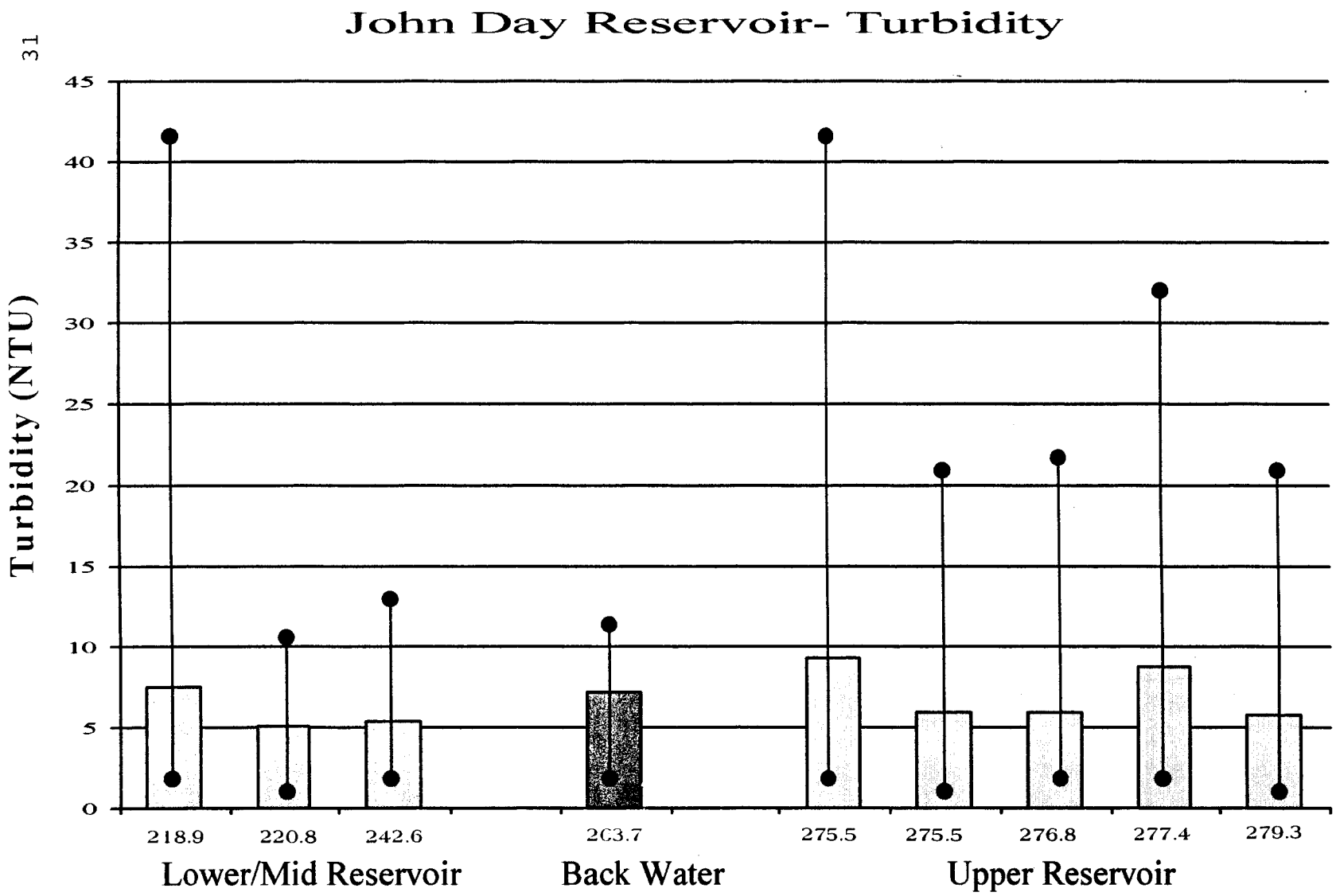

Figure 15. Annual mean and range of turbidity measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

Secchi depth.- Mean monthly Secchi depths exhibited a gradual increase from April to December 1994, and attained their lowest values in January and February 1995 (Figure 16). The highest mean monthly Secchi depth exceeded 4 m. Annual mean Secchi depth was slightly lower (~0.2 m) in the upper reservoir than the lower reservoir, and it was lowest in the backwater at RM 264 (Rkm 424; Figure 17). Secchi depths during the smolt out-migration were generally between 1 and 2 m.

Chlorophyll a.- Chlorophyll a, an index of primary productivity, was variable in John Day Reservoir but generally similar annually from upper reservoir and lower reservoir stations (Figure 18). Highest annual mean chlorophyll a and the widest range of concentrations occurred in the backwater at RM 264 (Rkm 424). Mean monthly chlorophyll a varied throughout the sampling period and was generally higher at RM 279 (Rkm 449) than RM 219 (Rkm 352; Figure 19). Higher concentrations were measured in the spring and fall, whereas those measured at other times of the year were generally low (<5 mg/m^2).

pH.- pH values in John Day Reservoir indicated that waters were more basic than acidic. Highest annual pH values were measured at the backwater (RM 264; RKm 424) and averaged about 0.4 pH units higher than those measured in the upper and lower reservoir (Figure 20). Values of pH showed a decreasing trend, as they were higher in April 1994 and gradually decreased through July 1995, after which they increased about 0.5 pH units (Figure 21). Values of pH were generally similar between sampling stations at RM 219 and 279 (Rkm 352 and 449).

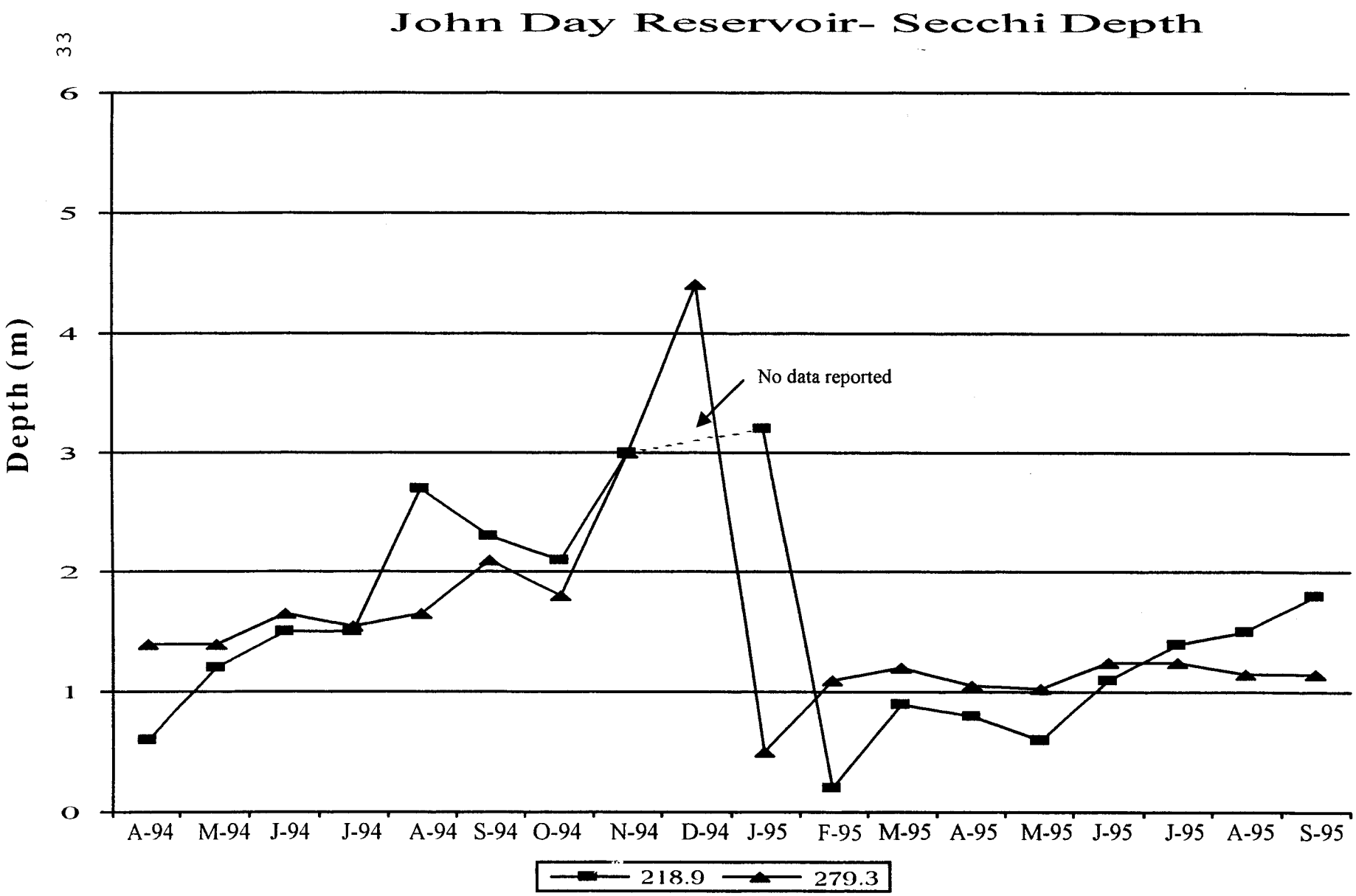

Figure 16. Monthly mean secchi disk depth measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

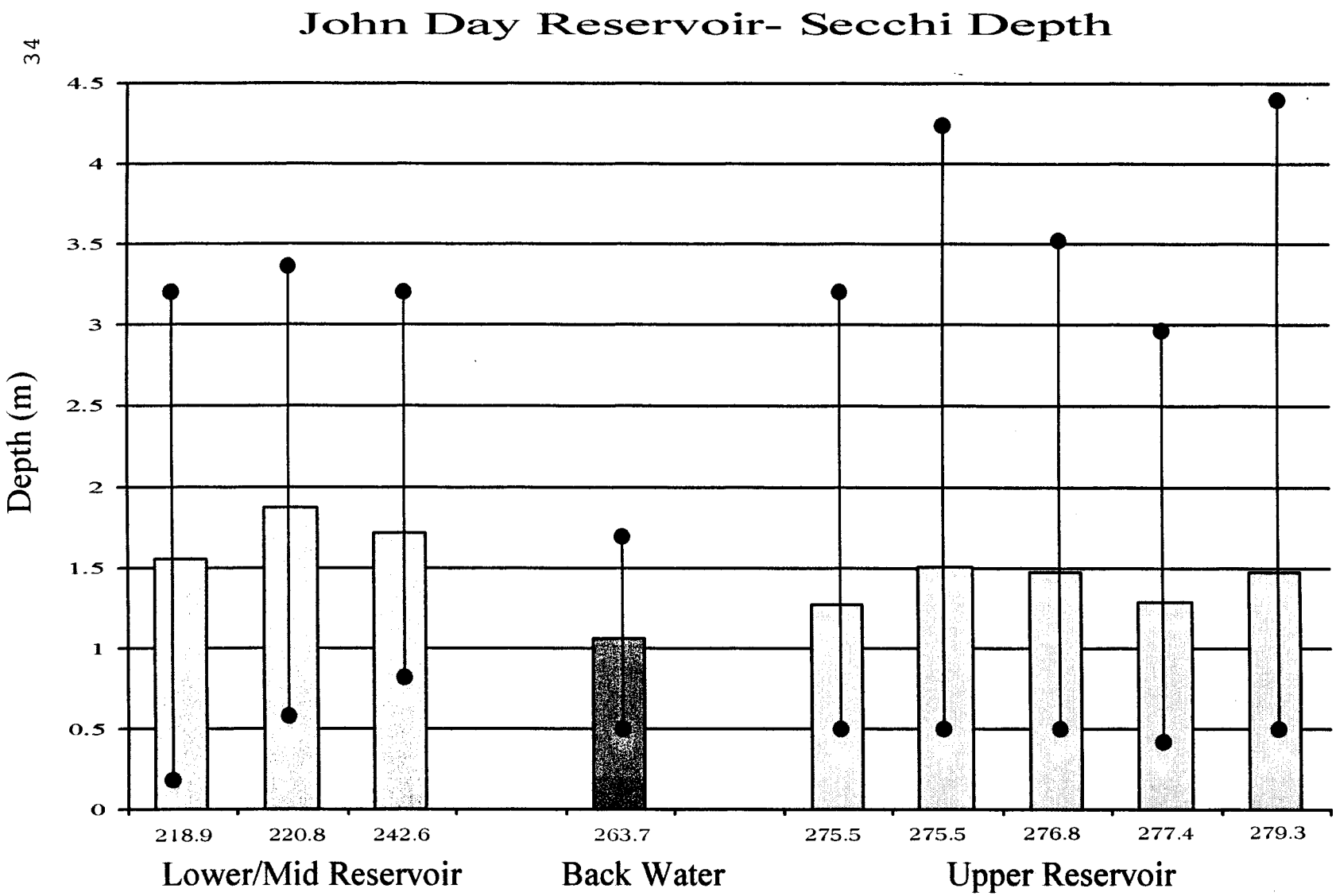

Figure 17. Annual mean and range of secchi disk depth measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

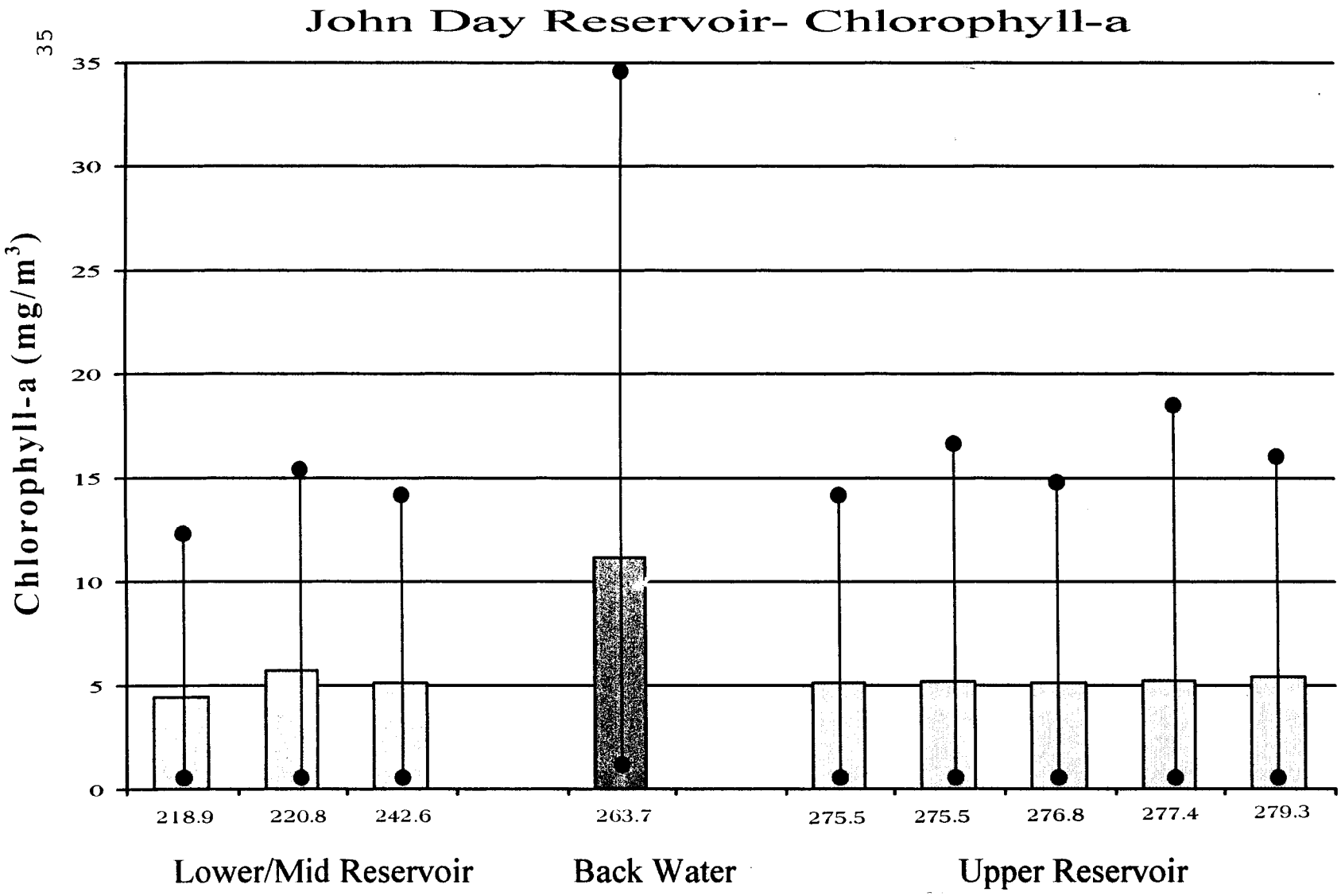

Figure 18. Annual mean and range of chlorophyll-a measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

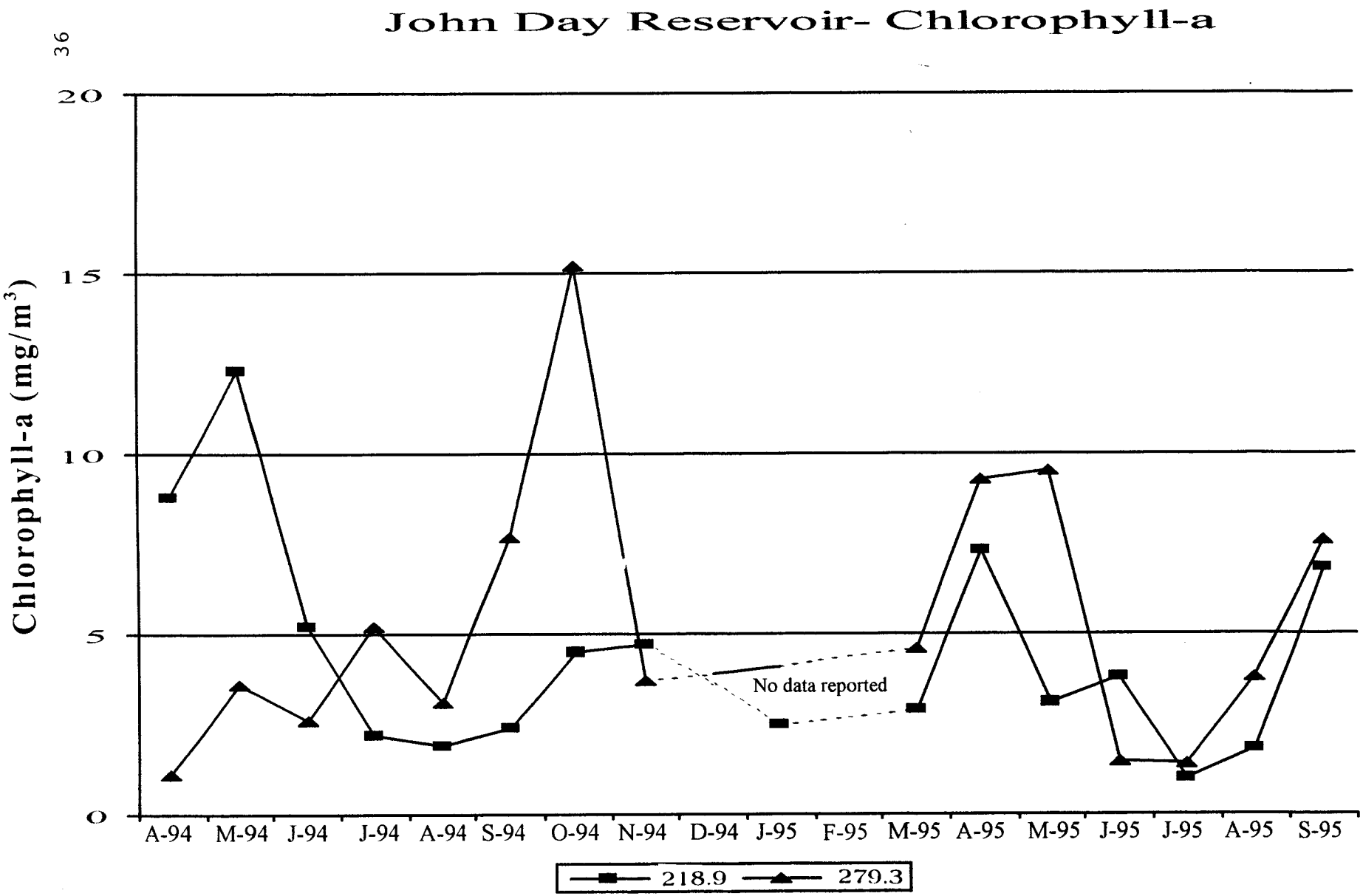

Figure 19. Monthly mean chlorophyll-a measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

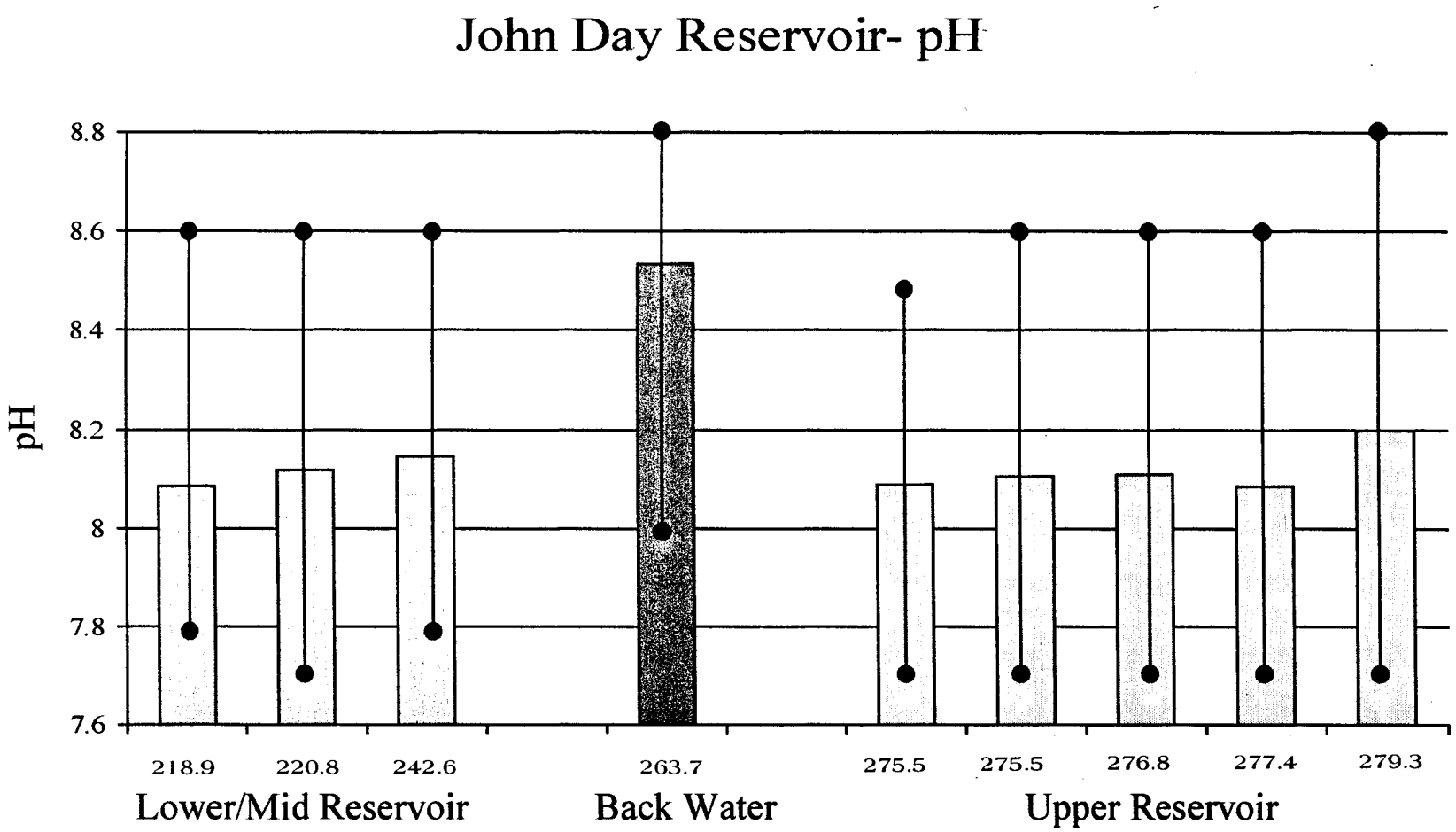

Figure 20. Annual mean and range of pH measured from April 1994 through September 1995 in John Day Reservoir, Columbia River. Data from station 263.7 are from March 1995 to September 1995.

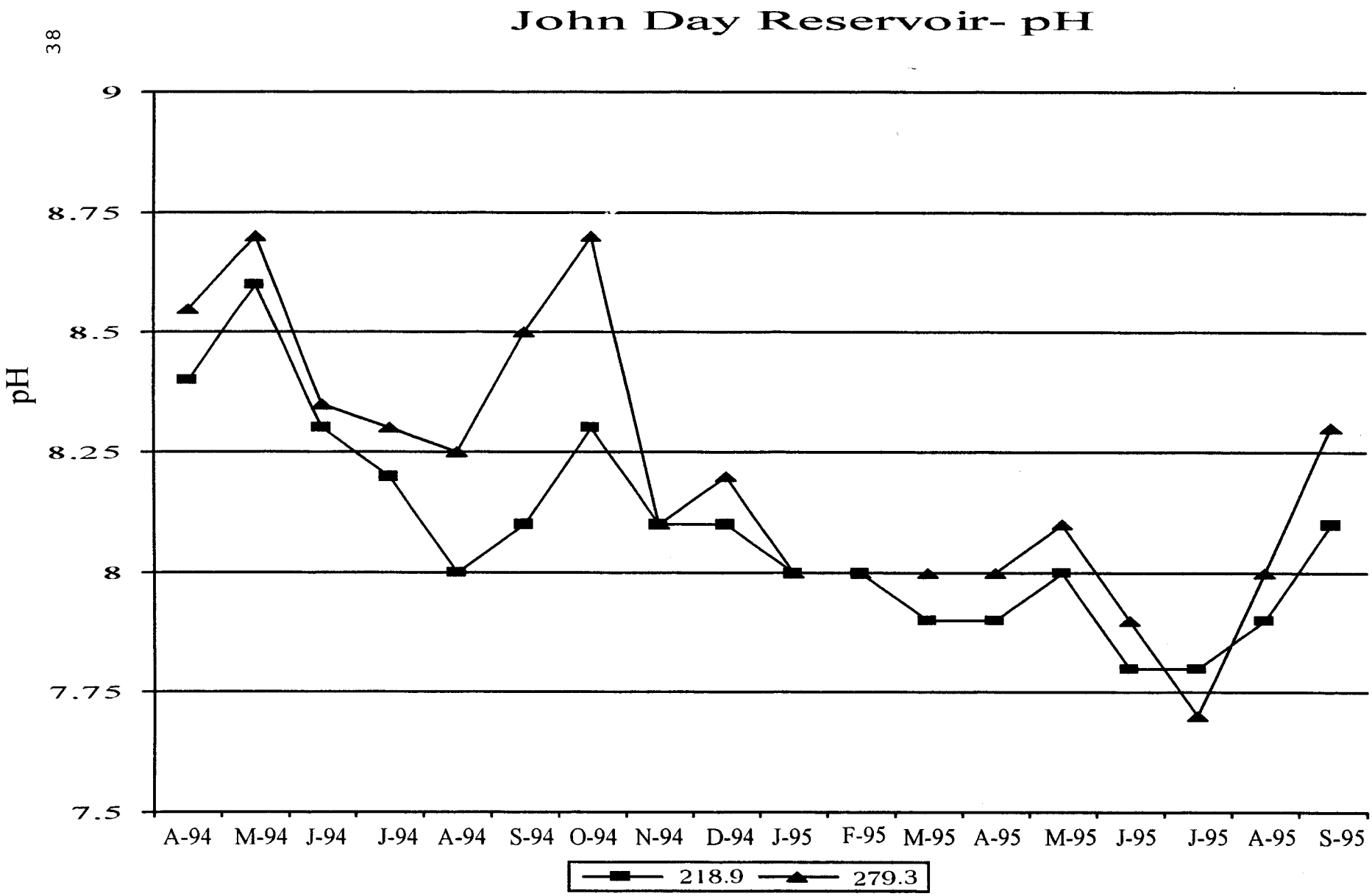

Figure 21. Monthly mean pH measured from April 1994 through September 1995 at river miles 218.9 and 279.3 in John Day Reservoir, Columbia River.

Zooplankton

Rotifers were the most abundant zooplankter (Figures 22 and 23) in John Day Reservoir from April 1994 through September 1995. Cladoceran and copepod zooplankton showed similar trends in abundance for both 1994 and 1995 in shallow water sites. Copepods were consistently low in abundance (<10 mg/m^3) at shallow water sites except for during October 1994. Copepod and Cladoceran zooplankton were generally higher in abundance at deep water sites and exhibited peaks of abundance in early fall 1994 and 1995. Mean copepod numbers were approximately 2 to 15 times more abundant than cladocerans from April through July. Both copepod and cladoceran numbers were similar in November 1994. Highest copepod and cladoceran densities were <20/L at shallow water sites but were approximately double that at deep water sites.

Benthic Macroinvertebrates

Several studies have been conducted to examine the abundance and distribution of benthic macroinvertebrates in John Day Reservoir. Shallow water habitats generally have higher biomass of benthic macroinvertebrates based on studies in Lower Granite Reservoir (Bennett and Shrier 1986, 1987; Bennett et al. 1988, 1990, 1991, 1993a, 1993b, 1995). Current studies of benthic macroinvertebrates in John Day Reservoir also support this generalization. Although diversity of benthic macroinvertebrates in John Day Reservoir is high, the majority of the community consists of oligochaetes and chironomids (D. Ledgerwood, Nation Marine Fisheries Service, Hammond, Washington, unpublished data). An earlier examination of the benthic macroinvertebrate community in

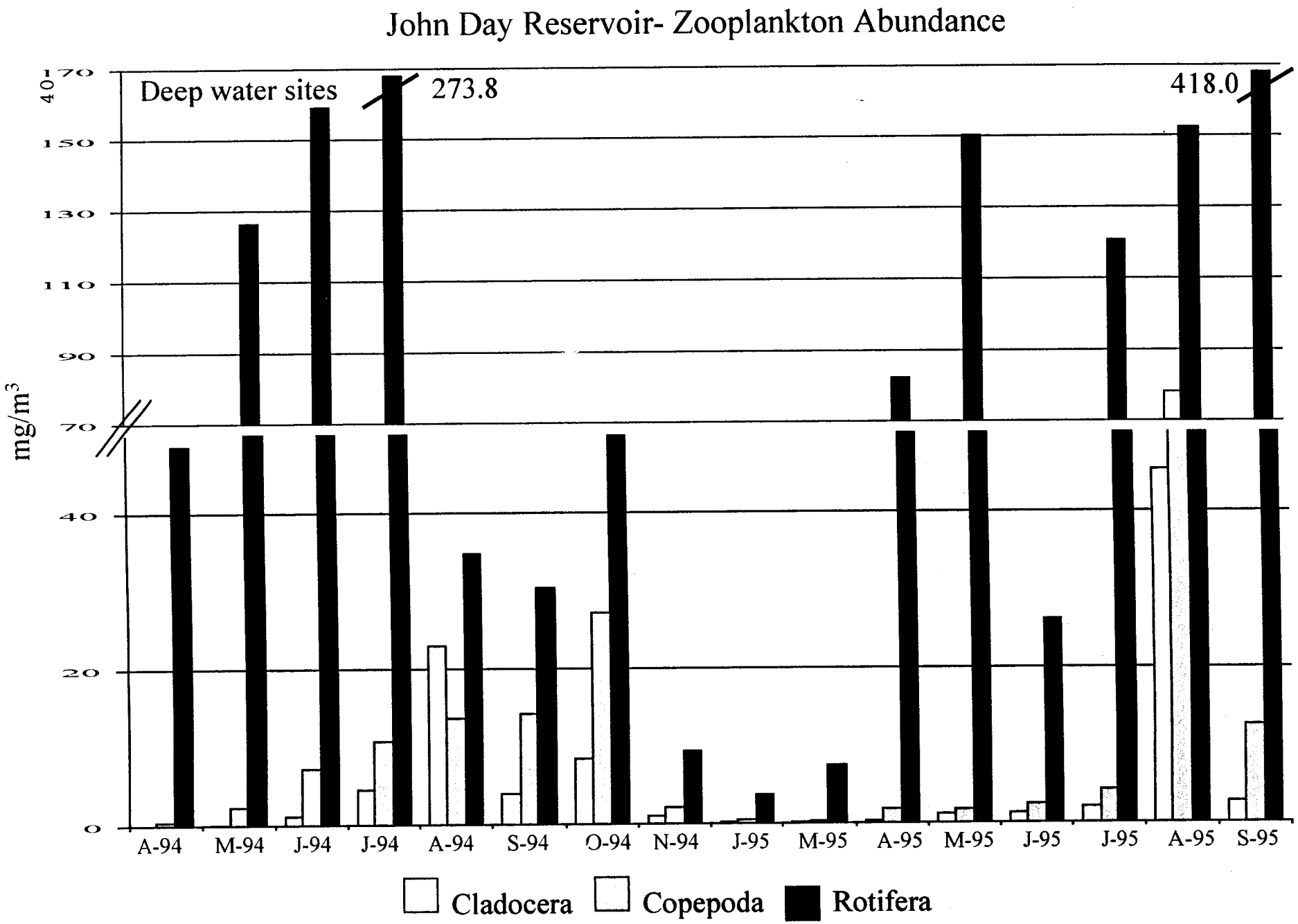

Figure 22. Mean zooplankton abundance sampled from April 1994 through September 1995 in deep water sites, John Day Reservoir, Columbia River. No February 1995 sample was taken.

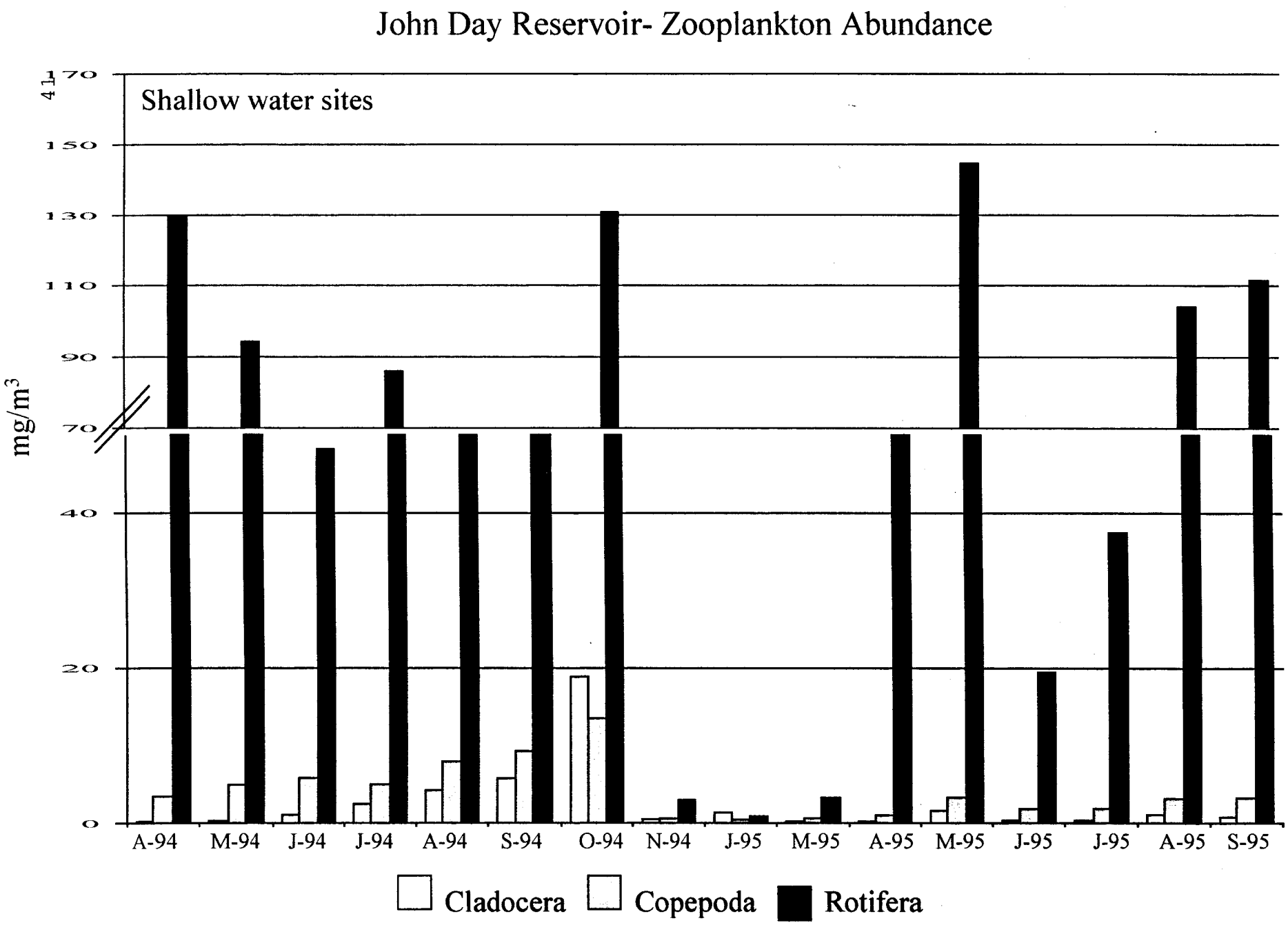

Figure 23. Mean zooplankton abundance sampled from April 1994 through September 1995 in shallow water sites, John Day Reservoir, Columbia River. No February 1995 sample was taken.

John Day Reservoir revealed higher diversity of invertebrates in Paterson Slough than the reservoir (Table 1).

Fishes

More data are available on fishes than other biota in the John Day ecosystem. Anonymous (1994), Vigg et al. (1991), and Poe et al. (1991) collectively reported 35 species/genera of fishes (Table 2) associated with John Day Reservoir. Five species are anadromous and migrate through or enter tributary streams, whereas the remaining 30 species are resident and complete their life cycle within the reservoir. Four of the five migratory species are salmonids and the fifth is the American shad *Alosa sapidissima*. Fourteen of the resident fishes and all but three resident game fishes are introduced to the Columbia River. Vigg et al. (1991) classified four fishes as piscivorous predators: northern pikeminnow (previously northern squawfish) *Ptychocheilus oregonensis*, walleye *Stizostedion vitreum*, smallmouth bass *Micropterus dolomieu*, and channel catfish *Ictalurus punctatus*. Beamesderfer and Rieman (1991) considered the native northern pikeminnow as the most abundant predator species followed by the introduced species, walleye and smallmouth bass.

Critical Life Stages of Selected Species

Walleye

Walleye are considered an important sport fish in John Day Reservoir by some fisheries managers and anglers in Washington and Oregon (Hallock and Fletcher 1991). The walleye fishery in John Day Reservoir has received national recognition as a result of

Table 1. Micro- and macroinvertebrate composition in Paterson Slough and near Paterson Slough in John Day Reservoir (Zimmerman and Rasmussen 1981).

Species	Paterson Slough	Columbia River at Paterson Slough
Nematoda		
Unidentified nematode	X[1]	NP[2]
Annelida		
Oligochaete		
Unidentified oligochaete	X	X
Branchiura sowerbyi	X	NP
Polychaete		
Neanthes limicola	X	X
Crustacea		
Cladocera		
Bosmina longirostra	X	NP
Daphnia spp.	X	NP
Copepoda		
Cyclops spp.	X	NP
Insecta		
Odonata		
Unidentified odonata	X	NP
Diptera		
Chironomidae	X	X
Miscellaneous		
Mollusc		
Corbicula manilensis	X	NP
Gastropoda		
Lymnea spp.	X	NP
Amnicolidae	X	NP

X[1] indicates present at the site during sampling
NP[2] indicates absent at the site during sampling

Table 2. Fishes collected in the John Day Reservoir (Anonymous 1994; Vigg et al. 1991).

Family	Common Name	Scientific Name	Origin
Acipenseridae	White sturgeon	Acipenser transmontanus	Native
Clupeidae	American shad	Alosa sapidissima	Introduced
Cyprinidae	chiselmouth	Arcocheilus alutaceus	Native
	carp	Cyprinus carpio	Introduced
	peamouth	Mylocheilus alutaceus	Native
	northern pikeminnow	Ptychocheilus oregonensis	Native
	redside shiner	Richardsonius balteatus	Native
	speckled dace	Rhinichthys osculus	Native
	longnose dace	Rhinichthys cataractae	Native
	tench	Tinca tinca	Native
	goldfish	Carassius auratus	Native
Catostomidae	bridgelip sucker	Catostomus columbianus	Native
	largescale sucker	Catostomus macrocheilus	Native
Ictaluridae	black bullhead	Ameirus melas	Introduced
	brown bullhead	Ameirus nebulosus	Introduced
	channel catfish	Ictalurus punctatus	Introduced
	tadpole madtom	Noturus gyrinus	Introduced
Salmonidae	sockeye salmon	Oncorhynchus nerka	Native
	rainbow trout	Oncorhynchus mykiss	Native
	chinook salmon	Oncorhynchus tshawytscha	Native
	coho salmon	Oncorhynchus kisutch	Native
	mountain whitefish	Prosopium williamsoni	Native
Percopsidae	sand roller	Percopsis transmontana	Native
Gasterosteidae	threespine stickleback	Gasterosteus aculeatus	Native
Cottidae	Prickly sculpin	Cottus asper.	Native
	Paiute sculpin	Cottus beldingi	Native
Centrarchidae	pumpkinseed	Lepomis gibbosus	Introduced
	bluegill	Lepomis macrochirus	Introduced
	smallmouth bass	Micropterus dolomieu	Introduced
	largemouth bass	Micropterus salmoides	
	Introduced		
	black crappie	Pomoxis nigromaculatus	Introduced
	white crappie	Pomoxis annularis	Introduced
Percidae	yellow perch	Perca flavescens	Introduced
	walleye	Stizostedion vitreum	Introduced

large sized fish in the population.

Population Characteristics:

Abundance.- Beamesderfer and Nigro (1989) reported the walleye population in John Day Reservoir consisted of about 15,000 fish >250 mm fork length (FL). As most of the walleye were found in the upper half of the reservoir, because of the extensive littoral habitat, Beamesderfer and Rieman (1991) indicated a general decrease in abundance of walleye from upstream to downstream. In 1994, Tinus and Beamesderfer (1994) reported that the average size of walleye in John Day Reservoir increased from 53.8 cm in 1983 to 64.6 cm in 1986. Growth rates for walleye were considered the highest in North America, although those high growth rates were probably attained because of the low density of the population.

Spawning and incubation.- Little information is currently known about the early life history stages of walleye in John Day Reservoir. Limited information suggests that walleye year-class strength is inversely related to flow conditions; higher flows result in lower survival. The mechanism and timing of setting walleye year-class strength are not known, but it is probably related to reproduction and early life history. Walleye commonly migrate to the tailwaters of dams or waterfalls for spawning (Scott and Crossman 1973). Kienst and Smith (1976) reported that optimum temperatures for walleye fertilization were 6 to 12° C, and optimum incubation temperatures were 9 to 15° C. Johnson (1961) found that walleye egg survival during incubation was highest on gravel-rubble (35.7%) and was lowest on muck 0.6%. Highest survival from egg to larval stage was associated with the shortest incubation period that occurred when

minimum water temperatures were higher than other years. Incubation periods were temperature dependent ranging from 20 to 24 days to 12 to 14 days. Houde (1969) showed that unusually cold water during the incubation period may be an important factor in establishing year-class strength. Kienst and Smith (1976) reported that sharp rises or drops in temperature had no effect on walleye fry and juvenile survival, although unusually cold weather after emergence may inhibit fry survival. Strong year-classes may be dependent on gradual warming of the water during egg deposition, incubation, and after emergence of fry. Water temperatures in John Day Reservoir probably warm more slowly than water temperatures in the native range of walleye and may be related to their variable year-class strength and survival. No known information exists on timing of spawning and emergence in John Day Reservoir so additional speculation is not justified.

Food and feeding.- Survival of walleye from emergence to the juvenile stage may be the period in the life cycle when year-class strength is established. Several authors have reported on the importance of food and its availability for walleye fry. Houde (1969) found that walleyes 7.0 to 9.5 mm fed before their yolk sac was completely absorbed, whereas Mathias and Li (1982) reported that initial walleye feeding first began after yolk sac absorption at 8.5 mm in length, but not as prolarvae. Houde (1969) found copepods comprised 85% to 91% of the food items of walleye 7.0 to 12.0 mm. *Epischura* was the most common copepod being positively selected by walleye, probably because of their large size. Larger walleye, 13.0 to 24.0 mm, contained copepods, cladocera, and fish. Mathias and Li (1982) reported that the first feeding by walleye was a strike response using sight. They reported that walleye did not select daphnids over copepods, although walleyes in ponds fed on *Daphnia pulex* and *Cyclops bicuspidatus*, which were

80% of the available zooplankton. Size of zooplankton selected by walleye increased slightly with increased body size. Priegel (1969) found that walleyes 10 to 50 mm feed principally on *Diaptomus* spp., *Leptodora* spp., and chironomid larvae. Mathias and Li (1982) showed that walleye 54 mm selected *Chaoborus* spp. larvae and mayfly nymphs more strongly than zooplankton, and as juveniles grew larger, gammarids and mayflies were selected. Piscivory started at 34 mm. Others reported lengths of 50 to 100 mm as the size of piscivory, although this may be more related to availability of suitable sized prey. Some fish fry consumed to a limited extent were yellow perch *Perca flavescens* and suckers *Catostomus* spp. Cannibalism was a factor for walleye >30 mm (Mathias and Li 1982). Priegel (1969) indicated that walleyes <75 mm should be considered plankton feeders and this only differs when forage fish are extremely abundant. Forage fish are eaten in proportion to their abundance. Although these studies emphasize the importance of food, Houde's (1969) data suggested variation in initial food levels was not a critical factor affecting survival of walleye. Food abundance in his study area however, was probably several orders of magnitude higher than in John Day Reservoir.

Environmental conditions.- Environmental conditions are the major factors influencing the walleye population abundance in John Day Reservoir (Anonymous 1991). Reproduction and the early life stage were reported as most adversely affected by environmental conditions. Specifically, walleye reproduction was related to high flows, high velocity waters, and fluctuations in water level.

High flows through John Day Reservoir in the spring from upstream runoff create highly unfavorable conditions for walleye embryo incubation and early feeding. Water temperatures remain low during most years, high flows create low water retention, which

is undesirable for increased plankton abundance. Plankton is highly important for survival of walleye fry; this is probably the reason several authors have strongly supported water management to enhance walleye recruitment. Hallock and Fletcher (1991) indicated that although natural production of walleye occurs in John Day Reservoir, it is highly variable. Strong negative correlations have been found between year-class strength and mean annual flows. Reduced water velocities would increase survival by lowering displacement of fry from littoral rearing areas and increase retention time that would enhance food abundance for fry. Beamesderfer and Nigro (1989) strongly believed that recruitment limitations of walleye were responsible for the low density in John Day Reservoir.

Northern Pikeminnow

Although the northern pikeminnow has been the subject of an extensive bounty system, little ecological information is really known about this species.

Population Characteristics:

Abundance.- An average estimated 85,316 northern pikeminnow (or 4.4 fish/ha) >250 mm were found in John Day Reservoir from 1984 to 1986 (Beamesderfer and Rieman 1991). Abundance of northern pikeminnow (>250 mm) varied during those years with 12 to 18 times more northern pikeminnow in the McNary boat restricted zone than in other areas throughout the reservoir where they were generally similar in abundance.

Spawning and incubation.- Abundance of northern pikeminnow in John Day Reservoir is probably not limited by successful reproduction. The literature suggests that

spawning habitat of northern pikeminnow is not narrow, and suitable habitats are varied. Patten and Rodman (1969) reported that spawning of northern pikeminnow occurred over steep (gradients 20-45°) talus slopes of shale, extending underwater to about 15 m in depth; a habitat reported by Patten and Rodman to be highly abundant in John Day Reservoir. Eggs from northern pikeminnow are demersal and adhesive, suggesting that high flows would probably not dislodge incubating embryos and adversely affect spawning success.

Timing of spawning was reported to be mid-May to late June in lower Snake River reservoirs (Bennett et al. 1983), a little earlier than that reported by Horjt et al. (1981) for the John Day pool (June to August). Northern pikeminnow reportedly spawn from May to early July (Carl et al. 1959), both in lakes and tributary streams (Jeppson and Platts 1959; Patten and Rodman 1969). In Cascade Reservoir, central Idaho, Casey (1962) reported that northern pikeminnow spawn during June with peak spawning activity in the latter part of June. Water temperatures at the time of spawning in Cascade Reservoir ranged from 14.0 to 20.4° C, similar to those reported by Casey (1962) and Stewart (1966), 14.5 to 16.7° C and 18.0° C, respectively. These temperatures occur in John Day Reservoir from May to August (Figure 2).

A similar paucity of information on spawning of northern pikeminnow is true for the Snake River reservoirs. The importance of the early rearing period has been emphasized by studies done by Bennett et al. (1994) and Cichosz (1996). Cichosz (1996) examined factors that limit the abundance of northern pikeminnow in Lower Granite Reservoir. He found that abundance of northern pikeminnow is probably limited in the egg through larval stage, although juvenile mortality is also important. Density

independent factors were most important in controlling egg through juvenile survival. Timing of temperature conditions was most important in predicting survival of northern pikeminnow. Survival was also positively related to growth.

Ward et al. (1995) correlated biological characteristics of northern pikeminnow populations and found only a significant correlation of density with relative fecundity, implying that northern pikeminnow populations were not limited by density.

Food and feeding.- A number of studies have reported on the food habits of northern pikeminnow. Ricker (1937) first reported the significance of predation by northern pikeminnow on salmonid fishes. Others such as Falter (1969) reported no incidence of salmonid consumption in the St. Joe River, Idaho. Bennett et al. (1983) reported that salmonids comprised a significant proportion of the diet of northern pikeminnows in Little Goose Reservoir, although a wide diversity of food items were found. Crayfish were a highly common food item. Chandler (1993) was the first to quantify actual predation on downstream migrating salmonids in the lower Snake River reservoirs. Chandler found that salmonids were the most abundant food item (by weight) consumed by northern pikeminnow during spring 1987-1991, and crayfish were second in importance. In John Day Reservoir, Poe et al. (1991) reported that juvenile salmonids were the principal food item during May and August whereas during other months, prickly sculpin *Cottus asper* were the most important dietary item. Crayfish and insects constituted a much lower proportion of the diet of northern pikeminnow in John Day Reservoir.

Environmental conditions.- A number of factors could affect northern pikeminnow abundance in John Day Reservoir. Larval abundance of northern

pikeminnow and factors affecting it are currently being examined in John Day and other Columbia River system reservoirs (T. Poe, USGS, personal communication). Barfoot et al. (1995) reported that limited information is known about the early life history of northern pikeminnow in the Columbia River. Larval northern pikeminnow were sampled in highest abundance at sandy sites in the upper John Day pool. Barfoot et al. (1995) found a strong linear relationship (r=0.84) between larval northern pikeminnow abundance (No./100 m^2) and water velocity from ichthyoplankton tow net samples, although this relationship was not found in nearshore sled samples. However, Bennett et al. (1991) found highest abundance of larval northern pikeminnow in backwaters of Lower Granite Reservoir. Also, Bennett et al. (1995) found that larval cyprinid abundance varied by several orders of magnitude in Lower Granite Reservoir, Snake River, between usual operating conditions and when the reservoir was maintained at minimum operating pool. They hypothesized that fluctuating water levels probably deleteriously affected survival of northern pikeminnow fry. However, the strength of year-classes from those years of extremely high larval abundance does not appear to be more than other years, suggesting that factors other than larval fish abundance may limit northern pikeminnow abundance, at least in Lower Granite Reservoir .

Dresser (1996) examined the influence of habitat factors on fish assemblages in Lower Granite Reservoir. Through the use of multivariate analysis, he reported that northern pikeminnow selected shallow, vegetated areas with substrate <2.0 mm. These findings were considerably different from those of Dupont (1994) who found that northern pikeminnow in the Pend Oreille River, Idaho, selected rocky shorelines with deeper depths and higher water velocities. Dresser (1996) believed differences in selected

habitats could be attributed to interspecific associations with other species. Similar habitats occupied by the northern pikeminnow in the Pend Oreille River are occupied by smallmouth bass in Lower Granite Reservoir. Smallmouth bass were not present in the Pend Oreille River. Some evidence supports the hypothesis that predation on northern pikeminnow from smallmouth bass may account for differences in habitat use. Werner et al. (1997) reported that predation on small size classes may result in habitat segregation. Pollard (Idaho Department of Fish and Game, retired, Portland, Oregon) observed that the abundance of northern pikeminnow decreased following the introduction of smallmouth bass into Anderson Ranch Reservoir, Idaho. He further suggested that similar habitats inhabited by the northern pikeminnow in Brownlee Reservoir, Idaho, were absent of them following the introduction of smallmouth bass. Since the majority of northern pikeminnows collected by Dresser (1996) were 120-250 mm and the smallmouth bass he collected ranged in length from 100 to 520 mm, his explanation seems plausible.

White Sturgeon

Population Characteristics:-

Abundance.- No known population estimates are available on white sturgeon *Acipenser transmontanus* in John Day Reservoir. Lepla (1994) conducted the most comprehensive study on white sturgeon in the Columbia River system in Lower Granite Reservoir. He sampled nearly 1,000 white sturgeon and examined habitat use. Overall absolute abundance of white sturgeon >40 cm TL was estimated at 1,372 (95% confidence intervals 946-2,166). He found that 94% of the white sturgeon in Lower Granite were <125 cm total length (TL) with the majority in the 0-8 age group.

John Day Reservoir provides suitable habitat for juvenile and adult white sturgeon (Beamesderfer and Nigro 1992a, 1992b), although recruitment of white sturgeon in the reservoir is low (Parsley et al. 1993). Dams on the Columbia River have restricted movement of white sturgeon and have functionally isolated reservoir populations. Production of white sturgeon in the Columbia River reservoirs is less than in the unimpounded section downstream of Bonneville Dam, suggesting that impounded waters may provide less suitable habitat.

Spawning.- Spawning of white sturgeon occurs in John Day Reservoir over coarser substrate from May through July, primarily at 14° C based on collection of recently spawned eggs. Some juvenile white sturgeon rear in embayments adjacent to the original river channel (Anonymous 1978). Parsley and Beckman (1994) quantified spawning habitat in three of the lower Columbia River reservoirs using geographic information systems (GIS). They showed that spawning habitat was available downstream of each of the dams, although the quantity of habitat was affected by flows. Rearing habitat for age 0 and juvenile white sturgeon was also quantified, and the impounded river contained more than the unimpounded lower Columbia River reach. Samples of numerous juvenile white sturgeon (<16 cm) from Lower Granite Reservoir suggest that juvenile rearing habitat is probably highly abundant in the upper John Day Reservoir (Bennett et al. 1993a, 1993b).

Rearing habitat for white sturgeon seems to be linked to water velocity. Apperson (1990) suggested that white sturgeon in the Kootenai River, Idaho, were found at water velocities between 0.10 and 1.83 ft/s. Velocities in this range would be found throughout John Day Reservoir if either of the drawdown alternatives were implemented.

Food and feeding.- Crayfish are reportedly an important food item of white sturgeon in the Snake River (Coon et al. 1977; Cochanuer 1983). Relative crayfish abundance has been quantified in Lower Granite Reservoir and crayfish distribution correlates well with that of white sturgeon (Bennett et al. 1993a, 1993b; Lepla 1994). Bennett et al. (1993a, 1993b) could not ascertain whether crayfish abundance was responsible for the upstream abundance of white sturgeon or both species had similar habitat preferences.

Environmental conditions.- Lepla (1994) attempted to develop a stepwise discriminant model to explain white sturgeon distribution but could account for only 26% of the variation in locations using habitat data. However, he found 56% of all fish sampled were from a 3.4 river mile (5.5 Rkm reach; Port of Wilma to Red Wolf Crossing), having highest water velocities in Lower Granite Reservoir. Catches in the mid to lower reservoir were consistently low. Coon (1977) suggested the importance of moving water to white sturgeon based on tracking fish with sonic tags. His implanted fish moved to the upstream portion of Lower Granite Reservoir during impoundment and resided in the same area as the majority of where Lepla (1994) sampled his fish.

Although subadult and adult white sturgeon find suitable habitat in John Day Reservoir, their lack of recruitment and overharvest of adults limits abundance of the population. Reservoir management also could enhance habitat for subadult and adult sturgeon. Beamesderfer and Nigro (1992a) reported that sturgeon production could be enhanced during low flow years in Columbia River reservoirs by increasing river discharge during the peak of the spawning season.

Smallmouth Bass

Population Characteristics:

Abundance.- Annual population estimates of smallmouth bass in John Day Reservoir ranged from 31,948 to 37,959, an average density of about 1.8 fish/ha (Beamesderfer and Ward 1994). Highest densities of smallmouth bass within the reservoir were in the forebay, sloughs, and backwaters, which is similar to reports by Beamesderfer and Rieman (1991). The proportional stock densities (PSD) of 27% to 60% suggest varied abundance of larger fish. Mesa et al. (1990) showed that PSD varied temporally and spatially in John Day Reservoir. Proportional stock density was lowest in the forebay, ranging from 8% to 46% but was significantly higher at Irrigon from 1983 to 1986 ranging from 31% to 100%. Differences in PSD were attributed to food abundance and water temperature.

Year-class strength of smallmouth bass in John Day Reservoir is highly variable as Beamesderfer and Ward (1994) reported a five-fold difference in recruitment between 1976 and 1982. Connolly and Rieman (1988) reported that first-year growth, not the environmental variables examined, influenced year-class strength. Mesa et al. (1990) believed that some environmental influence affected year-class strength in John Day Reservoir.

Spawning and incubation.- The spawning season of smallmouth bass in the northwest is generally later than that reported throughout its range. Bratovich (1985) analyzed the reproductive cycle of smallmouth bass from examination of gonads from smallmouth bass sampled in Lower Granite Reservoir. Largest ovaries were measured in April, although the time of spawning from examination of ovaries was estimated to be

May, June, and July. Other observations suggest spawning largely occurs in June and July, based on attainment of suitable water temperatures of about 15.9° C (Coble 1975). Water temperatures of 15° C occurred in John Day Reservoir in May and June (Figure 2). Pflieger (1975) reported smallmouth bass spawning as early as the first of April in Missouri, compared to the presence of suitable temperatures in May and June in John Day Reservoir. Henderson and Foster (1957) observed smallmouth spawning in the Columbia River the latter part of July. Bennett et al. (1983) suggested a spawning period of longer than 60 days, similar to that reported for smallmouth bass in Missouri (Pflieger 1975).

Bennett et al. (1983) observed spawning to occur over a range of temperatures from 14 to 19.6° C, which was within the range reported in the literature (12.8 to 26.7° C, Henderson and Foster 1957; Reynolds 1965) for smallmouth bass. These temperatures were generally close to the commonly reported temperatures of 15 to 18.3° C (Turner and McCrimmon 1970; Coble 1975; Coutant 1975; Pflieger 1975).

Habitat used for spawning by smallmouth bass is largely gravel substrate along the shoreline, a habitat type highly abundant in John Day Reservoir. Substrate used by smallmouth bass for spawning in Little Goose Reservoir was similar to that reported in the literature (Bennett et al. 1983). In Little Goose Reservoir, all observed smallmouth bass spawning activity was on low gradient shorelines of sand and/or gravel, 85% of the spawning nests located were constructed on substrate 6 to 50 mm in diameter. Spawning areas in Little Goose Reservoir were frequently found in gulch and embayment areas in the lower reservoir. The areas were generally protected from direct wind and wave action and experienced little to no perceptible current. In upper Little Goose Reservoir,

smallmouth bass nests were more commonly observed in shoal areas, usually exposed to wind, wave action, and/or higher water velocities. Differences in habitat used by bass were attributed to few gulch and embayment habitats in the upper reservoir (Bennett et al. 1983). In 1985, Bennett and Shrier (1986) reported that smallmouth bass spawning nests were located in Lower Granite Reservoir from the confluence of the Snake and Clearwater rivers downstream to nearly Lower Granite Dam. Highest nest abundance was in the lower part of the reservoir, an area of lower water velocities.

Spawning habitat is probably not limiting smallmouth bass abundance in John Day Reservoir based on preferred habitat described above and by Scott and Crossman (1973). However, reservoir conditions probably adversely affect spawning and egg to fingerling survival by affecting timing of spawning. In addition to spawning late in the spring, duration of incubation is temperature dependent and may exceed 10 days at 12.2° C. Actual water temperatures in John Day Reservoir are probably not directly affecting survival of smallmouth bass, but the timing of spawning is most critical. Lower flow years have higher water temperatures, and smallmouth bass spawning commences earlier. Earlier spawning is important as it advances earlier life stages and promotes increased growth, an important characteristic since body length is a highly critical factor affecting young-of-year survival. De Angelis et al. (1993) showed that smallmouth bass must exceed 50 mm entering their first winter or survival is low. Based on available water temperature data, smallmouth bass spawning probably occurs in June and July in John Day Reservoir.

Food and feeding.- Food abundance for fry may be another factor that affects year-class strength for smallmouth bass in John Day Reservoir. As with walleye, food

abundance is critical for survival of smallmouth bass, and high flows through the reservoir are an impediment for plankton production. This may be why Beamesderfer and Ward (1994) reported that smallmouth bass production is low in John Day Reservoir and limited by short water retention, lack of nutrients, an unstable littoral zone, and fluctuating water levels. Mesa et al. (1990) found that fish accounted for 96% (weight) of the dietary items of smallmouth bass at Irrigon compared to 62% in the forebay. Crayfish were the second most important item in the diets of smallmouth bass in the forebay of John Day Dam. Differences in diet between the forebay and Irrigon may have accounted for differences in growth of smallmouth bass between the two areas. Food items found in stomachs of smallmouth bass in John Day Reservoir are similar to those in the lower Snake River reservoirs.

Bennett et al. (1983) found that smallmouth bass (n=484) consumed crayfish, fish, and terrestrial and aquatic insects in decreasing order of importance in Little Goose Reservoir during 1979 and 1980. Crayfish accounted for 72% by volume of the food items eaten and appeared in 64% of all bass stomachs. Fishes consumed accounted for 25.4% by volume and were found in 32% of the smallmouth bass stomachs that contained food. Fishes eaten were sculpin *Cottus* spp., white crappie *Pomoxis annularis*, redside shiner *Richardsonius balteatus*, northern pikeminnow, catfish *Ictalurus* spp., bluegill *Lepomis macrochirus*, yellow perch, chinook salmon *O. tshawytscha*, bridgelip sucker *C. columbianus*, and pumpkinseed *L. gibbosus*. In Lower Granite Reservoir, crayfish were consistently the dominant food item in Lower Granite Reservoir in 1995, although salmonids and other fishes accounted for nearly 50% of the diet in the spring of 1995, (Anglea 1997).

Anglea (1997) examined food items from more than 4,000 smallmouth bass in Lower Granite Reservoir and found that fish were the most important food item, by weight, from April to June 1994 and 1995, whereas crustaceans and insects increased in abundance after June. As others (Coble 1975) have reported, larger smallmouth bass consumed a higher proportion of fish. Crayfish were the most abundant food item by weight for smallmouth bass 175-249 mm while finfish and crayfish were equally important for bass 250-389 mm. Fish were the dominant food item of smallmouths >389 mm in length. In 1996 and 1997, Bennett and Naughton (1998) examined more than 8,500 smallmouth bass stomachs from the tailwater, forebay, and Snake and Clearwater River arms of Lower Granite Reservoir. They found that nonsalmonid fishes were the most abundant prey item by weight in the tailrace (46.9%), tailrace boat restricted zone (BRZ; 71.6%), forebay BRZ (51.5%), and Clearwater River arm in 1996. In 1997, crustaceans were clearly the dominant food item by weight in the tailrace (73.4%), tailrace BRZ (60.8%), forebay (58.8%), and Snake River arm (50.3%). Temporal differences in food items were not high within study sites. From these findings, it is obvious that smallmouth bass in the lower Snake River reservoirs and John Day Reservoir consume a large number of crayfish, similar to that reported in the literature for other river and lake systems.

Growth and condition.- Mesa et al. (1990) showed that relative weight (Wr) of smallmouth bass varied temporally and spatially in John Day Reservoir from 1983 to 1986. Mean relative weights were generally <100% in the forebay for bass <300 mm, while those for bass >400 mm were >100%. Mean relative weights at Irrigon for bass 150 to 225 mm were <100%, while those for bass >300 mm were generally >100%.

Incremental growth was higher at Irrigon than in the forebay and can be attributed to a 2° C difference in water temperature (Mesa et al. 1990).

Environmental conditions.- Probably the main environmental constraint to the smallmouth bass population in John Day Reservoir is water temperature. Water temperature affects the timing of spawning of smallmouth bass, the incubation period, the abundance of food, in part, and the rate of growth. As indicated, earlier spawning is important as it advances earlier life stages and promotes increased growth, an important characteristic since body length is a highly critical factor affecting young-of-year survival. Highest water temperatures recorded in John Day Reservoir during 1994 and 1995 were about 22° C, 7° C below the 29° C reported preferred temperature for smallmouth bass (Coble 1975). Another environmental constraint is the availability of planktonic organisms during the early rearing period. Abundance of cladoceran and copepod zooplankters were fairly low during the period from May to July in John Day Reservoir (Figures 22 and 23).

Anadromous Fishes

Three key elements involving anadromous fishes are associated with water level management in John Day Reservoir: predator abundance and efficiency, use of backwaters by juvenile anadromous fish, and migration through the reservoir.

Use of backwaters for rearing.- Prior to construction of John Day Dam, salmon and white sturgeon were originally abundant in the riverine habitat. Dam construction and filling of the reservoir created a diversity of habitats that did not exist in the original river channel. Backwater areas are one type of the habitats that were created. Operating John Day reservoir at either water elevation (165 or 220 ft elevation MSL) would cause

substantial habitat alterations. Substantial loss of backwater habitat would minimize the rearing potential for anadromous and resident fishes in John Day Reservoir, although suitable "in-river" habitat may increase. Fish and habitat surveys conducted in 1978 and 1979 in Paterson Slough demonstrated the presence of downstream migrating juvenile anadromous salmonids in backwaters from April through July (Zimmerman and Rasmussen 1981). Salmonid growth during the period of sampling suggested the importance of food and feeding opportunities in backwaters for juvenile salmonids. Anonymous (1980) sampled backwaters throughout the lower Columbia River system. They reported that peak salmonid abundance occurred from spring and summer and decreased in September. Of seven backwaters sampled in John Day Reservoir, juvenile anadromous salmonids were collected in five. Phase II studies of one backwater (Sundale, RM 236.5, Rkm 381) in John Day Reservoir, sampled from October 1979 through mid-September 1980, also showed the importance of backwaters for chinook salmon (Parente and Smith 1981). Catches of juvenile chinook salmon at this site peaked in May and then declined. However, although these habitats are currently used for rearing by juvenile anadromous salmonids, the original river channel and associated shoreline "natural" habitat would provide superior rearing habitat.

Larval and juvenile resident fishes also rely heavily on backwaters for rearing. Barfoot et al. (1995) reported that northern pikeminnow larvae were collected in higher abundance in upper pools than in lower pools and their highest densities occurred in backwaters with fine sediment/sand substrates with seasonally variable amounts of aquatic vegetation.

Predator abundance and efficiency.- Completion of the predation study in John

Day Reservoir indicated that four resident fishes consumed juvenile salmonids (Vigg et al. 1991). The estimated mean annual loss of salmonid smolts to predation was 2.7 million (95% confidence interval, 1.9-3.3 million). The northern pikeminnow accounted for about 78% of the juvenile salmonid loss to predation, while walleye (13%) and smallmouth bass (9%) were less significant predators. Consumption rates were not calculated for channel catfish, although they too consumed juvenile salmonids.

Results of several studies suggest that the original estimate of salmonid smolt losses to predators may have been inflated. Petersen et al. (1994) showed that predation of dead smolts was considerably higher than on live smolts in the Bonneville Dam tailrace. Their results indicated that predator feeding behavior and prey condition were important factors when assessing the impact of predation on a prey population. Petersen and DeAngelis (1992) showed that smolt distribution in the reservoirs may affect predation estimates as current methods assume randomness, although smolt distribution is not random. Also, Petersen (1994) showed that when spatial distribution patterns of northern pikeminnow were factored into the estimates of salmonid smolt consumption, salmonid predation losses decreased to 1.4 million.

Predation was further monitored during the 4-year period of 1990 through 1993 for development of an index of predation (Ward et al. 1995). Ward et al. (1995) reported that the tailrace boat restricted zone downstream of McNary Dam was the major "hot spot" for predation and ranked second in intensity of predation to that of the Bonneville Dam tailrace.

The high predation rate determined in the study prompted establishment of the Northern Squawfish (pikeminnow) Sport Reward Program (Burley et al. 1993). Predator

abundance would probably change with sustained reservoir operations at lower elevations and therefore, salmonid losses would change.

Faler et al. (1988) reported that radio-tagged northern pikeminnow avoided high velocity (>100 cm/s) areas. They believed that smolt bypass outflows should be located in high velocity areas to reduce northern pikeminnow predation. Faler et al. (1988) found no evidence that northern pikeminnow actively moved to high velocity areas to prey on juvenile salmonid out-migrants. Mesa and Olson (1993) reported that water velocities from 100 to 130 cm/s may exclude or reduce predation by northern pikeminnow at dams because of their endurance at these velocities. Additionally, northern pikeminnow became more quickly fatigued in water temperatures of 12° C than 18° C.

Based on model predictions, Beamesderfer et al. (1990) concluded that predation observed in 1983 through 1986 was "representative" and predation was expected to have a significant effect on reservoir survival during most years. Further modeling showed that survival of juvenile salmonids could be maximized by reducing predator numbers, moving runs to earlier in the year, and maintaining size of runs. Reducing predator activity was suggested as one mechanism to decrease predation; one proposed method to reduce predator activity would be to enhance flows. Also, increasing abundance of alternative prey might reduce predation, if increased food abundance does not increase the number of predators.

Petersen and Gadomski (1994) showed that the rate of predation by northern pikeminnow was inversely related to light intensity. Based on this information, increasing the light conditions in tailrace areas, the location of highest predation, may enhance survival of salmonid smolts. Petersen and Gadomski (1994) suggested that light is critical

to structuring predator-prey interactions.

These results suggest that higher velocities in the John Day system associated with drawdown would probably reduce salmonid losses attributed to predation.

Migration through the reservoir.- The concept of reservoir drawdowns was developed in response to the perceived need to increase water velocities in the Columbia River system. Migrating fish evolved under and adapted to the natural hydrograph; construction of reservoirs has altered that hydrograph. Lowering reservoir levels is one method to minimize the need for extraneous sources of water and retain the infrastructure and also obtain near-riverine conditions within the reservoir for the desired target period. Increased migration times for juvenile salmonids have been attributed to the construction of reservoirs. Raymond (1969) reported that juvenile out-migrants, specifically in John Day Reservoir, migrate about one-third as fast through large impoundments as through free-flowing stretches of the river. Actual migration times were about 15 mi/day prior to construction of John Day Dam compared to 6 mi/day following impoundment.

One issue regarding the feasibility of reservoir drawdown to enhance smolt migrations through the system is the conditions that would exist under 220-ft elevation. As indicated, at 220-ft elevation, more riverine conditions would be created in the upstream areas, but not throughout the entire reservoir. Under these conditions, migration may only be partially enhanced with less favorable conditions in the downstream section. Anonymous (1994) reported the average water particle travel time through the reservoir varies with flow and pool elevations. Mean water particle times were estimated to range from 11.9 days at 268-ft elevation and 100 Kcfs to a minimum of 3.2 days at 257-ft elevation at 300 Kcfs. However, decreased water particle travel time within each of the

flows examined (100, 200, 300 Kcfs) ranged from 2.5 days (100 Kcfs) to 0.8 days (300 Kcfs). Thus, if out-migrants followed the water particles, the benefits of drawdown would be highest at the lower flows.

Assessment of Effects of Drawdowns to 220 and 165 Feet MSL

Methods

I quantified habitat changes associated with the proposed drawdown of John Day Reservoir to 220-ft and 165-ft MSL using GIS. I obtained depth soundings and substrate data collected by USGS personnel (Mike Parsley, Cook, Washington) at points approximately 50-ft apart along transects perpendicular to the shoreline spaced approximately at 500-ft intervals. From the soundings, "kriging" was conducted using Arc Info to interpolate the river bottom. I then used the "backwater curves" provided by personnel of the U.S. Army Corps of Engineers at flows of 500 Kcfs and 75 Kcfs (Appendix Figures 1 and 2). I selected a summer flow of 75 Kcfs to estimate available habitat for biota as habitat availability can be more critical during periods of higher temperatures and increased biotic activity. From these curves, I was able to estimate the depth of the proposed system as the difference of the surface elevation from the backwater curves and the river channel. Next, I divided the longitudinal distance between John Day Dam and McNary Dam into 10 sections. Within each of the 10 sections, a section was randomly selected and cross sections were developed by the model and plotted to show the sectional profiles at both 220-ft MSL and 165-ft MSL and resulting habitat. Depth was determined in each of five 5-ft increment classes (<5 ft, 5-10 ft, 11-15 ft, 16-20 ft, and >20 ft) and the proportion of depth in each class and substrate at each of

the 220-ft and 165-ft depths were calculated. Velocity was computed from the relationship of flow (75 Kcfs) divided by cross sectional area for each at 220-ft and 165-ft elevations.

Results

The John Day reach was divided into 10 sections, each ranging from 7 to 8 river miles (Figure 24). Substrate predictions are based on current findings and do not reflect the sorting that would occur under the higher velocities. Water velocities represent average cross-channel velocities.

Section Descriptions:

Section 1.- Section 1 extended from John Day Dam at RM 216 to 223 (Rkm 347-359; Figure 25). The river channel that would exist at section 1 if the water surface elevation at John Day Dam were at 165-ft elevation (natural river level) showed a split channel. Depth characteristics would consist mostly of water >20 ft with approximately equal areas of water in the <5 ft, 5-10 ft, 10-15-ft depth classes. River mile 222.5 (Rkm 358) was selected randomly to represent the river channel of section 1 when the surface elevation at John Day Dam is at 165 ft and 220 ft. The surface area at RM 222.5 would be substantially reduced when the surface elevation at the dam is 165 ft (natural river level) compared to that at 220 ft. At both surface elevations, the current substrate would be largely silt and clay. The modal depth classes would be similar (>20 ft) at both surface elevations; although at a level of 165 ft, about equal proportions of water depths would occur from <5 ft to 15-20 ft. Estimated water velocity at an elevation of 165 ft (natural river level) would be about 2.6 ft^{-S} compared to 0.4 ft^{-S}, about 20% of that at an elevation

Sections along the river channel

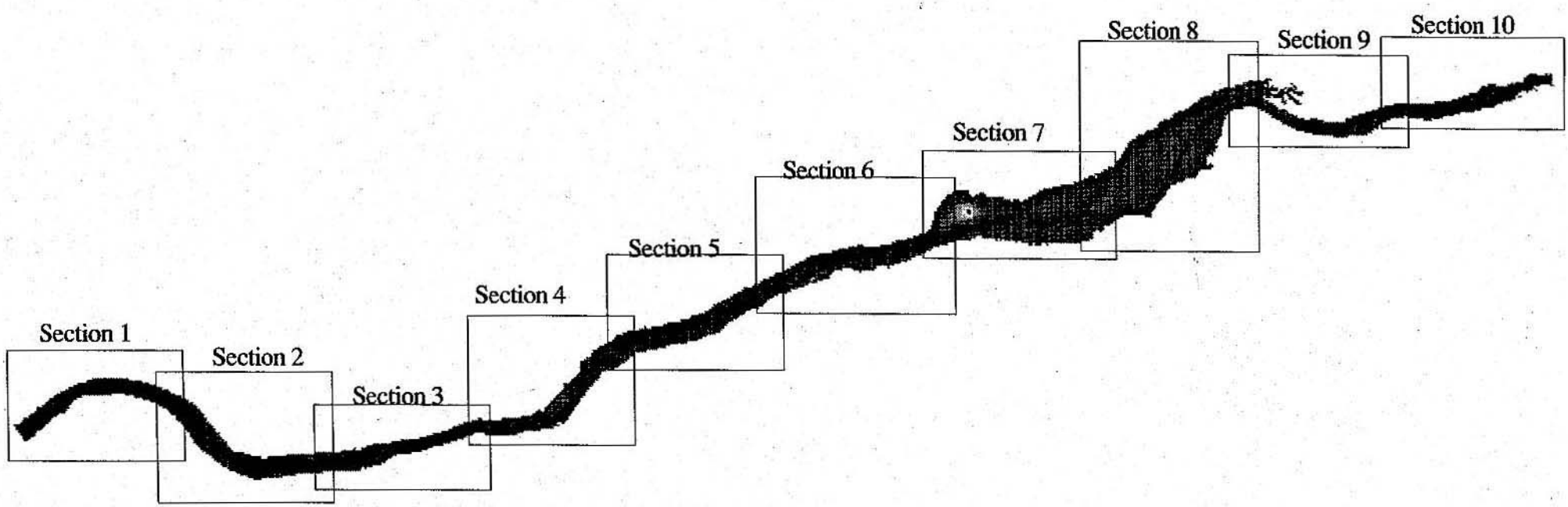

Figure 24. Map of John Day Reservoir. The reservoir was divided into 10 sections from mile 215.6 to 291.

Section 1, Rivermile 216 –223

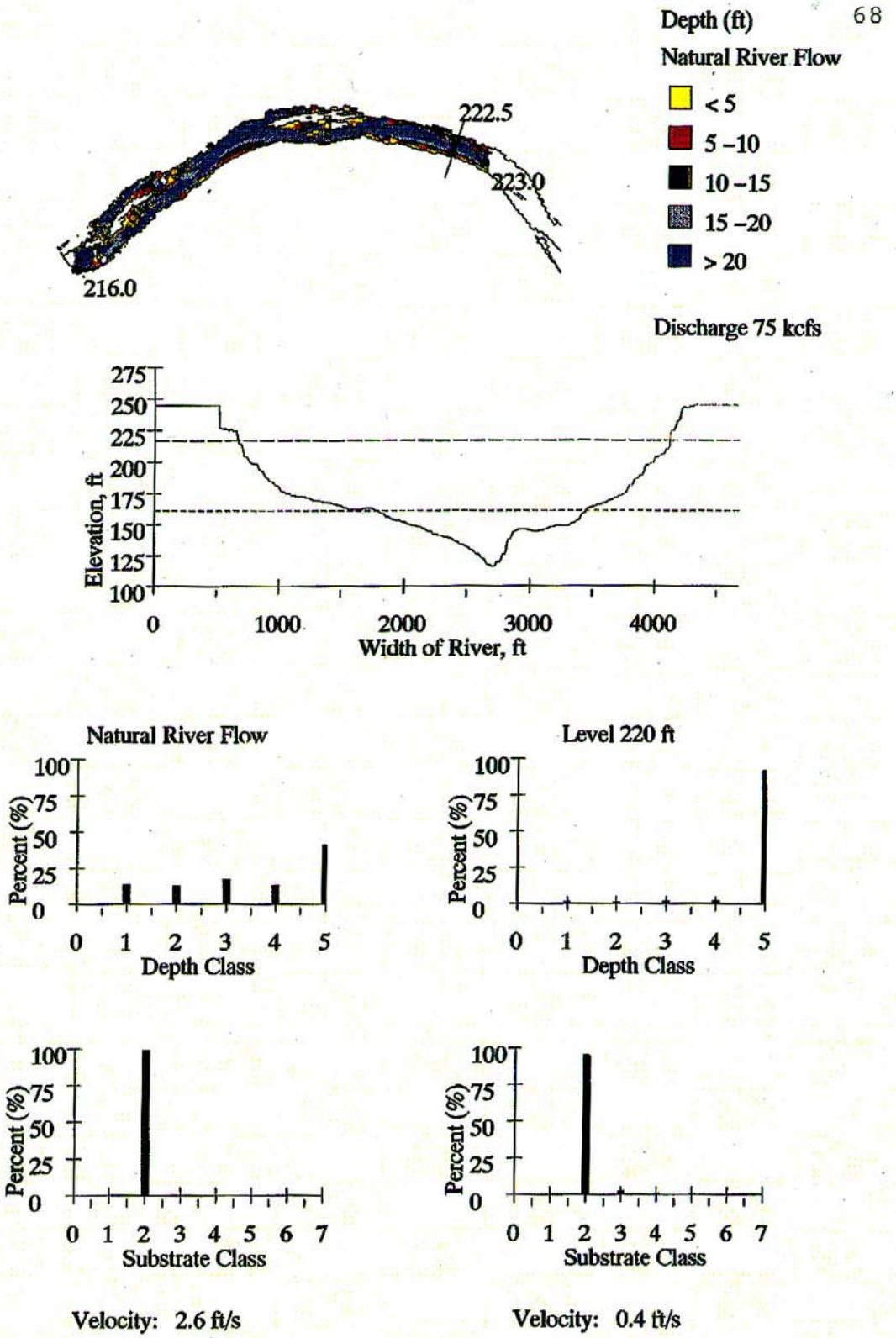

Figure 25. Physical habitat characteristics of a random transect (river mile 222.5) in Section 1 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

of 220-ft.

Section 2.- Section 2 extended from RM 223 to 231 (Rkm 359-372; Figure 26). The river channel that would exist at section 2 if the surface elevation at John Day Dam were 165 ft (natural river level) would show no side channels. Depth characteristics at this level would consist of approximately equal areas of water in the <5 ft, 5-10 ft, 10-15 ft, 15-20 ft, and >20 ft depth classes. River mile 228.4 (Rkm 367.7) was selected randomly to represent the river channel of section 2 when the surface elevation at John Day Dam is at 165 ft or 220 ft. The surface area at RM 228.4 would be substantially reduced at 165-ft elevation (natural river level) than at 220-ft elevation. At both surface elevations, the current substrate would be largely silt and clay. The modal depth classes would be >20 ft when the surface elevation is 220 ft at the dam, compared to about equal proportions at an elevation of 165 ft. The estimated water velocity at RM 228.4 when the surface elevation at the dam is 165 ft (natural river level) would be about 8.4 ft^{-S} as a result of the narrow channel width, compared to about 0.4 ft^{-S} when the a surface elevation at the dam is 220 ft.

Section 3.-Section 3 extended from RM 231 to 238 (Rkm 372-383; Figure 27). The river channel that would exist if the surface elevation at John Day Dam were 165 ft (natural river level) showed one small side channel. Depth characteristics at natural river level would consist of water with approximately equal areas in the <5 ft, 5-10 ft, 10-15 ft, 15-20 ft, and >20-ft depth classes. River mile 232.0 (Rkm 373.5) was selected randomly to represent the river channel of section 3 when the surface elevation at John Day Dam is at 165 ft and 220 ft. The surface area at this site would be substantially reduced at a surface elevation of 165 ft at the dam (natural river level), compared to that at 220-ft

Section 2, Rivermile 223 –231

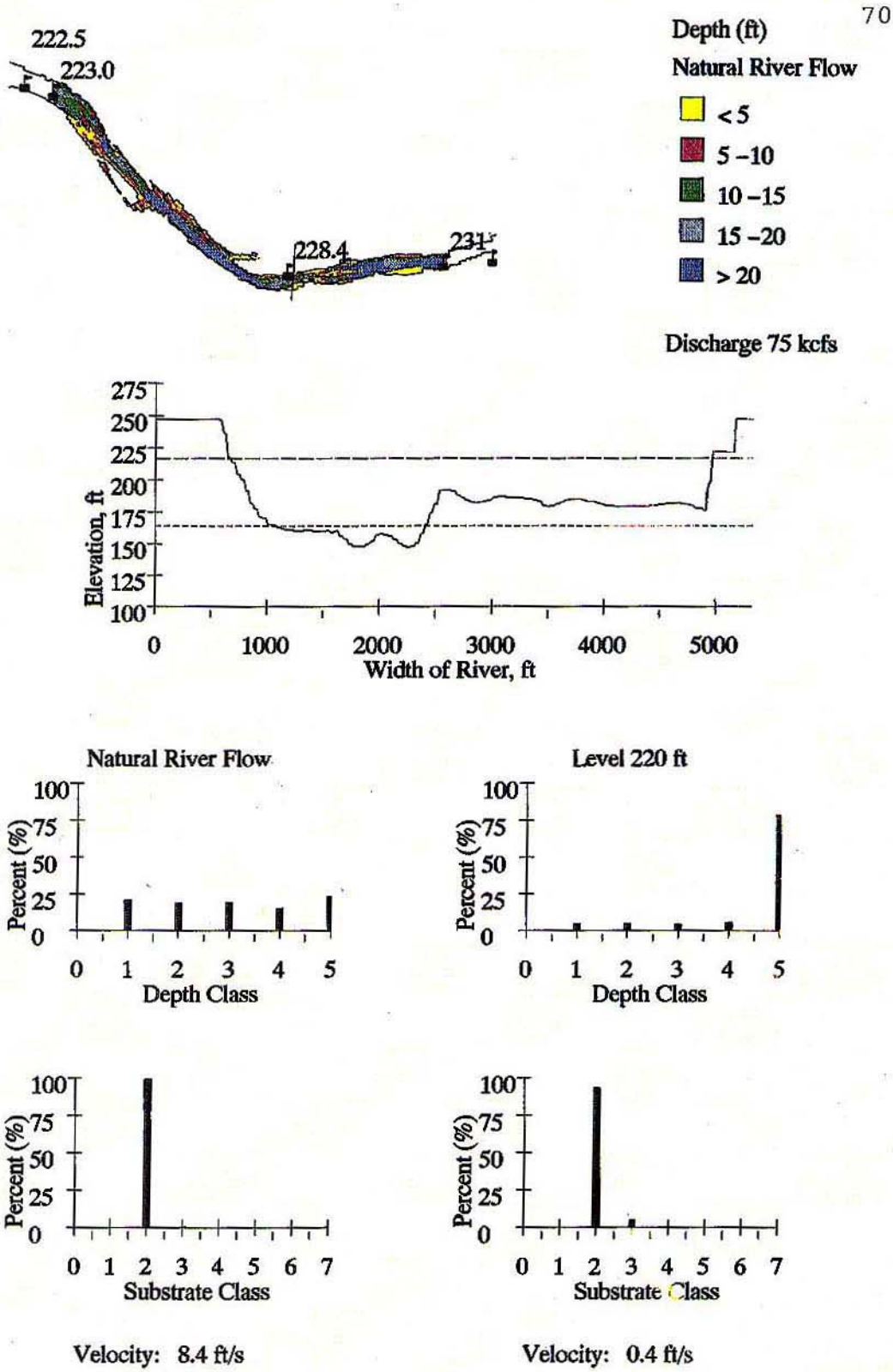

Figure 26. Physical habitat characteristics of a random transect (river mile 228.4) in Section 2 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

Section 3, Rivermile 231 –238

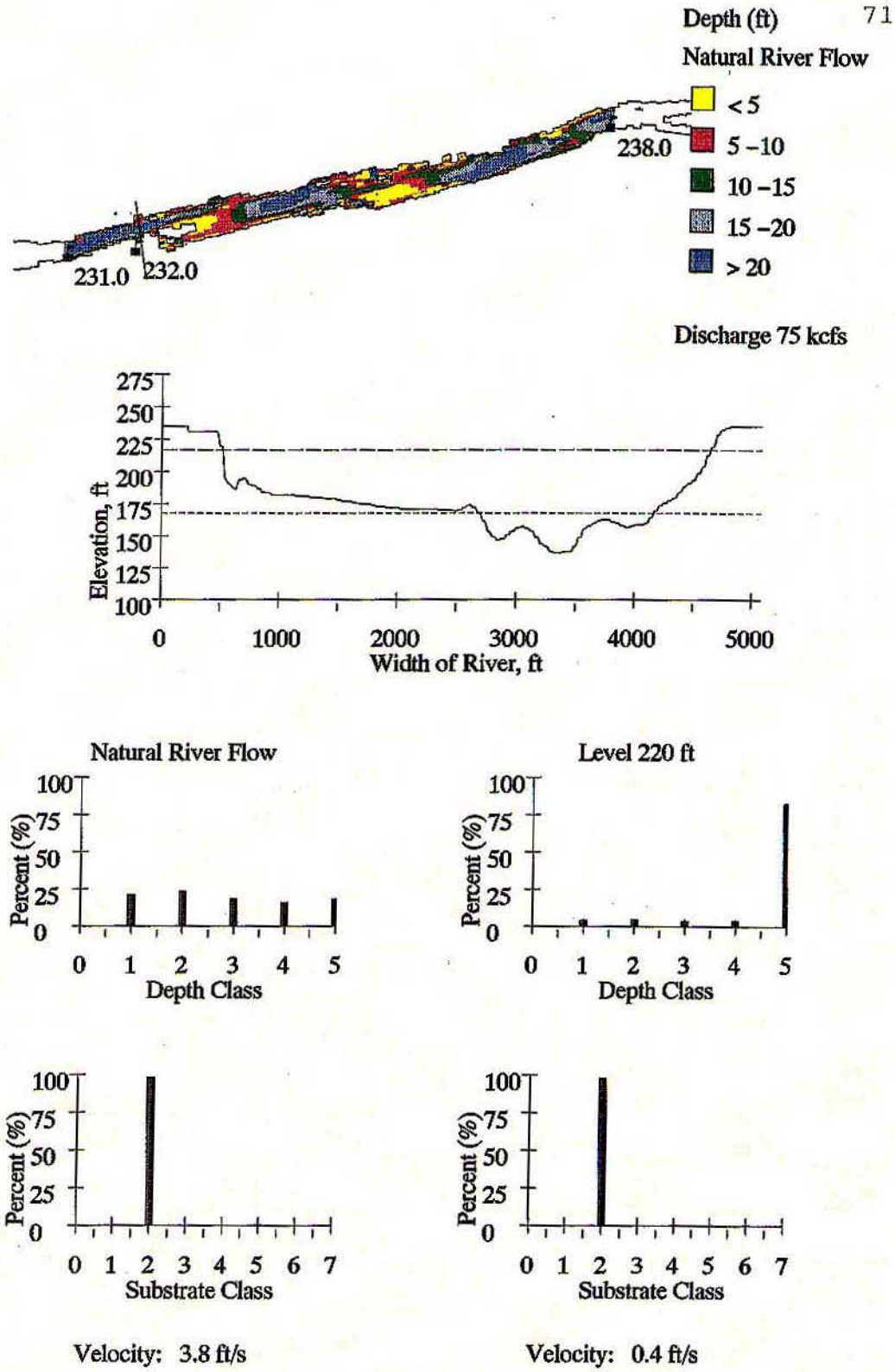

Figure 27. Physical habitat characteristics of a random transect (river mile 232.0) in Section 3 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

elevation. At both surface elevations, the current substrate would be largely silt and clay. The modal depth classes would be similar (>20 ft) at 220-ft elevation with about equal proportions of water depths at 165-ft elevation. The estimated water velocity when the surface elevation at the dam is 165 ft would be about 3.8 ft^{-S} compared to 0.4 ft^{-S} (about 10%) when the surface elevation at the dam is 220 ft.

Section 4.- Section 4 extended from RM 238 to 246 (Rkm 383 to 396; Figure 28). In this section, the elevation of the riverbed would be at approximately 190-ft MSL, developing two channels of nearly equal width at both the upstream and downstream portions of this section, if the surface elevation at John Day Dam were 165 ft. Depth characteristics in this section would consist mostly of water <5 ft if the surface elevation at the dam was 165-ft elevation and >20 ft if the surface elevation at the dame was 220 ft. River mile 245.7 (Rkm 395.6) was selected randomly to represent the river channel of section 4 when the surface elevation at John Day Dam is at 165 ft or 220 ft. The surface area at this site would be substantially reduced at 165-ft surface elevation (natural river level) compared to that at 220-ft elevation. The current substrate would be largely silt and clay at either surface elevation at the dam. The modal depth classes would be highly dissimilar with >20 ft predominating if the surface elevation at the dam were 220 ft; at a surface elevation of 165 ft, the modal depth classes would be about equal proportions of <5 ft and 5-10 ft. The remaining water depths at a surface elevation of 165 ft at the dam would be in the 10-15 ft and >20 ft classes. Estimated water velocity at a surface elevation of 165 ft at the dam (natural river level) would be about 5.8 ft^{-S} compared to about 0.8 ft^{-S} at 220-ft elevation.

Section 4, Rivermile 238 –246

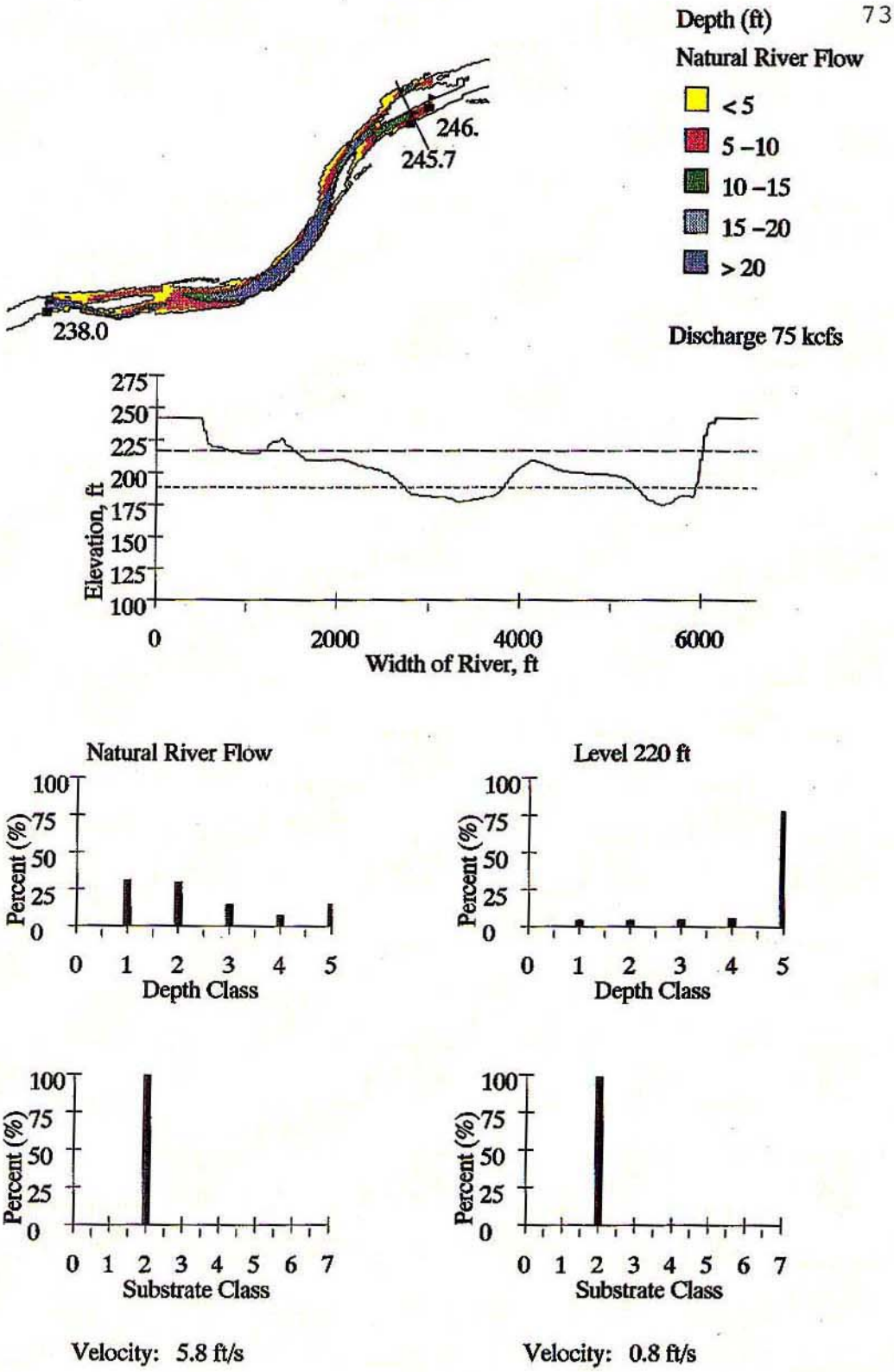

Figure 28. Physical habitat characteristics of a random transect (river mile 245.7) in Section 4 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

Section 5, Rivermile 246 –253

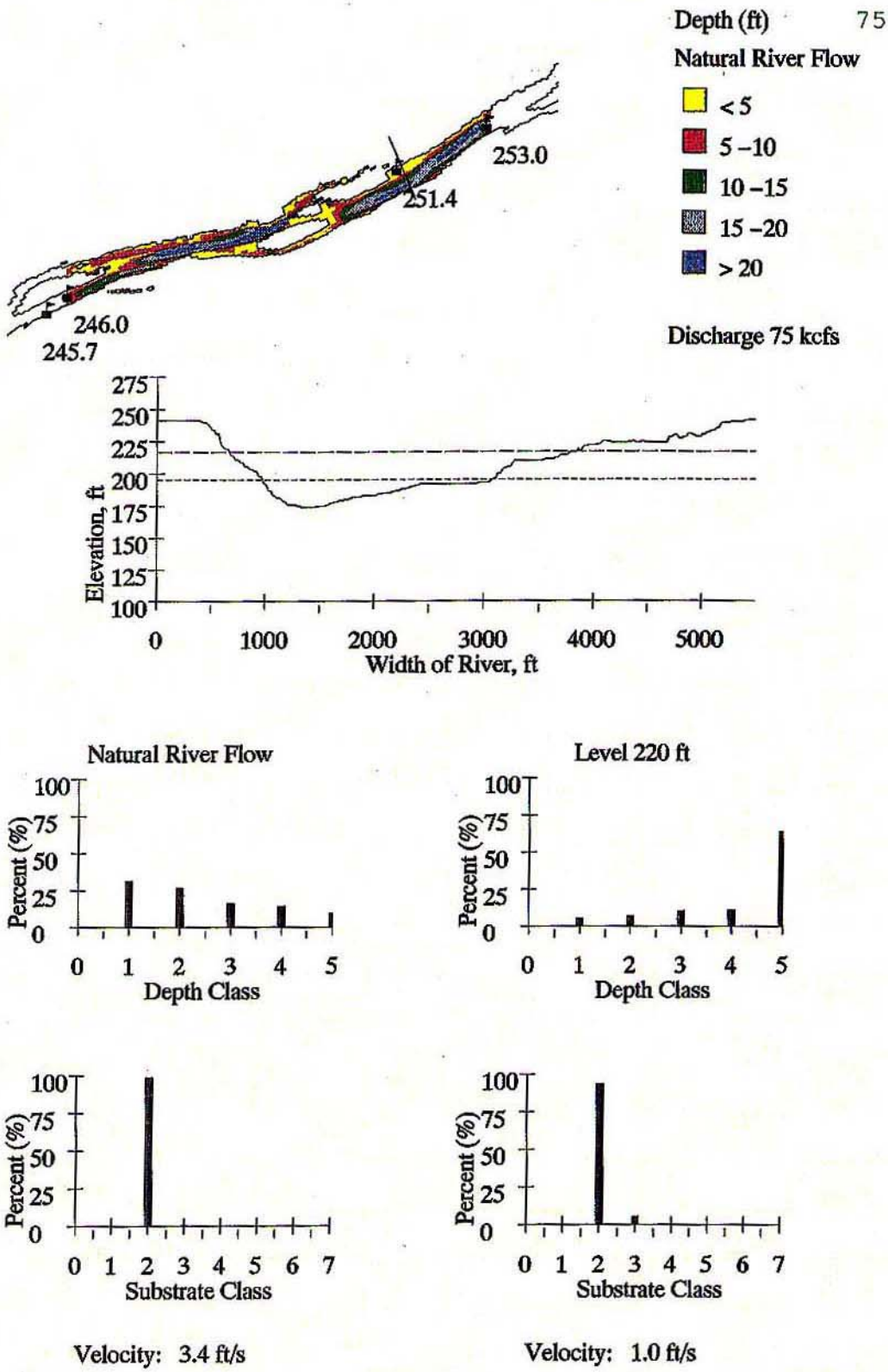

Figure 29. Physical habitat characteristics of a random transect (river mile 251.4) in Section 5 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

Section 6, Rivermile 253 –261

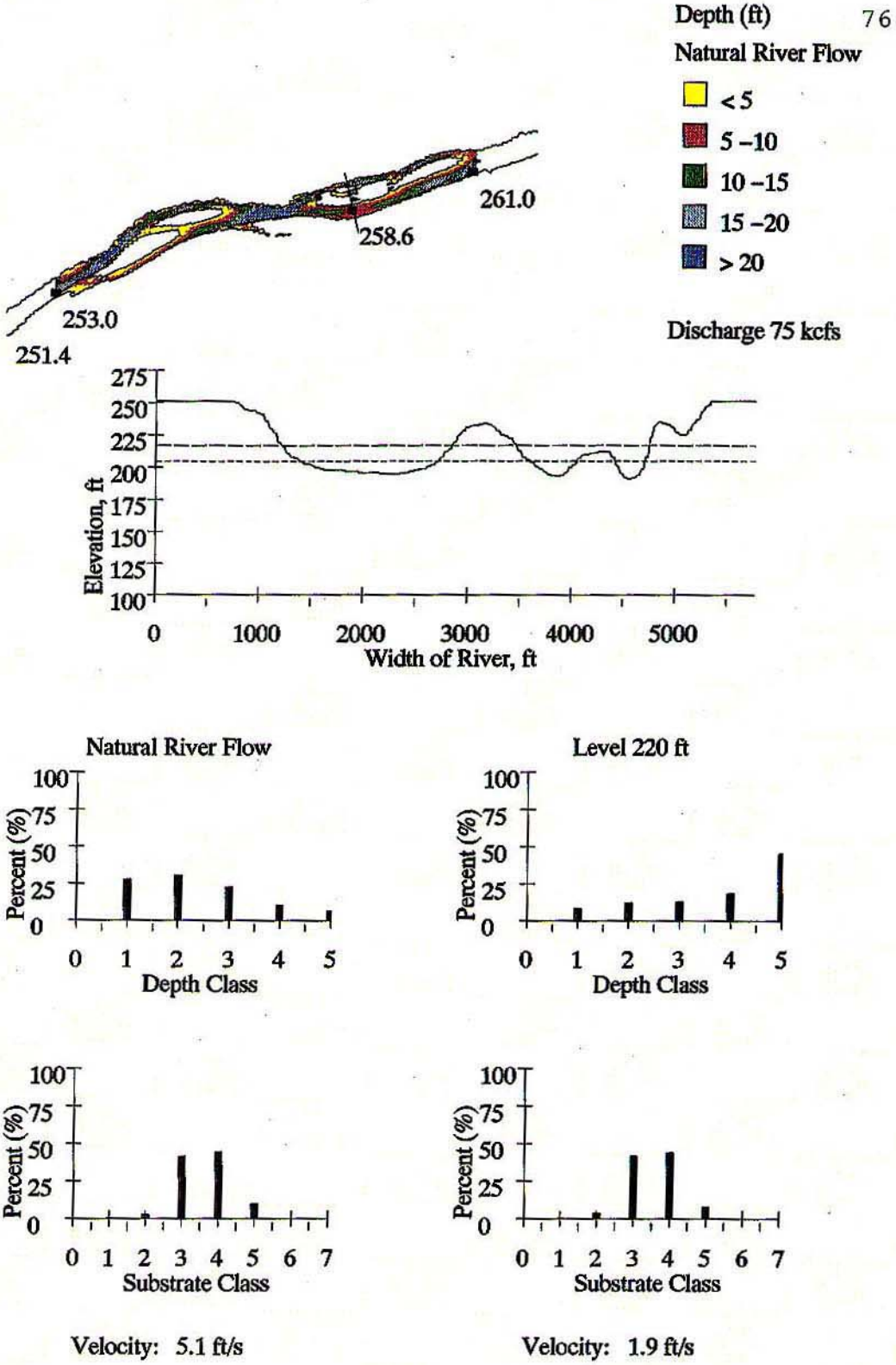

Figure 30. Physical habitat characteristics of a random transect (river mile 258.6) in Section 6 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

165 ft or 220 ft. The surface area at RM 258.6 would be reduced if the surface elevation at the dam were 165 ft (natural river level) or 220 ft. At either surface elevation at the dam, the current substrate would be largely sand and gravel with some cobble. The modal depth classes would be >20 ft if the surface elevation at the dam was 220 ft, and about equal proportions of water depths would occur from <5 ft to 15-20 ft if the surface elevation at the dam was 165 ft. The estimated water velocity at natural river level (165-ft surface elevation at the dam) would be about 5.1 ft^{-S}, compared to about 1.9 ft^{-S} at a surface elevation of 220 ft at the dam.

Section 7.- Section 7 extended from RM 261 to 268 (Rkm 420-431; Figure 31). The river channel that would exist at if the surface elevation at John Day Dam were at 220 ft or 165 ft (natural river level) showed a comparatively wide channel. Minimum water levels were nearly similar at both surface elevations at the dam. Depth characteristics would consist of approximately equal areas in all depth classes if the surface elevation at the dam was 220 ft, and would be predominately <5 ft if the surface elevation at the dam was 165 ft (natural river level). River mile 265.3 (Rkm 427.1) was selected randomly to represent the river channel of section 7 when the surface elevation at the dam is 165 ft or 220 ft. The surface area at RM 265.3 would be substantially reduced at either surface elevation at the dam. Additionally, the current substrate would be largely gravel if the surface elevation at the dam were at either 165 ft or 220 ft. The modal depth classes would be at about equal proportions if the surface elevation at the dam were 220 ft. If the surface elevation at the dam were 165 ft, the modal depth classes would be largely water <5 ft. Estimated water velocity at RM 265.3 would be about 9.0 t^{S} if the surface elevation at the dam were 165 ft, compared to 3.7 ft^{-S} if it were 220 ft.

Section 7, Rivermile 261 –268

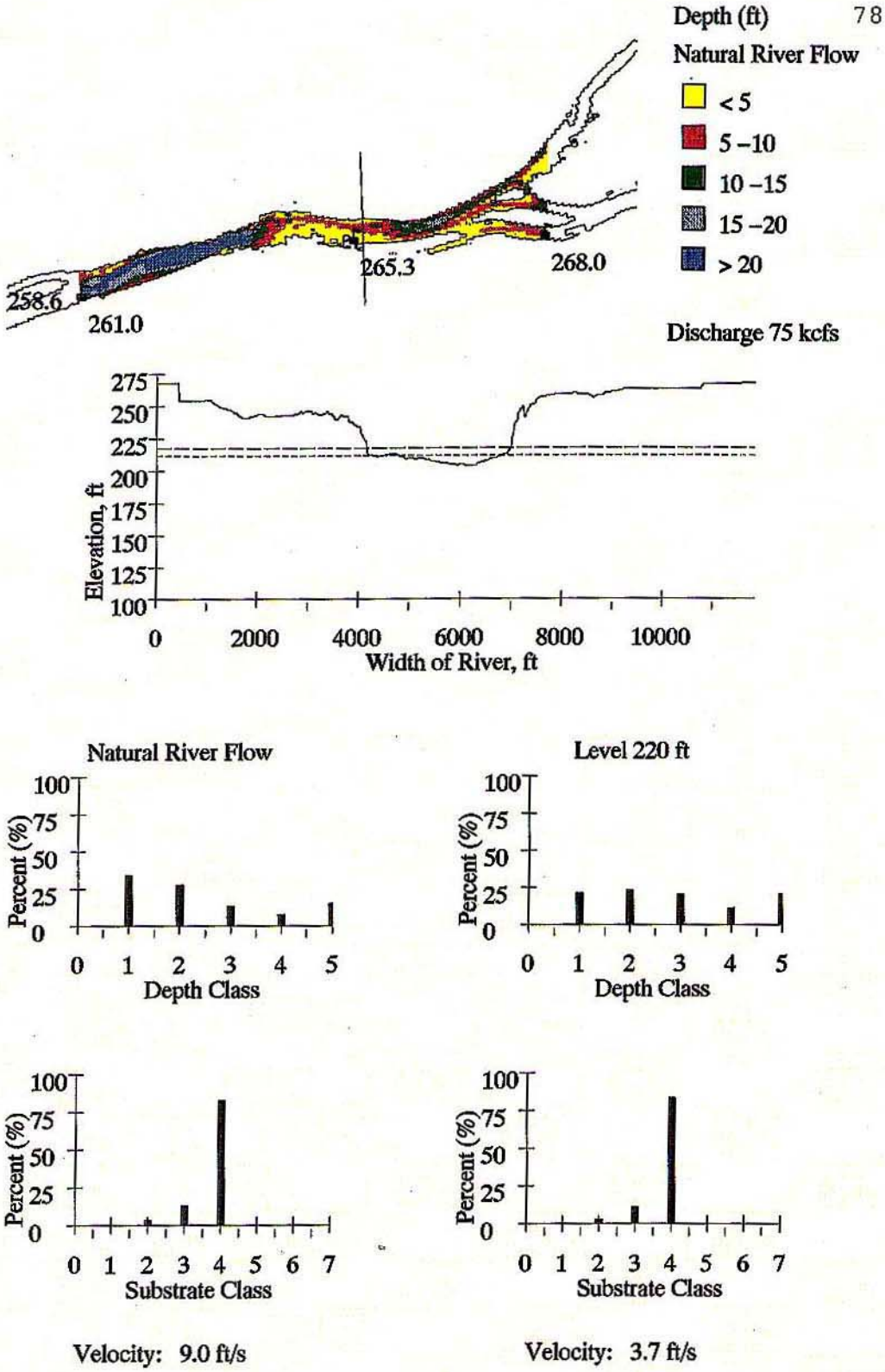

Figure 31. Physical habitat characteristics of a random transect (river mile 265.3) in Section 7 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

Section 8, Rivermile 268 –277

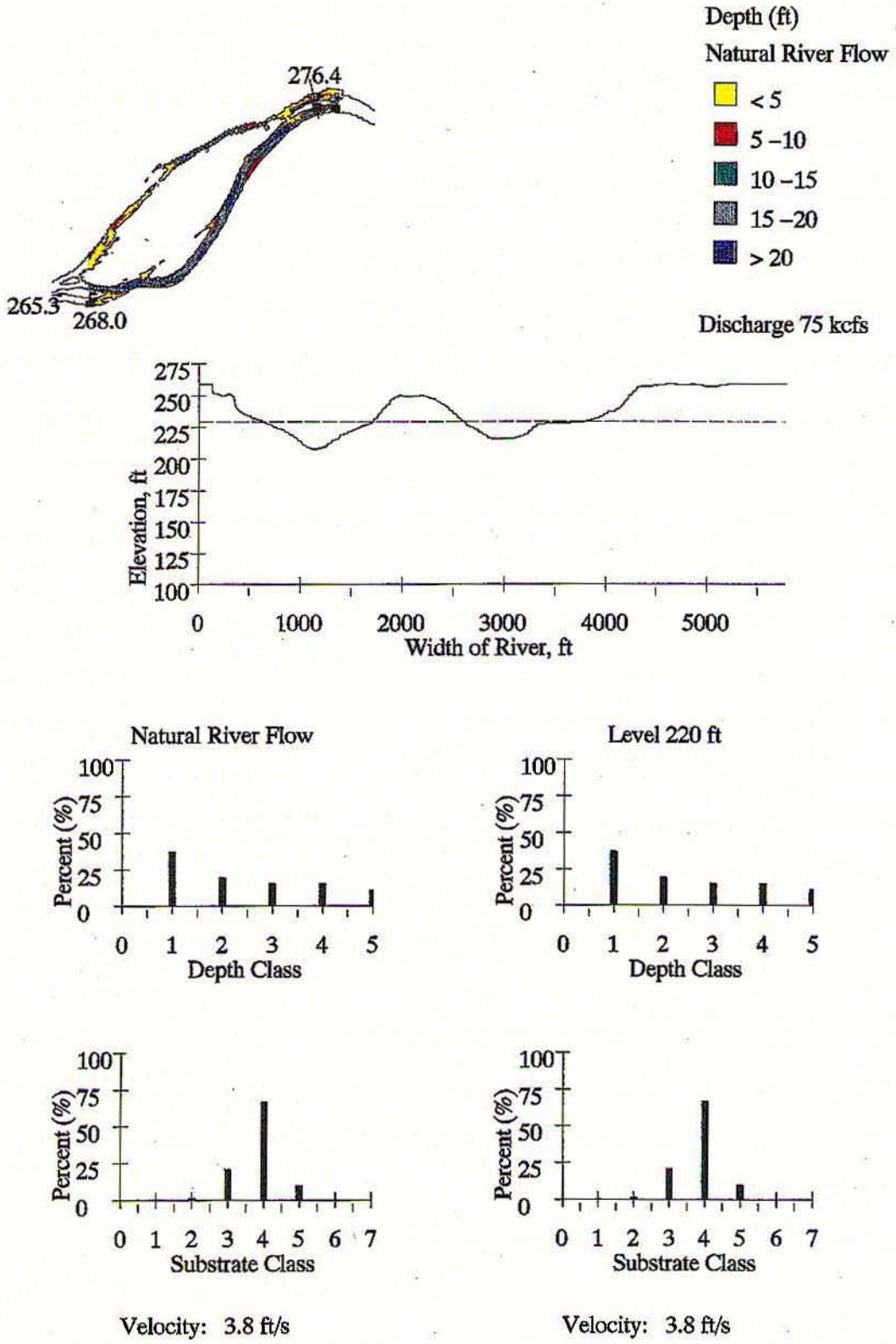

Figure 32. Physical habitat characteristics of a random transect (river mile 276) in Section 8 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

Section 9, Rivermile 277 –284

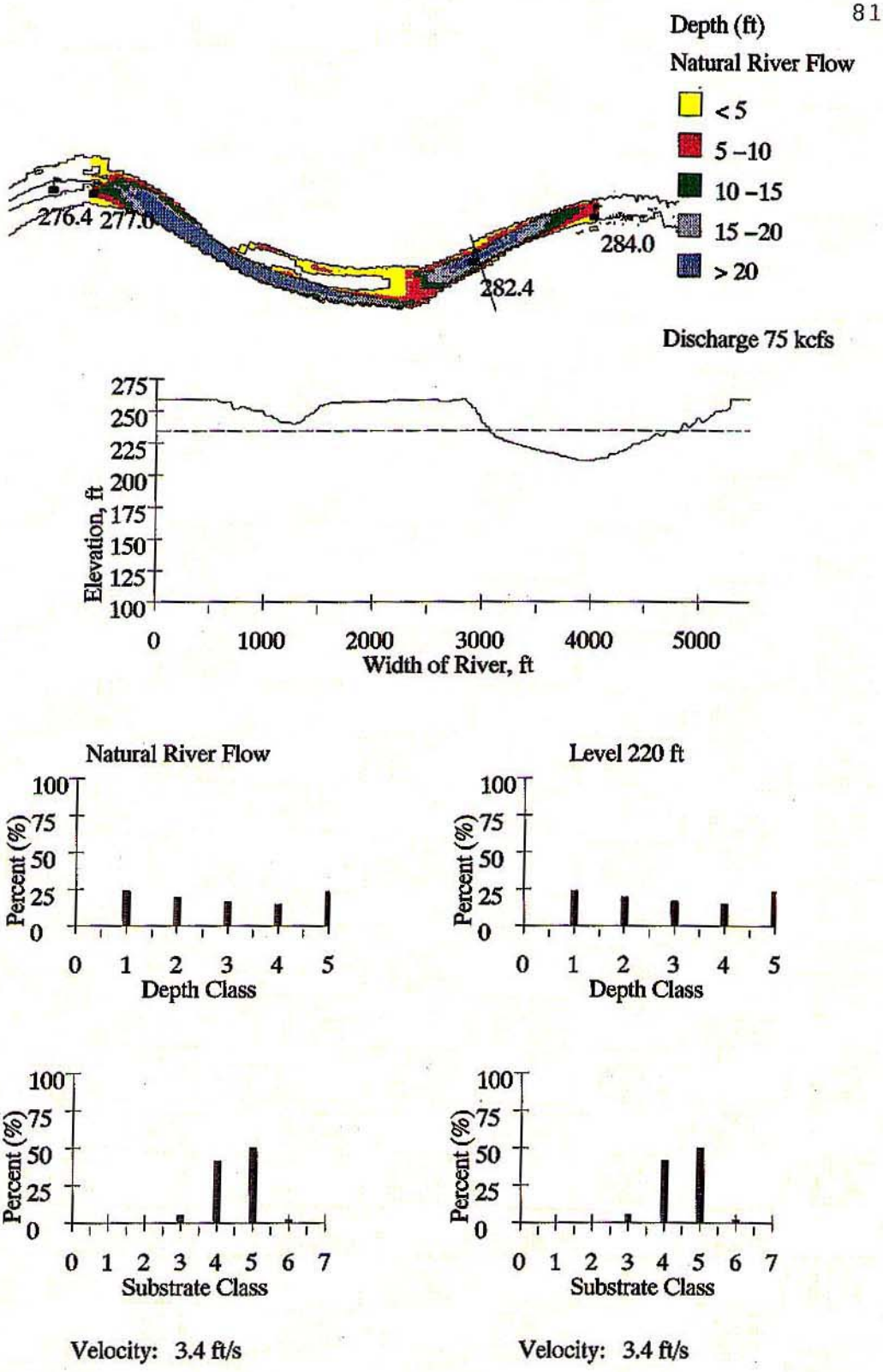

Figure 33. Physical habitat characteristics of a random transect (river mile 282.4) in Section 9 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

Section 10, Rivermile 284 –291

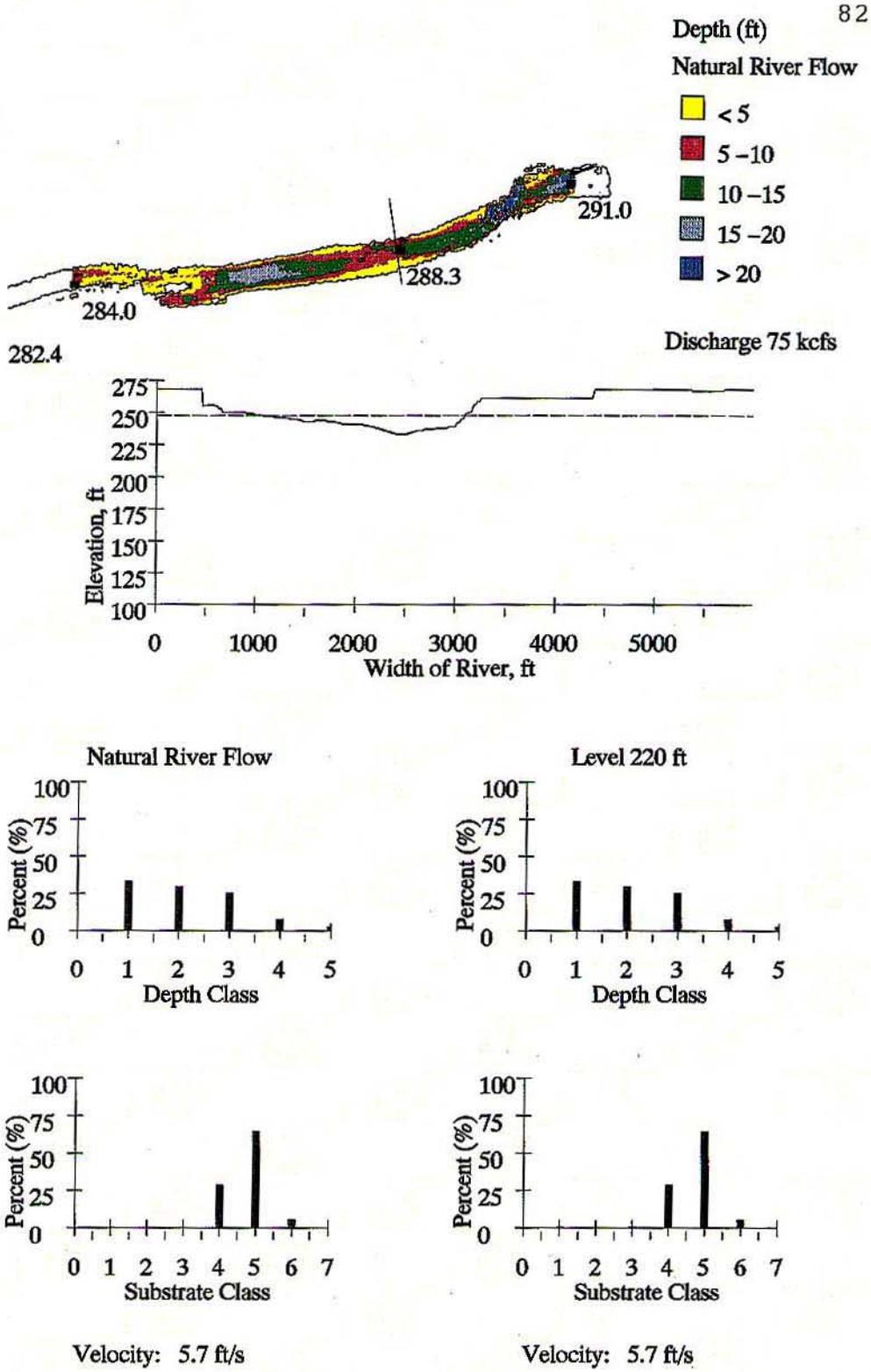

Figure 34. Physical habitat characteristics of a random transect (river mile 288.3) in Section 10 in John Day Reservoir. Cross-sectional profiles depict proposed drawdown elevations of 220 and 165 ft. Colors depict water depth. Substrate (1 = island , 2 = silt and clay, 3 = sand, 4 = gravel, 5 = cobble, 6 = boulder, 7 = bedrock) and depth (1 = <5 m, 2 = 5-10 m, 3 = 10-15 in, 4 = 15-20 m, 5 = >20 m) classes at 220 and 165 ft. elevations are shown in histograms.

surface elevation at RM 288.3 would be about 250-ft elevation if the surface elevation at the dam was either 165 ft or 220 ft, and the surface area at this site would be substantially reduced relative to the current elevation of the reservoir. If the surface elevation at the dam were either 165 ft or 220 ft, the current substrate in this section would be largely cobble, the modal depth class would be about equally distributed between <5 ft to >20 ft, and the estimated water velocity would be about 5.7 ft^{-S}.

Possible Responses of Biota to Habitat Changes with Drawdown

If drawdown were implemented, changes in physical habitat attributes (i.e., depth, velocity, and substrate) would result in changes in biotic communities. Changes in physical habitat would be of lesser magnitude if the level were drawn down to an elevation of 220-ft MSL than to the natural river level of 165 ft. A consequence of the lesser changes in habitat at 220-ft elevation would result in fewer changes in the biota. Reservoir-like habitat conditions would prevail in the downstream sections and the biota would exhibit fewer changes in community structure. Specific changes that are anticipated to occur at each of the water levels are largely speculated from observed differences between the lotic and lacustrine sections of the Snake River and based on current studies of the Columbia River reservoirs and tailwaters.

Elevation of 220-ft MSL

Drawdown of John Day Reservoir to an elevation of 220 ft at John Day Dam would create both reservoir and lotic habitats; downstream areas would be, in part, generally similar to present conditions in the lower 15 miles of the reservoir. Upstream at RM 246

(Rkm 396), however, physical habitat conditions would be altered. Water velocities would increase, and with higher water velocities the overall substrate size would increase. Decreases in depth would enhance light penetration to the substrate that would stimulate growth of attached benthic algae as the primary producer. In downstream areas, phytoplankton would still predominate as the primary producer. In channel areas upstream of RM 246, macroinvertebrate composition would also change to more of an attached community and probably be more similar to upstream areas in the Hanford Reach, Columbia River, where midges and caddisflies predominate. Downstream benthic communities would continue to consist of mainly burrowing midges and oligochaetes as observed in Lower Granite Reservoir. Attached forms of macroinvertebrates are probably more available than burrowing forms as fish food organisms. Presence of attached macroinvertebrates would probably increase food abundance during the downstream juvenile salmonid migrations.

The fish community structure would change considerably and several species would likely predominate in their sections of preferred habitat. Herbivorous species such as chiselmouth *Acrocheilus alutaceus*, and omnivorous suckers and redside shiners would probably dominate the fish fauna in the "more riverine" habitat, upstream of RM 246. Northern pikeminnow abundance probably would not change appreciably throughout the system as has been observed in the Snake River in lotic versus lacustrine habitats. Walleye abundance would decrease in upstream areas as the combination of higher velocities and higher light penetration would create less favorable habitat. Walleye population abundance would decline, as recruitment would decrease as a result of the higher velocities throughout the system and less planktonic food production, which are

critical for juvenile walleye survival. It is anticipated that smallmouth bass abundance would not change appreciably throughout the system.

Predation on rearing or migrating juvenile salmonids has received considerable attention because of several mechanisms related to impoundment (Gray and Rondorf 1986). Each dam acts as a funnel, concentrating salmonids into the forebay. Below the dam, tailraces provide a steady supply of migrating fish, a proportion of which are potentially injured or disoriented. Predation is exacerbated during low flow years when predation seems highest, probably related to improved prey detection capabilities by predators, slower movement by salmonids through reservoirs, and generally more suitable habitat for predators to facilitate predation. Impoundments have slowed smolt emigration and decreased turbidity, factors that increase the likelihood of predation. Similarly, the reservoirs have enhanced habitat for non-native predators such as channel catfish and walleye.

Numerous factors have the potential to affect the magnitude of salmonid predation in the John Day system. Water clarity, water temperature, and predator and prey sizes are all important when attempting to assess predation. Under conditions of low water clarity, visual predators such as the smallmouth bass, northern squawfish, and crappies have less ability to see prey at a given distance and require closer proximity to see their prey (Obrien and Vinyard 1976). Because fishes are ectothermic vertebrates, their metabolic activities are highly dependent upon water temperature; numerous fishes such as smallmouth bass typically do not actively search for food at temperatures less than 10° C (Coble 1975). Feeding activity of channel catfish is similarly low at low water temperatures but not nearly as low for northern pikeminnow whose feeding activity is

less affected by low water temperatures. Application of these principles to assess salmonid prey consumption suggests that far less prey are consumed early in the downstream migration of yearling chinook salmon and juvenile steelhead because their downstream migration coincides with lower water temperatures and higher turbidities. Downstream migration of these salmonid fishes generally coincides with peak flow events; higher flows are fed by snow melt that suspend more particles that creates less than favorable conditions for sight-feeding ectothermic predators. Conversely, as flows decrease in late spring, water temperatures increase and turbidities decrease, increasing the potential for higher predation. For this reason, predation on subyearling chinook salmon is considered to be more significant in both the lower Snake River reservoirs (Curet 1994; Anglea 1997) and John Day Reservoir (Poe et al. 1991) because they out-migrate in late spring and early summer.

Effects of drawdown on the John Day ecosystem can at best be speculated. With drawdown, more shallow water habitat will be exposed to desiccation. As with the drawdown of Lower Granite Reservoir, backwaters will be exposed, fish and other shallow water biota will be exposed, and mortalities associated with stranding and ultimately desiccation will initially occur. Water velocities will be increased and water retention times will be shortened. Based on habitat requirements of species during life history stages, species that require low water velocities for enhanced food production, such as walleye and some sunfishes, will be disadvantaged and may exhibit lower survival. Survival of other species may be enhanced under conditions of higher water velocity, such as white sturgeon. Although juvenile white sturgeon may require backwater habitat with low water velocities for rearing, suitable rearing habitat would

exist in backwaters under a drawdown elevation of 220 ft.

Elevation of 165-ft MSL

The proposed drawdown of 165-ft MSL would alter the habitat considerably, changing the entire John Day system from lacustrine to riverine. Changes in physical habitat from lacustrine to riverine would alter the substrate from that predicted relative to the current reservoir conditions to substrata that would ultimately stabilize and not be altered by higher velocities associated with drawdown. Under the predicted water velocities at various locations in the system, finer substrates would be moved through the system and larger ones, such as gravel, cobble, and boulders, would prevail. In the backwaters, substrate would be composed of finer materials. Movement of these fines would increase turbidity.

Another change in the physicochemical habitats would probably be water temperature. Maximum water temperature would probably increase slightly above the current maximum of about 22° C, although water temperatures would probably not exceed the upper thermal limit of salmonid fishes (~26° C). In the fall, water temperatures would probably decrease faster, but because of the immediate upstream location of McNary Reservoir, water temperatures entering the John Day system would be higher than would normally occur under "complete" riverine conditions.

The major changes in the physical habitat would result in substantial changes to the biota in the John Day system if the surface elevation at John Day Dam were 165 ft. Under impoundment, the John Day system is largely composed of herbivorous and omnivorous fishes that feed on phytoplankton and attached algae and burrowing and

attached invertebrates. Under the 165-ft drawdown scenario, the food production for fishes would shift entirely to an attached community of plants and benthos. Attached benthic algae would become the principal primary producer and attached organisms would have higher diversity than what currently exists under impoundment. Chronomids that are attached rather than burrowing forms, caddisflies, mayflies, and stoneflies would exist in considerably higher numbers than they do under impounded conditions. The fish community would rapidly shift to one more historically representative of resident fishes found in that section of the Columbia River, and there would be a substantial reduction in many of the introduced fishes. Native cyprinids (chiselmouth, redside shiners, northern pikeminnow, and peamouth *Mylocheilus caurinus*) and suckers would constitute the bulk of the fish community both numerically and in biomass. For example, suitable habitat for species like crappies, pumpkinseed, and bullheads (*Ameiurus* spp.) would be largely eliminated with return to the natural river condition. Yellow perch and its habitat would be nearly eliminated also.

Riverine specialists such as the native cyprinids and cottids would probably exhibit significant increases in abundance under this alternative. Deeper pools could support salmonid fishes.

White sturgeon is another riverine species that would benefit from drawdown to 165-ft elevation. White sturgeon have been found to be higher in abundance in lotic areas (e.g., upstream end of Lower Granite Reservoir, Lepla 1994; downstream of Bonneville Dam, Beamesderfer et al. 1994) and their abundance would increase if the natural riverine condition were restored. White sturgeon spawning and recruitment would probably increase, as their embryos have been found in areas with water velocities of

about 2.0 m^{-S} (Parsley et al. 1993). These water velocities coincide with those predicted throughout most of the restored section.

Walleye would be one species to drastically change in abundance. The current population in John Day Reservoir is probably recruitment limited as it has all the characteristics (i.e., rapid growth, low abundance). Currently, recruitment of walleye from successful spawning and rearing in John Day Reservoir probably occurs during years of lower flows, create continuous warming and result in good planktonic food production for fry. Under the restored river condition, high water velocities could preclude successful embryo lodging in the interstices of the substrate and could further reduce planktonic food production. Shallower waters would also enhance light penetration into the water; this would reduce diurnal activity of walleye because of their negative light response (Scott and Crossman 1979).

Channel catfish abundance would probably not change under restored river conditions. Although channel catfish are considered to be a lacustrine species, unpublished data suggest that adult channel catfish prefer water velocities <1.0-ft^{-S} (Kallemyn and Novotny, U.S. Fish and Wildlife Service, File No. R0024). Juvenile channel catfish prefer slightly higher water velocities than adults (to 2.0-ft^{-S}; Hilgert 1981). Although average water velocities are predicted to be considerably higher in the restored river channel, suitable habitat would exist on the areas adjacent to the main channel, backwaters, and side channels. Also, predicted water velocities were for mid-water column locations, and velocities near the bottom would be considerably reduced.

Smallmouth bass abundance, as with channel catfish, would probably not change under the restored river condition. Under current impounded habitat conditions, water

temperature probably limits the smallmouth bass population from expanding. Highest summer water temperatures of 22° C in John Day Reservoir are 7° C below the 29° C preferred water temperature indicated by Coble (1975) for smallmouth bass. Lower water temperatures reduce growth and juvenile survival. Under the natural river level option (165-ft surface elevation at John Day Dam), water temperatures may increase, which could enhance juvenile smallmouth bass recruitment. Gradient of the river bottom in the John Day system would average about 0.03%, less than those considered optimum for smallmouth bass at 0.08%-0.46%, although estimated mean velocities would exceed those considered optimum (Paragamian 1987, Edwards et al. 1983). Rankin (1986) indicated that preferred water velocities for smallmouth bass were < 0.5-ft^{-S} and fish were rarely found in waters >0.67 ft^{-S}. Water velocities in the thalweg would exceed the preferred water velocities for smallmouth bass, although suitable habitat would exist on the areas adjacent to the main channel, backwaters, side channels, and downstream side of boulders and other obstructions. Nelle (1999) reported that abundance of smallmouth bass in the flowing section of the Snake River, upstream of Lower Granite Reservoir, is higher than that in the reservoir (Anglea 1997; Naughton 1998).

Northern pikeminnow could increase in abundance. Some researchers suggest that their abundance may increase relative to current abundance under impounded conditions. Nelle (University of Idaho, Moscow, Idaho, personal communication) and Petersen (Jim Petersen, USGS, Cook, Washington) have both suggested that abundance of northern pikeminnow is similar or slightly higher in the lotic portion of the Snake River, upstream of Lower Granite Reservoir. Also, Ward et al. (1995) found that the highest abundance of northern pikeminnow was downstream of Bonneville Dam of all the sections sampled. A

number of possible explanations exist for the apparent increase in abundance of northern pikeminnow beyond that of lotic versus lacustrine conditions, but current information suggests the population in John Day Reservoir would not decrease.

Possible Studies to Monitor Effects of Drawdown

To further assess effects of drawdown in John Day Reservoir several studies should be conducted:

1. To evaluate predation efficiency under lotic conditions, especially during the fall chinook rearing period;
2. To evaluate predator distribution in relation to drawdown;
3. To determine factors limiting the abundance of fish predators in John Day Reservoir and assess how populations respond to changes in habitat;
4. To assess factors affecting year-class strength of predators including smallmouth bass, northern pikeminnow, and walleye;
5. To evaluate species interactions and effects of environmental factors affecting year-class strength of resident fishes (predators) in John Day Reservoir;
6. To identify factors that limit the abundance of resident fishes in lotic conditions to compare with those under impoundment; and
7. To assess the role of alternative prey items in affecting fish predator abundance.

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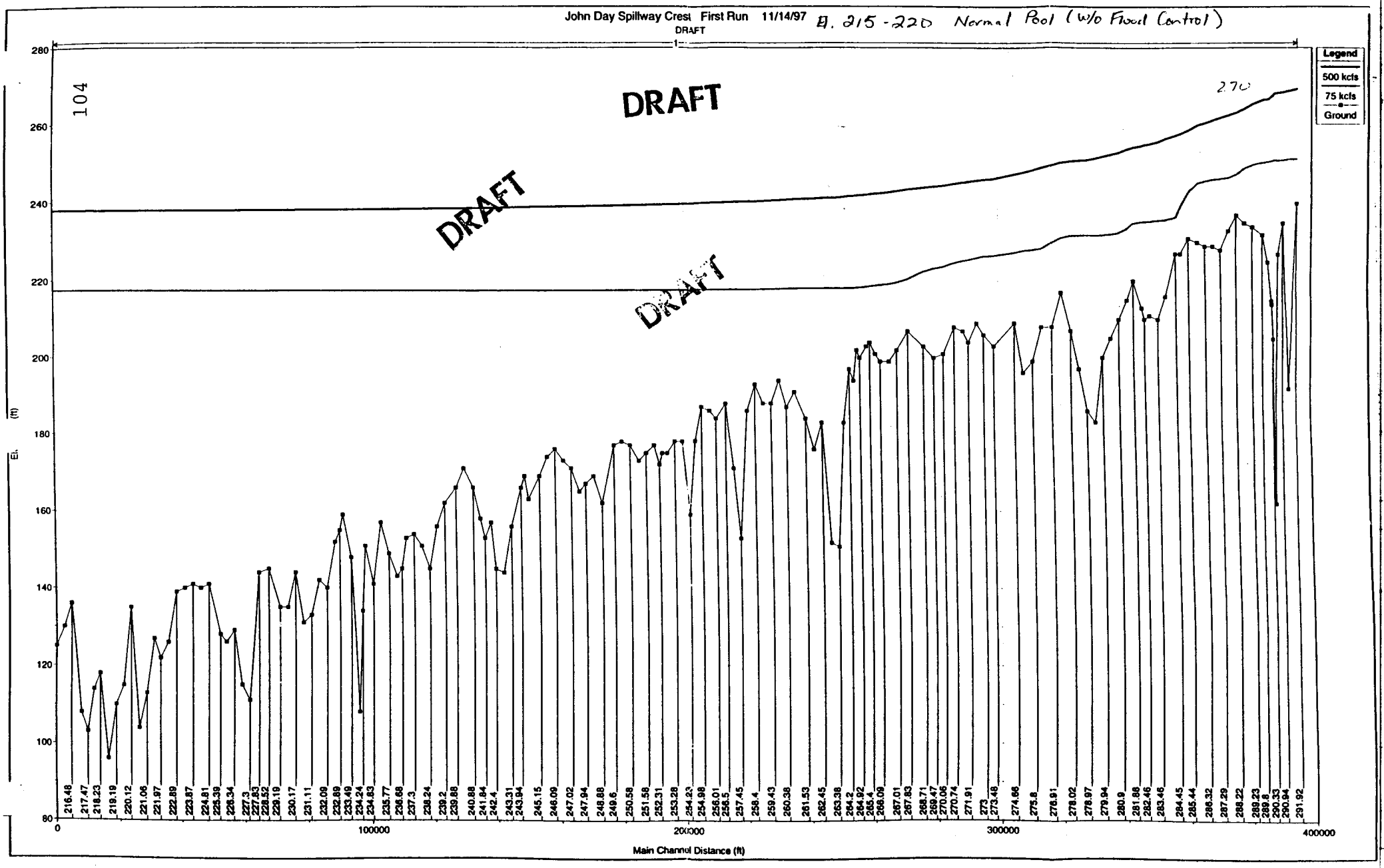
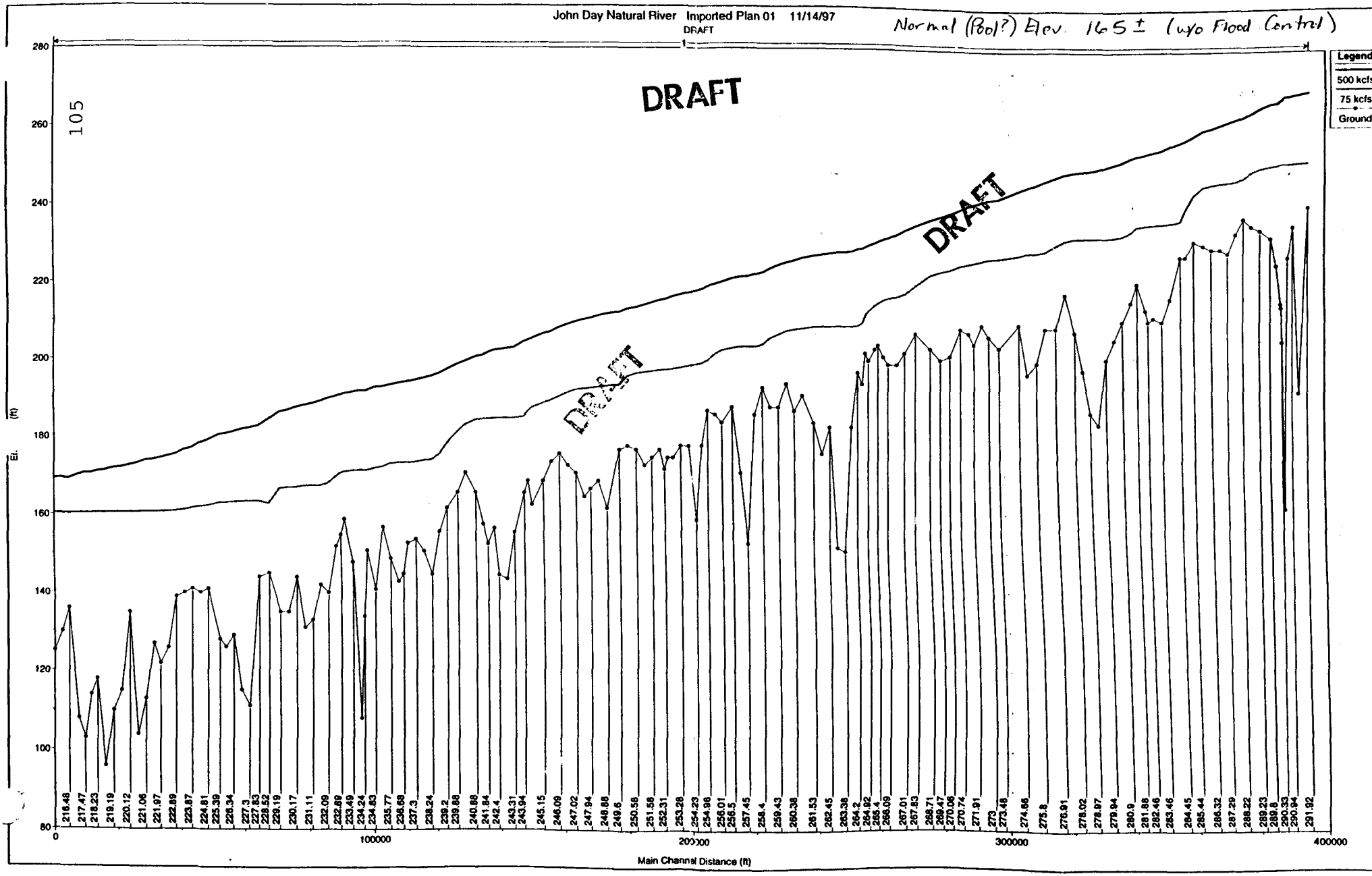
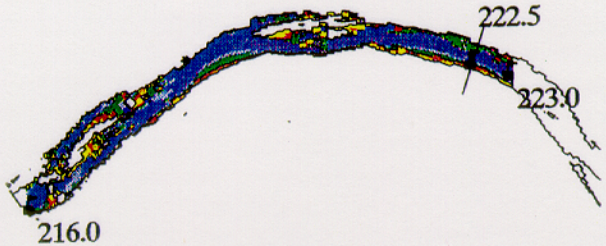


Appendix Figure 1. Bottom surface elevations and predicted water levels at 215 –220 ft MSL for John Day Reservoir, Columbia River.



Appendix Figure 2. Bottom surface elevations and predicted water levels at 165 ft MSL for John Day Reservoir, Columbia River.

Section 1, Rivermile 216 –223



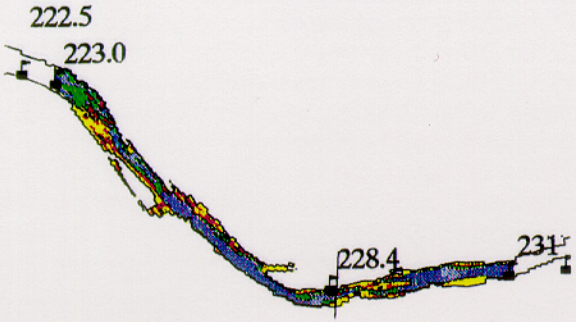
Depth (ft)

Natural River Flow

- < 5
- 5 –10
- 10 –15
- 15 –20
- > 20

Discharge 75 kcfs

Section 2, Rivermile 223 –231



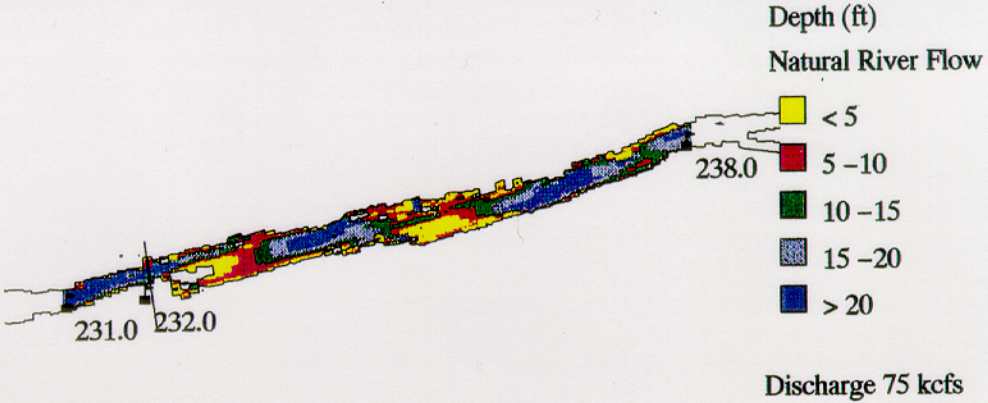
Depth (ft)

Natural River Flow

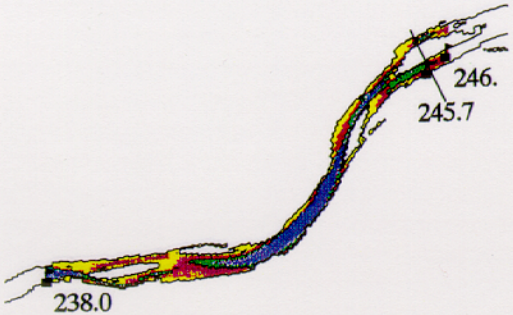
- < 5
- 5 –10
- 10 –15
- 15 –20
- > 20

Discharge 75 kcfs

Section 3, Rivermile 231 –238



Section 4, Rivermile 238 –246



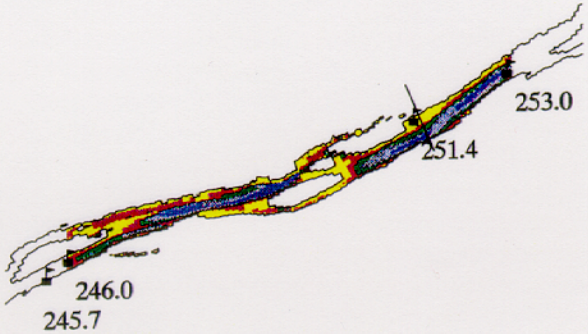
Depth (ft)

Natural River Flow

- < 5
- 5 –10
- 10 –15
- 15 –20
- > 20

Discharge 75 kcfs

Section 5, Rivermile 246 –253



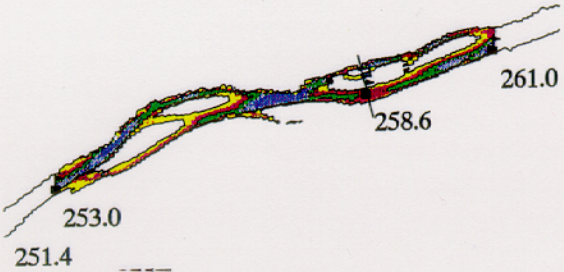
Depth (ft)

Natural River Flow

- < 5
- 5 –10
- 10 –15
- 15 –20
- > 20

Discharge 75 kcfs

Section 6, Rivermile 253 –261



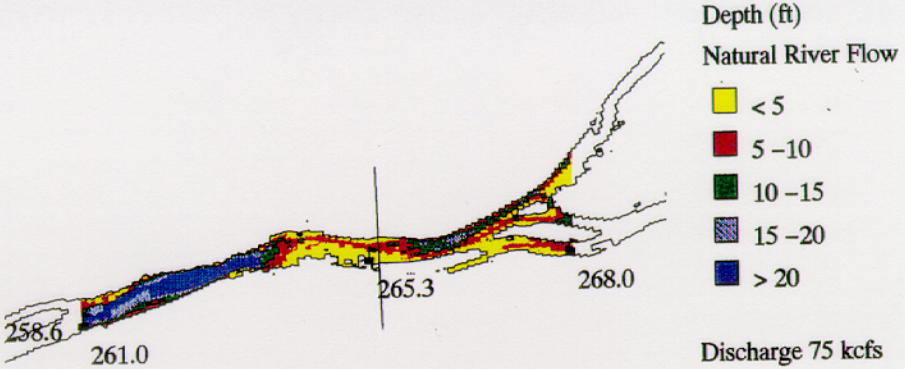
Depth (ft)

Natural River Flow

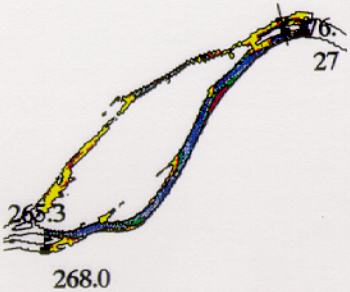
- < 5
- 5 –10
- 10 –15
- 15 –20
- > 20

Discharge 75 kcfs

Section 7, Rivermile 261 –268



Section 8, Rivermile 268 –277



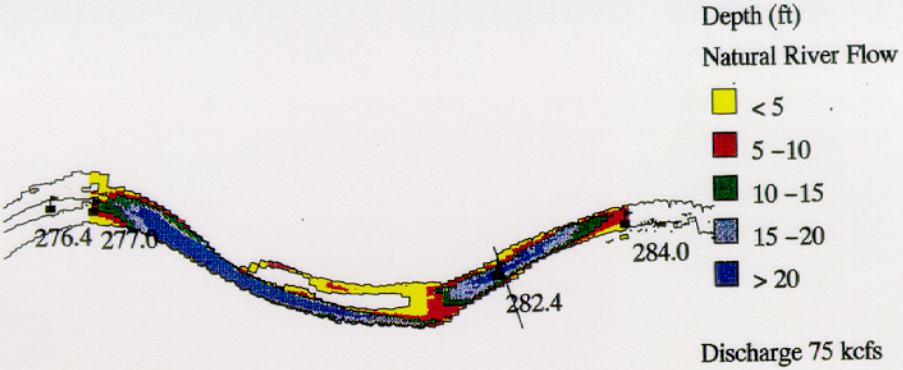
Depth (ft)

Natural River Flow

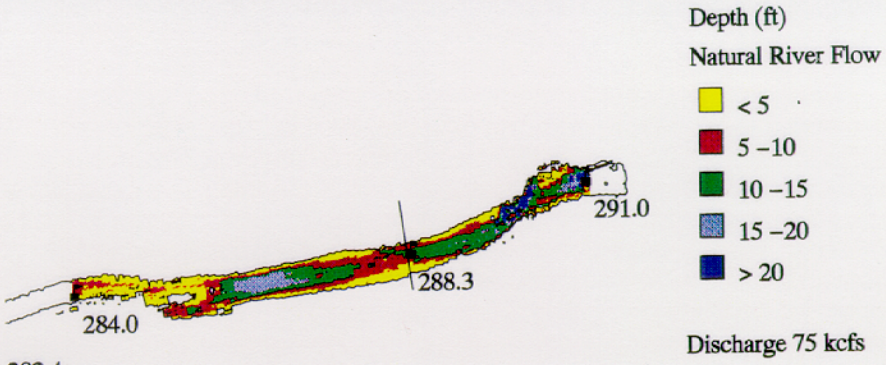
- < 5
- 5 –10
- 10 –15
- 15 –20
- > 20

Discharge 75 kcfs

Section 9, Rivermile 277 –284



Section 10, Rivermile 284 –291



Attachment D: *Analysis of Smolt Travel Time and Survival Through the John Day Project Using Available PIT-tag Data Skalski and Townsend.*

Analysis of Smolt Travel Time and Survival Through the John Day Project Using Available PIT-tag Data

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13 May 1999

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I. Introduction

The purpose of this report is to summarize available PIT-tag information on the outmigration dynamics of salmonid smolts through the John Day Dam project—in particular, to estimate smolt travel time and survival through the John Day Dam project. In addition, to investigate possible relationships between travel time or survival and ambient river conditions during outmigration.

John Day Dam is located at R km 347 and downriver of McNary Dam (R km 470). PIT-tag detectors at both McNary and John Day Dams permit estimation of travel time from the last detection in the bypass at McNary Dam to the PIT-tag detectors at John Day Dam. The PIT-tag detectors at Bonneville Dam (R km 234) in conjunction with the detectors at McNary and John Day Dams permit estimation of smolt survival through the John Day project. Low detection probabilities of tagged smolt at the Lower Columbia PIT-tag detection facilities (i.e., $p < 0.10$), however, severely limits the precision and availability of tag-recovery information. The purpose of this report is to summarize available baseline information on smolt outmigration dynamics prior to a possible pool drawdown at John Day Dam. Available information as will be seen is largely limited to baseline data beginning in 1998. Prior to 1998, either detection probabilities were too low or detection facilities were unavailable to provide the necessary detections to estimate travel times and survival.

II. Methods

A. Available PIT-tag Data

To estimate travel times, individual PIT-tagged smolt need to be detected at both McNary and John Day Dams in order to estimate travel times through the John Day project. Table 1 summarizes the numbers of PIT-tag detections at McNary Dam, John Day Dam, and both dams by salmonid stocks for 1997-98. Inspection of Table 1 suggests mean travel times can only be reliably and precisely estimated from 1998 PIT-tag data.

Table 1. Summary of PIT-tag detections at McNary and John Day Dams available for travel time estimation.

Year	Stock	McNary	John Day	Both Dams
1997	Chinook			
	Spring	4,521	66	7
	Summer	3,246	57	7
	Fall	7,454	38	7
	Steelhead	7,702	664	63
1998	Chinook			
	Spring	17,653	8,563	2,126
	Summer	5,437	3,748	694
	Fall	30,120	14,076	3,277
	Steelhead	4,737	8,342	1,346

For survival estimation, smolt salmonid upstream needed to be potentially detected at McNary, John Day, and Bonneville Dams. Criteria were established that at least 20 individuals from a tag-group had to be detected at each of McNary, John Day, and Bonneville Dams to be sufficient for survival estimation. These criteria were considered a minimum to consider using the release group in PIT-tag survival estimation. So few detections may still result in inaccurate and/or imprecise survival estimates through the John Day project. Table 2 summarizes numbers of release groups that meet minimum sample size requirements. Review of available information from the PTAGIS database operated by the Pacific States Marine Fisheries Commission suggests only PIT-tag data for 1998 provide adequate numbers for subsequent statistical analysis.

B. Estimating Mean Travel Time

Travel time of PIT-tagged salmonids through the John Day project will be summarized using histograms of individual travel times over the season. Mean travel time will be summarized over tagged smolt using the harmonic mean $(\bar{t}_H)$ and its associated variance estimate where

$$\bar{t}_H = \frac{1}{\frac{1}{n}\sum_{i=1}^{n}\frac{1}{t_i}} = \frac{n}{\sum_{i=1}^{n}\frac{1}{t_i}}$$

where

t_i = travel time from McNary Dam to John Day Dam for the ith smolt recovered $(i = 1, \ldots, n)$.

The variance for the harmonic mean travel time can be estimated by the approximate formula

Table 2. Summary of the number of release groups with a minimum of 20 detections at each of McNary, John Day, and Bonneville Dams.

Year	Stock	Number of Release Groups
1997	Chinook	
	Spring	0
	Summer	0
	Fall	0
	Steelhead	0
1998	Chinook	
	Spring	107
	Summer	54
	Fall	56
	Steelhead	53

$$\hat{Var}(\bar{t}_H) = \frac{\left(\frac{s_{1/t}^2}{n}\right)}{\left(\frac{1}{n}\sum_{i=1}^{n}\frac{1}{t_i}\right)^4}$$

and where

$$s_{1/t}^2 = \frac{\sum_{i=1}^{n}\left(\frac{1}{t_i} - \hat{u}_{1/t}\right)^2}{(n-1)},$$

$$\hat{u}_{1/t} = \frac{1}{n}\sum_{i=1}^{n}\frac{1}{t_i}.$$

Asymptotic $(1-\alpha)$ 100% confidence interval for harmonic mean travel time will be computed as

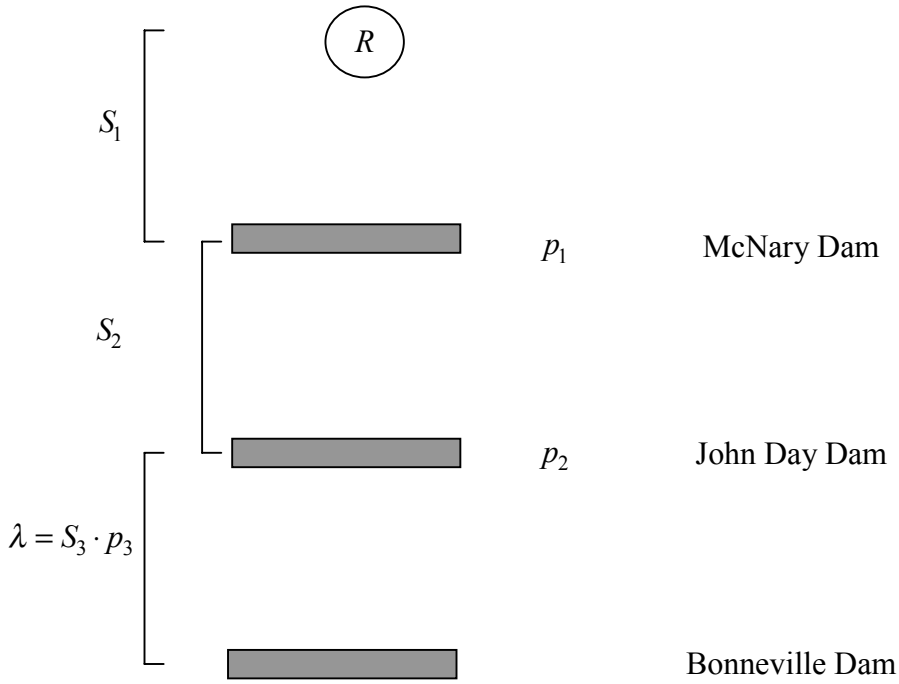
$$\bar{t}_H \pm Z_{1-\frac{\alpha}{2}}\sqrt{\hat{Var}(\bar{t}_H)}$$

for both PIT-tagged and radio-tagged smolt. Arithmetic mean travel time $(\bar{t})$ and its associated variance estimate will also be computed for the purpose of comparison with other studies.

C. Estimating Smolt Survival

The single release-recapture model of Burnham et al. (1987), a special case of the release-recapture model of Cormack (1964), will be used to estimate smolt survival through John Day project. Using the PIT-tag detection and rerelease facilities at McNary, John Day, and Bonneville Dams, survival through John Day project will be calculated for upstream smolt releases (Figure 1). Survival through the John Day project was also calculated based on daily accumulations of PIT-tagged smolt released from McNary Dam (Figure 2). These generated release groups were formed in order to

Figure 1. Schematic of PIT-tag releases (R) and associated survival rates at and below McNary Dam. Survival probabilities denoted by S and detection probabilities denoted by p.

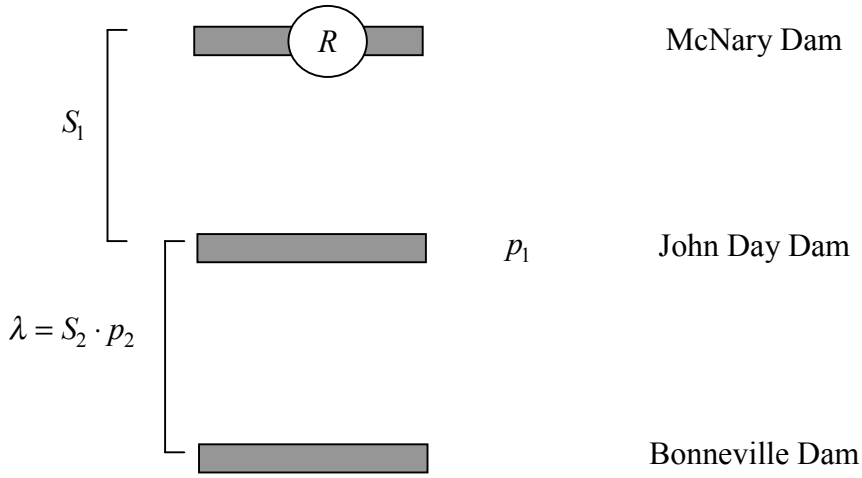


S_1 = smolt survival from point of release to the tailrace of McNary Dam.

S_2 = John Day Dam project survival, i.e., survival from the tailrace of McNary Dam to the tailrace of John Day Dam.

λ = joint probability of survival from the John Day Dam tailrace to Bonneville Dam and being detected at Bonneville Dam.

Figure 2. Schematic of PIT-tag releases (R) from McNary Dam and subsequent estimates of survival (S) and detection (p) probabilities.



S_1 = smolt survival from the diversion gate in the bypass at McNary Dam to the tailrace of McNary Dam.

λ = joint probability of survival from the John Day Dam tailrace to Bonneville Dam and being detected at Bonneville Dam.

compare survival rates with ambient river conditions during known days of passage. For purposes of improved precision, release groups at McNary Dam were pooled over five consecutive day periods.

A weighted average of the survival estimates from the replicate releases was calculated according to the formula

$$\hat{\bar{S}} = \frac{\sum_{i=1}^{k} W_i \hat{S}_i}{\sum W_i}$$

where

k = number of replicate releases;

$\hat{S}_i$ = survival estimate from the ith release $(i = 1, \ldots, k)$;

$$W_i = \frac{1}{\left(\frac{\hat{Var}\left(\hat{S}_i\right)}{\hat{S}_i^2}\right)} = \frac{1}{CV\left(\hat{S}_i\right)^2};$$

with variance

$$Var\left(\hat{\bar{S}}\right) = \frac{\sum_{i=1}^{k} W_i \left(\hat{S}_i - \hat{\bar{S}}\right)^2}{(k-1)\sum_{i=1}^{k} W_i}.$$

It was found that by weighting simply inversely proportional to $Var\left(\hat{S}_i\right)$, the weights were correlated with the point estimates, resulting in downward bias in the average survival. By using the relative variance, this correlation between the weights and the point estimate is eliminated. The weights W_i were also used in weighted regression analyses of survival estimates against ambient river conditions.

The $(1-\alpha)$ 100% confidence interval for the weighted average of the survival estimates will be computed by

$$\hat{\bar{S}} \pm Z_{1-\frac{\alpha}{2}} \sqrt{Var\left(\hat{\bar{S}}\right)}.$$

III. Travel Time Analysis

This section summarizes the travel time information at the John Day project during 1998. Because of only one year of information, interyear variability in travel time relationships cannot be characterized.

A. Mean Travel Time

Using the available 1998 travel time data from PIT-tag detections at both McNary and John Day Dams, histograms of travel times were computed by salmonid stock (Figures 3a-d). The harmonic mean travel times through the John Day project ranged from a minimum of 3.06 days (SE = 0.05) for summer chinook salmon smolt to a maximum of 3.74 days (SE = 0.03) for fall chinook salmon smolt (Table 3). Fall chinook traveled significantly slower ($\alpha = 0.05$) than the other three salmonid stocks in 1998.

In 1997, too few fish were detected at both McNary and John Day Dams in most cases to permit reliable estimation of travel times (Table 1). For steelhead in 1997, harmonic mean travel time was estimated to be 1.69 days (SE = 0.10). This estimate of 1997 steelhead travel time is significantly ($\alpha = 0.10$) less than steelhead travel time in 1999 of 3.13 days (SE = 0.04). This difference emphasizes the importance of interannual variation which cannot be estimated for travel time calculations for chinook. Interpretations of travel time results in Table 3 should therefore be treated cautiously until further data become available.

B. Travel Time Relationships

Harmonic mean travel times were computed on a daily basis from PIT-tagged salmonids released daily through the bypass at McNary Dam (Figure 4). These travel times through the John Day project were then correlated with ambient river conditions to determine if possible relationships may exist that might influence travel times (Table 4).

Figure 3. Histograms of travel time frequencies at the John Day Dam project during 1998 for (a) spring chinook, (b) summer chinook, (c) fall chinook, and (d) steelhead.

(a) Spring chinook salmon

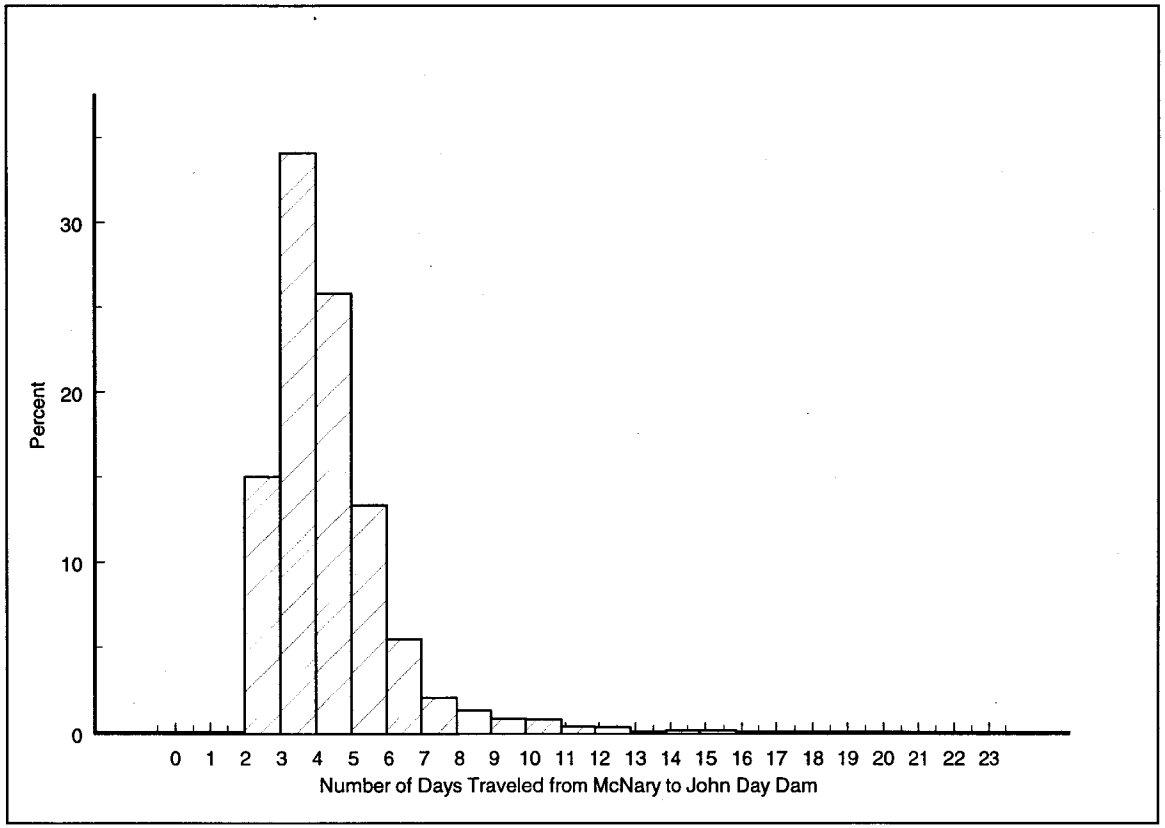


Figure 3. (Continued)

(b) Summer chinook salmon

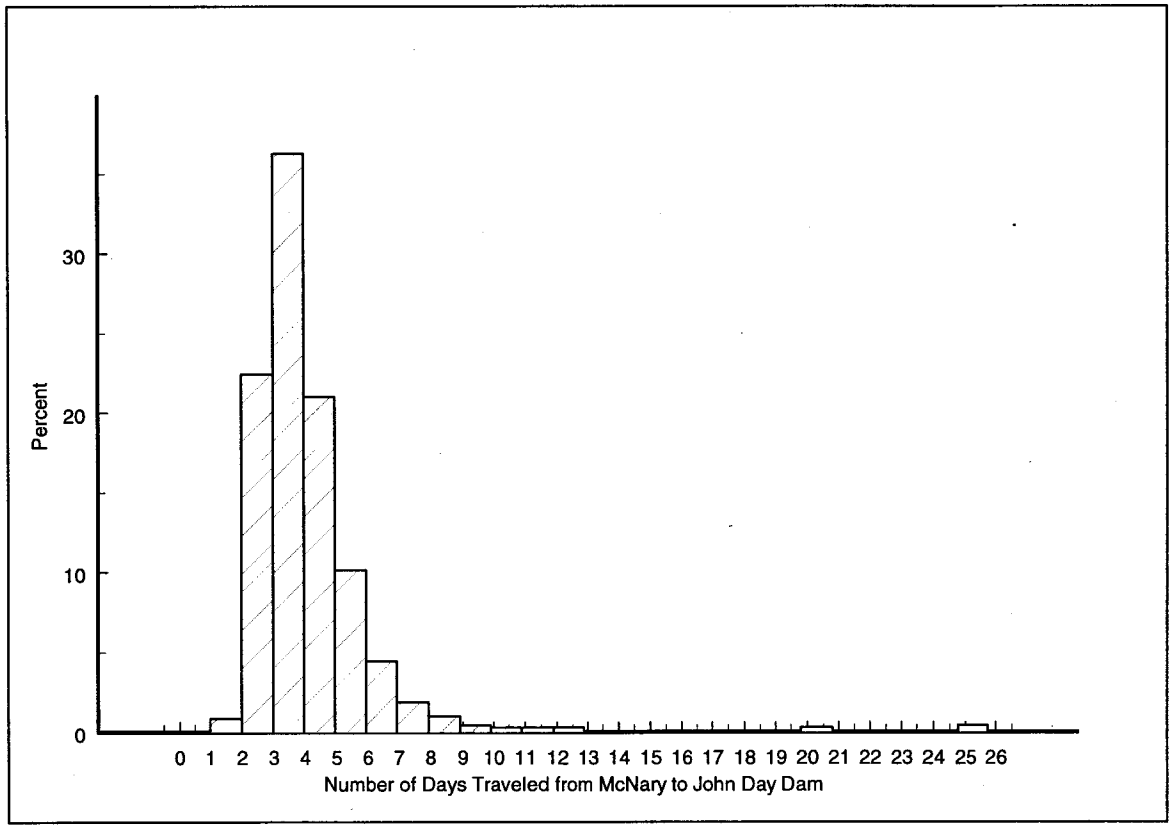


Figure 3. (Continued)

(c) Fall chinook salmon

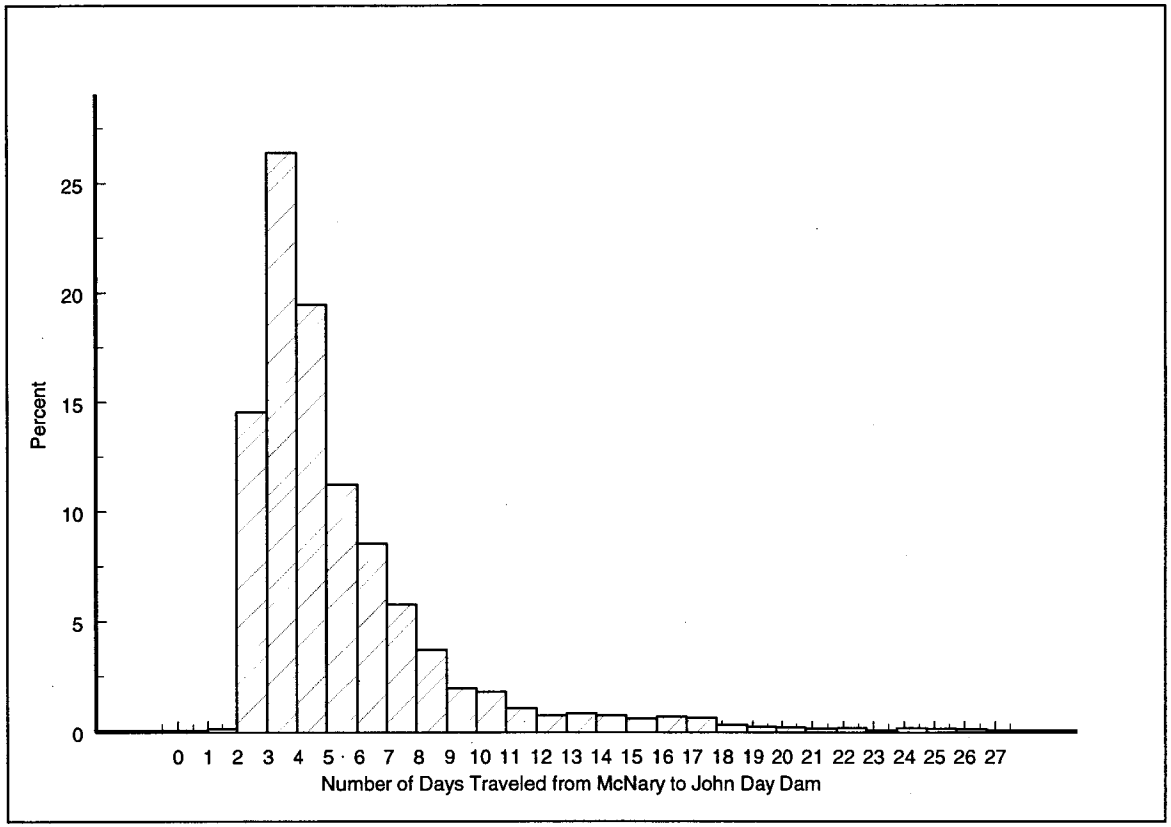


Figure 3. (Continued)

(d) Steelhead

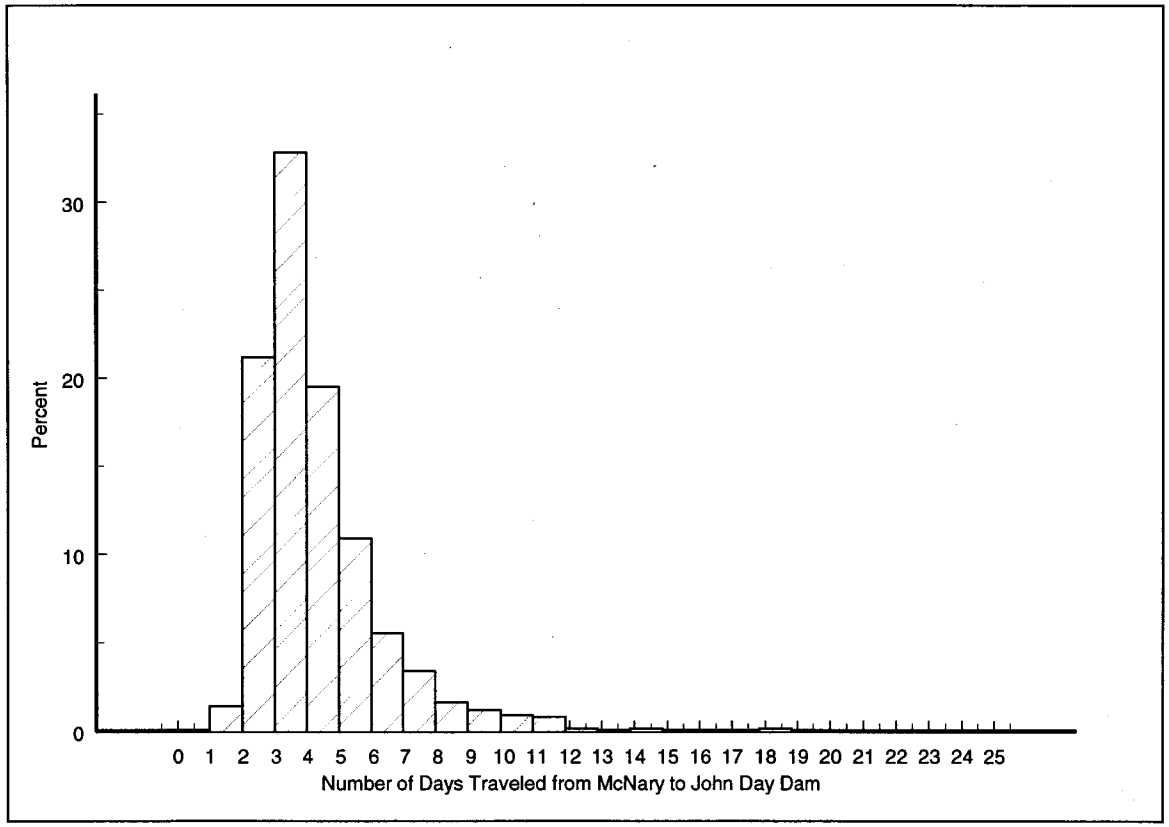


Table 3. Summary of harmonic $(\bar{t}_H)$ and arithmetic $(\bar{t})$ mean travel times for salmonid stocks through the John Day project in 1998.

Year	Stock	Harmonic			Arithmetic		
		Mean	SE	95% CI	Mean	SE	95% CI
1997	Steelhead	1.69	0.10	1.49-1.89	2.25	0.25	1.76-2.74
1998	Chinook Salmon						
	Spring	3.37	0.03	3.31-3.43	3.92	0.04	3.84-4.00
	Summer	3.06	0.05	2.96-3.16	3.67	0.09	3.49-3.85
	Fall	3.74	0.03	3.68-3.80	5.10	0.07	4.96-5.24
	Steelhead	3.13	0.04	3.06-3.21	3.86	0.06	3.75-3.97

Figure 4. Harmonic mean travel times for PIT-tagged salmonids released through the McNary Dam bypass on a daily basis in 1998 by stock. Vertical bars represent 95% confidence interval for the mean daily travel times.

(a) Spring chinook salmon

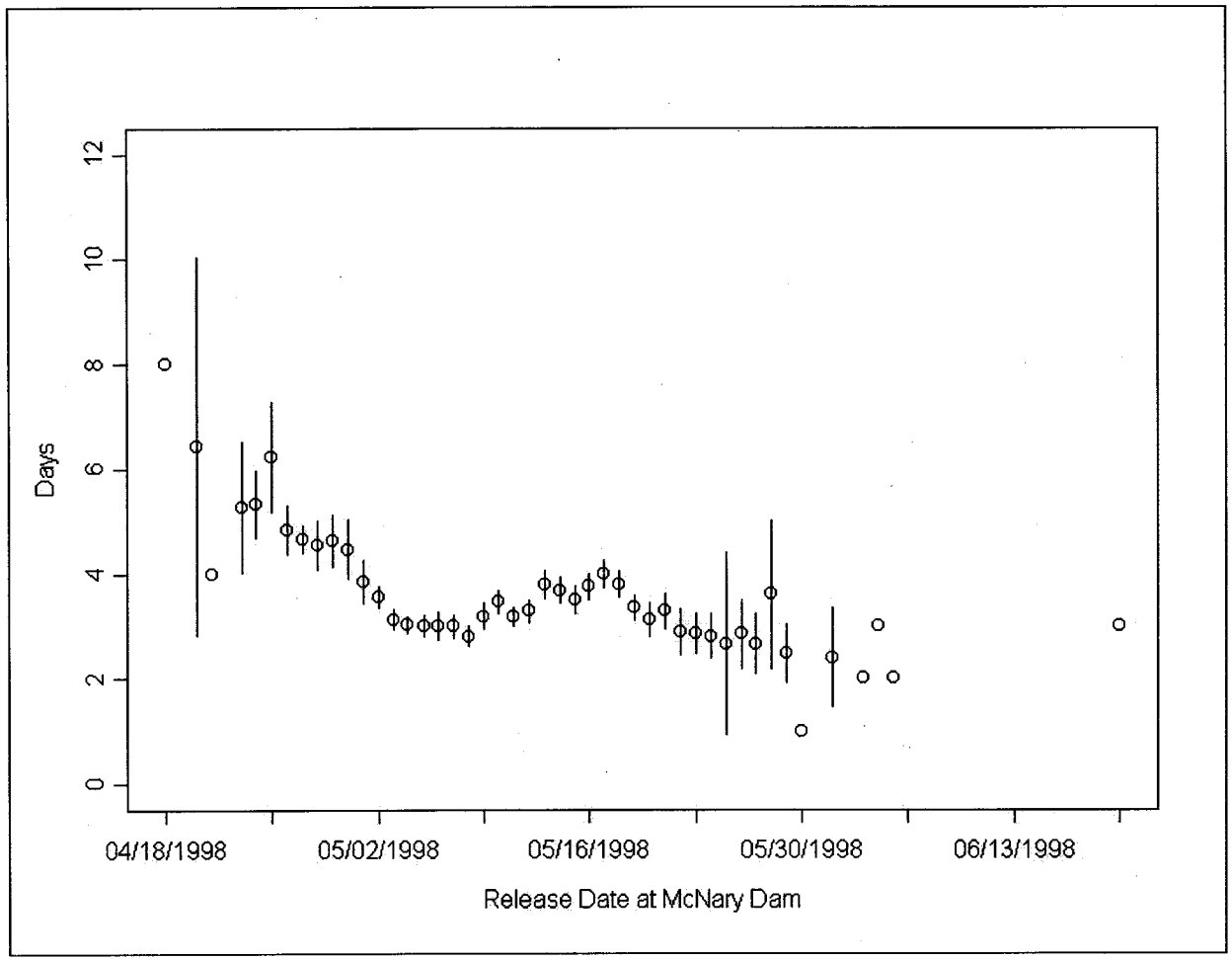


Figure 4. (Continued)

(b) Summer chinook salmon

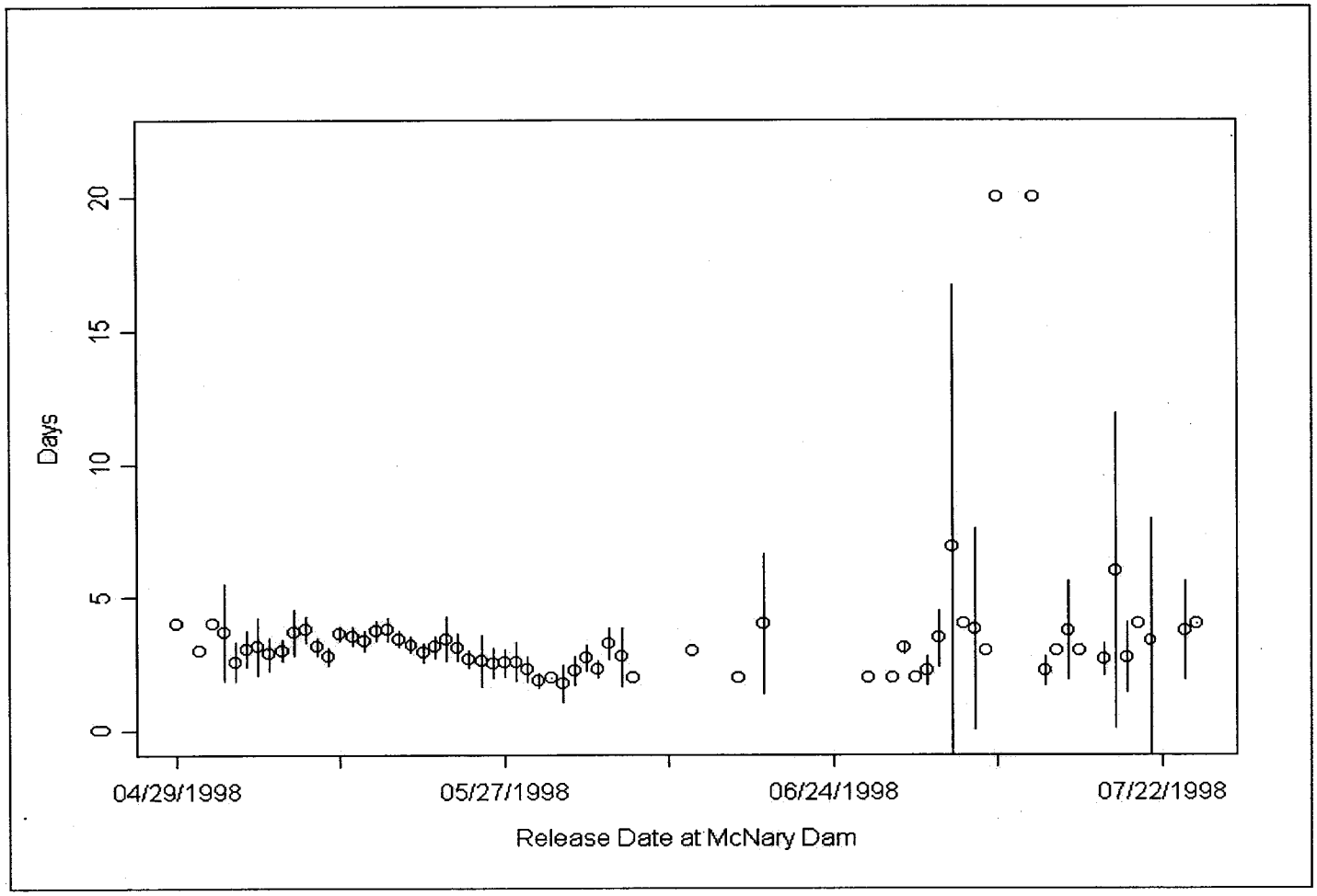


Figure 4. (Continued)

(c) Fall chinook salmon

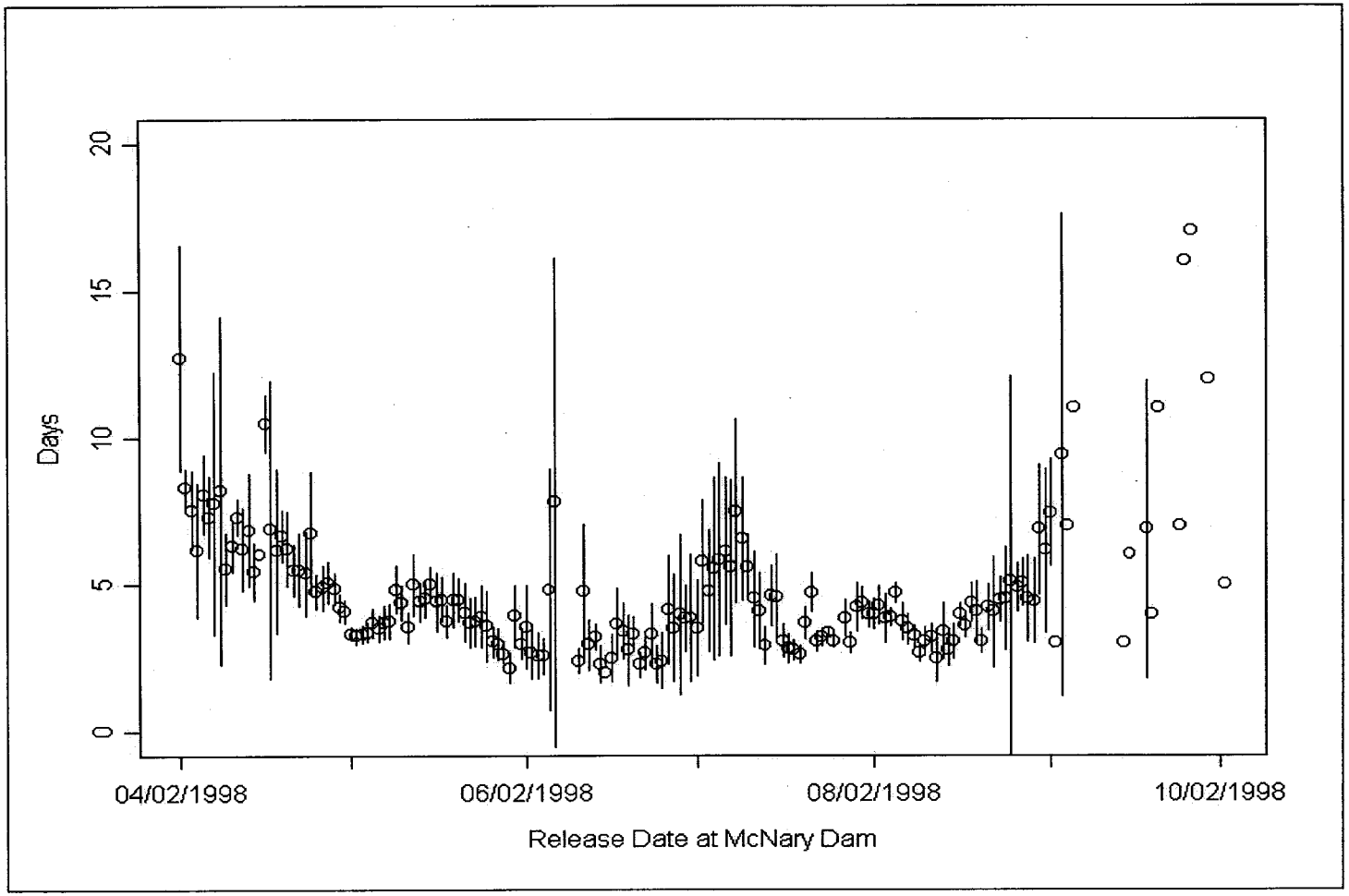


Figure 4. (Continued)

(d) Steelhead

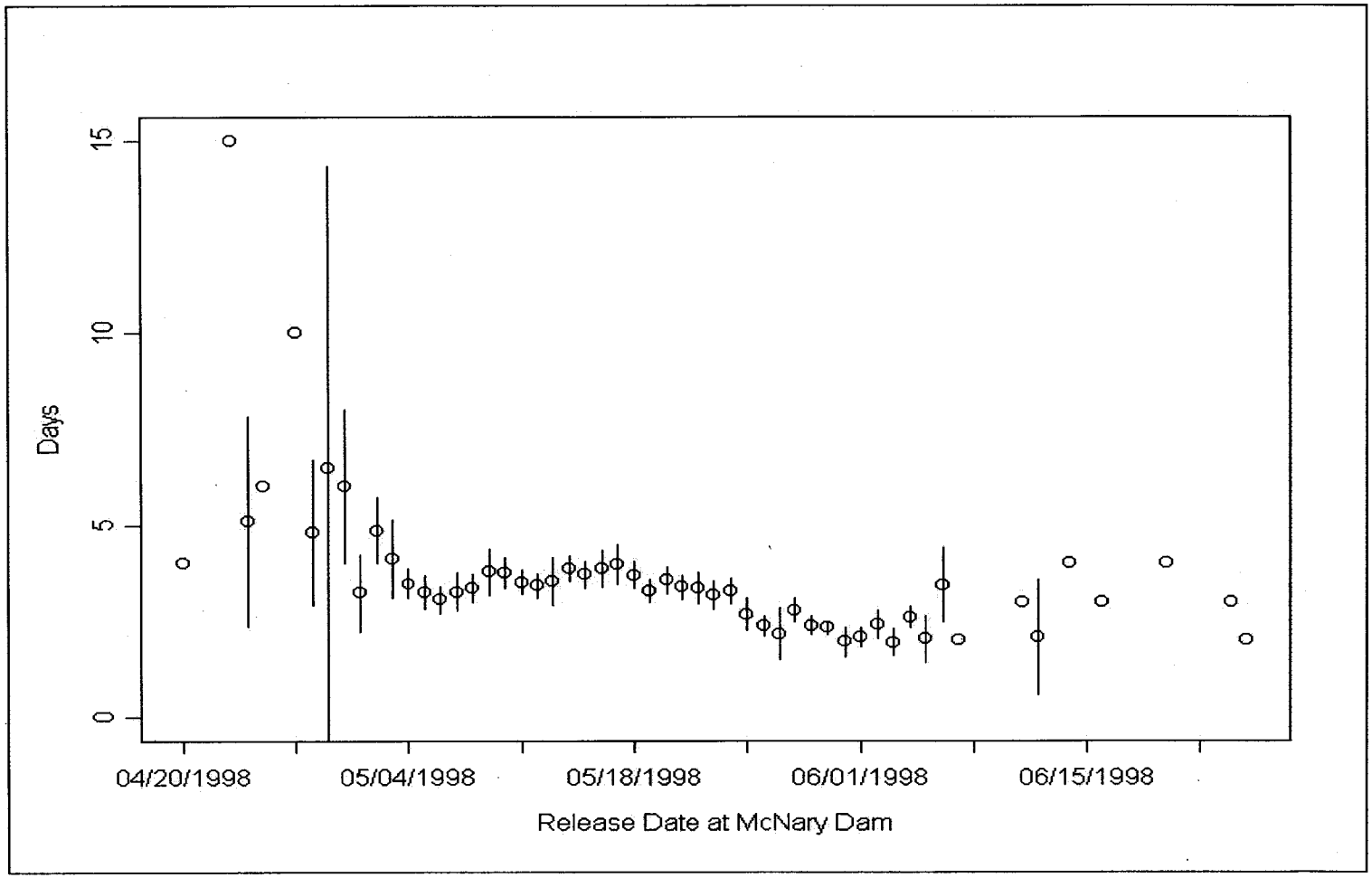


Table 4. Correlation coefficients between daily harmonic mean travel times and ambient river conditions during 1998.

Stock	*n*	McNary Dam		John Day Dam		Temperature	Turbidity
		Flow Volume	Spill Volume	Flow Volume	Spill Volume		
Chinook salmon							
Spring	39	-0.8144*	-0.7475*	-0.7643*	-0.6682*	-0.8590*	-0.3070*
Summer	49	-0.4385*	-0.4657*	-0.4605*	-0.4793*	0.3130*	0.2911*
Fall	133	-0.3811*	-0.3019*	-0.3681*	-0.3139*	-0.2113*	0.1668*
Steelhead	42	-0.7761*	-0.7601*	-0.7917*	-0.7180*	-0.5539*	0.1683

* Denotes significance at $\alpha = 0.10$.

Ambient conditions used in the correlation analysis were the river flow volumes and spill volumes from McNary and John Day Dams, daily mean water temperature, and turbidity on the day of release at McNary Dam. Table 5 summarizes the correlation among the independent environmental variables.

Increased river flow and increased spill volume were both found to be negatively correlated with travel times of chinook salmon and steelhead smolt during 1998 (Table 4). In other words, as flow and spill volumes increased, harmonic mean travel time decreased on a daily basis during 1998. Temperature was also found to be correlated with travel time (Table 4). In three of four cases, travel times decreased as temperatures increased. Turbidity showed less of a consistent relationship with travel times (Table 4). In interpreting these results, it is important to note that turbidity, temperature, flow, and spill volumes tend to have strong seasonal trends that may induce spuriously correlations when compared with travel time trends over the season. Interannual comparisons may better characterize important relationships with travel time. However, interannual comparisons are not available at this time for proper analysis of travel time relationships.

IV. Survival Analysis

This section summarizes the estimates of smolt survival for the John Day project (i.e., pool and dam) from PIT-tag releases by group upstream and by date of release from McNary bypass. Upstream release groups provide natural groupings for estimating reach survival. However, these release groups migrate over the John Day reach as a wavefront, making inferences to specific ambient conditions difficult over the protracted period of outmigration. To relate survival estimates to specific ambient conditions, smolt released on the same day or groups of consecutive days commencing from McNary Dam provides a better tag-release for investigating survival relationships.

Table 5. Correlation coefficients between ambient variables used in travel time and survival relationships for the period 2 April – 5 September 1998.

	McNary Flow	McNary Spill	John Day Flow	John Day Spill	Temperature	Turbidity
McNary Flow	1					
McNary Spill	0.9775	1				
John Day Flow	0.9805	0.9623	1			
John Day Spill	0.9148	0.9214	0.9259	1		
Temperature	-0.3457	-0.3964	-0.3845	-0.2075	1	
Turbidity	-0.4206	-0.4149	-0.4099	0.3424	0.0774	1

A. Survival Estimates

PIT-tag release groups from upstream release locations were used to estimate smolt survival through the John Day project. Survival was estimated for each tag-release group (Appendix A) and summarized across releases (Table 6) by stock. Survival for the spring, summer, and fall chinook salmon ranged from 0.826 – 0.890 and with no significant difference between stocks detected ($\alpha = 0.05$). Steelhead had a mean survival of 1.043 (SE = 0.038) through the John Day project and was significantly higher ($\alpha = 0.10$) than for spring and fall chinook salmon (Table 6).

The mean survival estimates may be inflated because of out-of-range estimates (i.e., $S > 0.10$) obtained for some release groups. Although slight overestimates may be expected from some release groups with small release size and true survival rates near 1, excessively high point estimates are problematic. Weighting the estimates by precision overcomes some but not all of the problem. Hence, the mean values reported in Table 6 may be positively biased to a small but unknown degree.

B. Survival Trends

Survival trends during the 1998 outmigration season were generated for the four salmonid stocks. Smolt released from the McNary Dam bypass on consecutive days for a five-day period were pooled to estimate survival in the John Day project. This five-day period window of time was incremented daily to estimate smolt survival using a five-day moving average. The moving averages of smolt survival by stock in 1998 at the John Day project are plotted over time in Figures 5a-d. For purposes of graphical comparison, survival trends are plotted along with seasonal trends in flow and spill volume, temperature, and turbidity. However, the correlations among moving averages precludes formal regression analysis against ambient factors.

Table 6. Average smolt survival $\left(\hat{\bar{S}}\right)$, number of release groups (n), and standard errors (SE) and 95% confidence intervals about mean survival by stock during 1998 in John Day project.

Stock	n	$\hat{\bar{S}}$	$\hat{S}E$	95% CI
Chinook salmon				
Spring	39	0.826	0.030	0.767-0.885
Summer	13	0.890	0.075	0.743-1.000
Fall	56	0.832	0.033	0.767-0.897
Steelhead	53	1.043	0.038	0.968-1.000

Figure 5. Estimates of smolt survival using a five-day moving average are plotted over the course of the 1998 outmigration season at the John Day project by salmonid stock and plotted against ambient river conditions.

(a) Spring chinook salmon

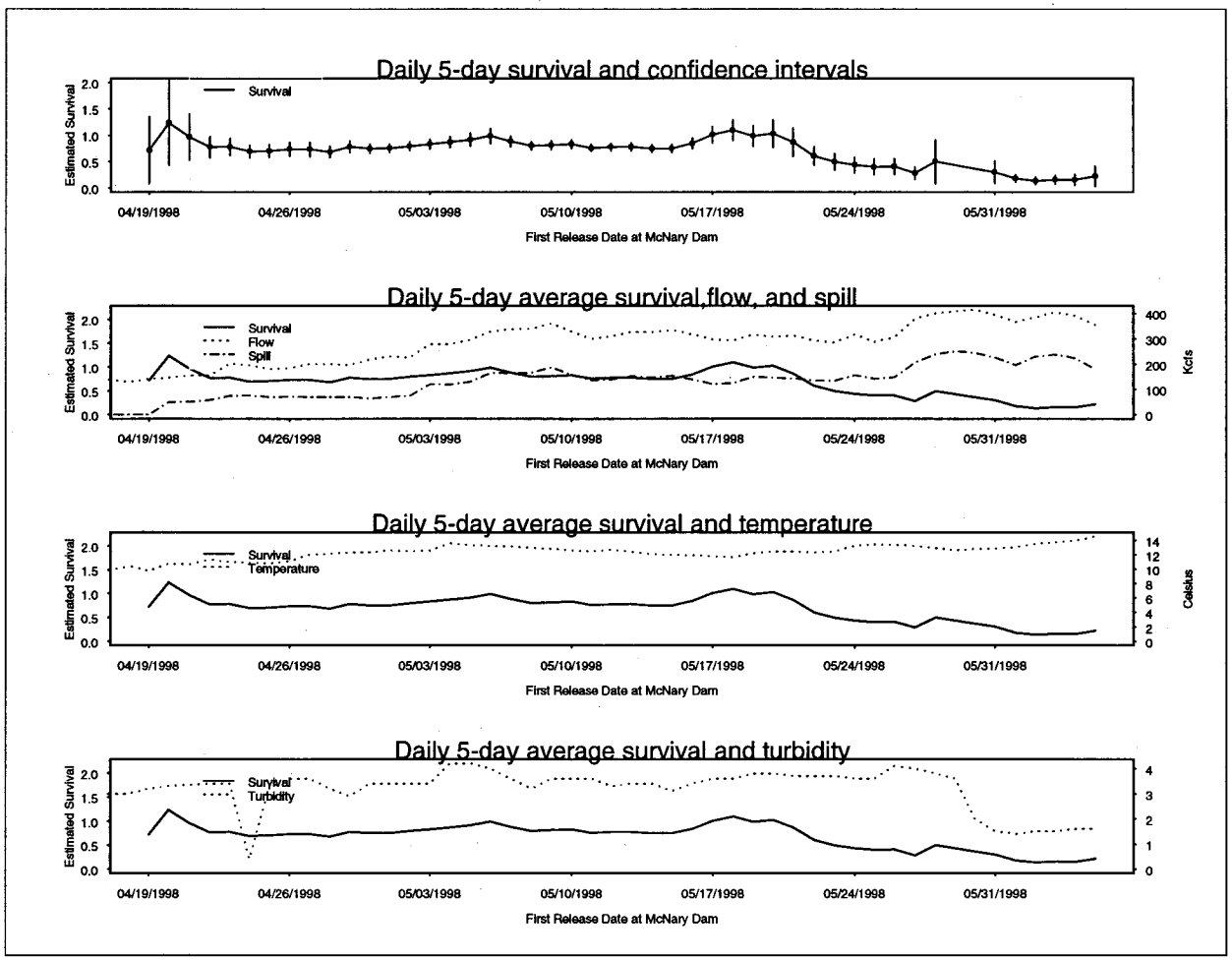


Figure 5. (Continued)

(b) Summer chinook salmon

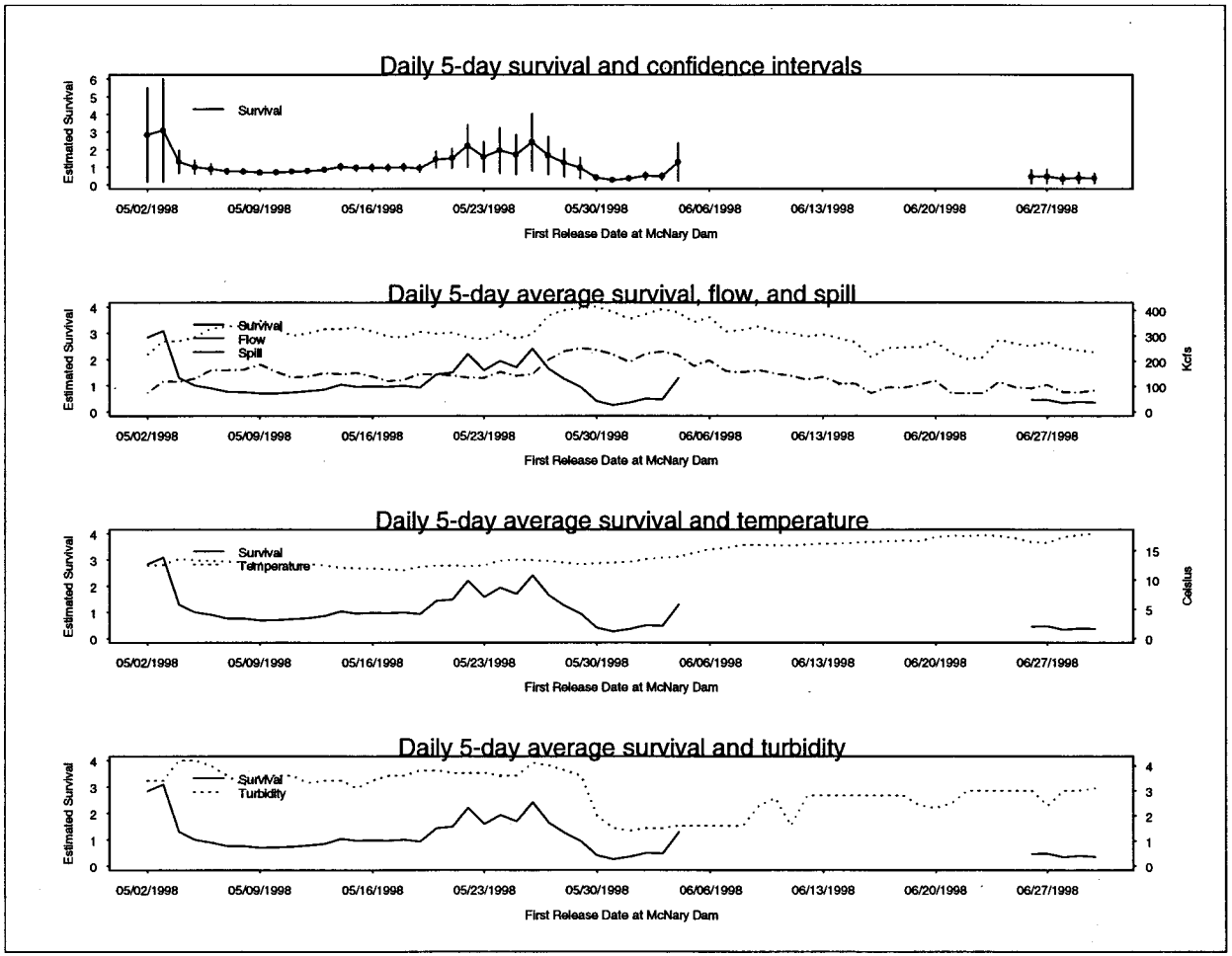


Figure 5. (Continued)

(c) Fall chinook salmon

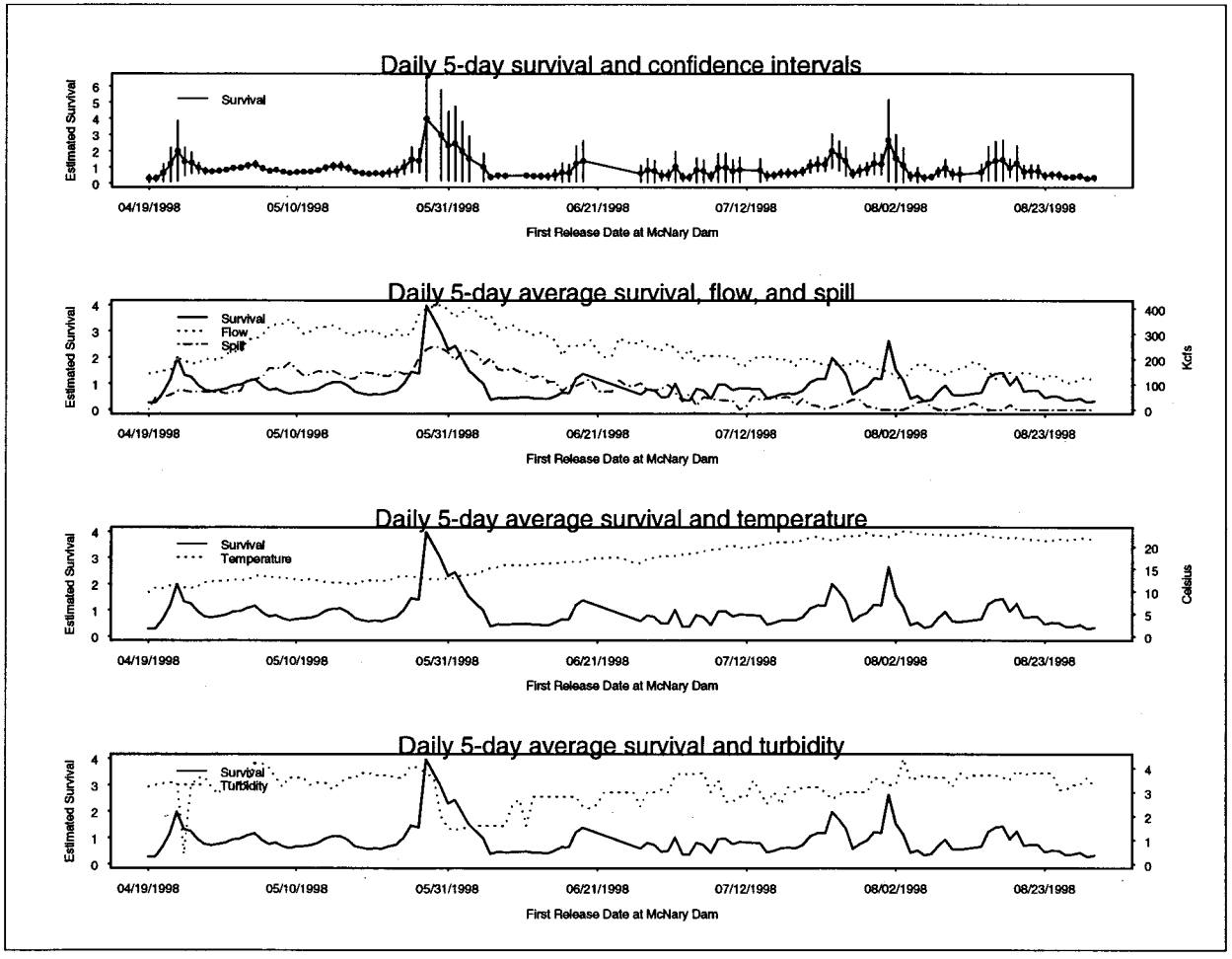
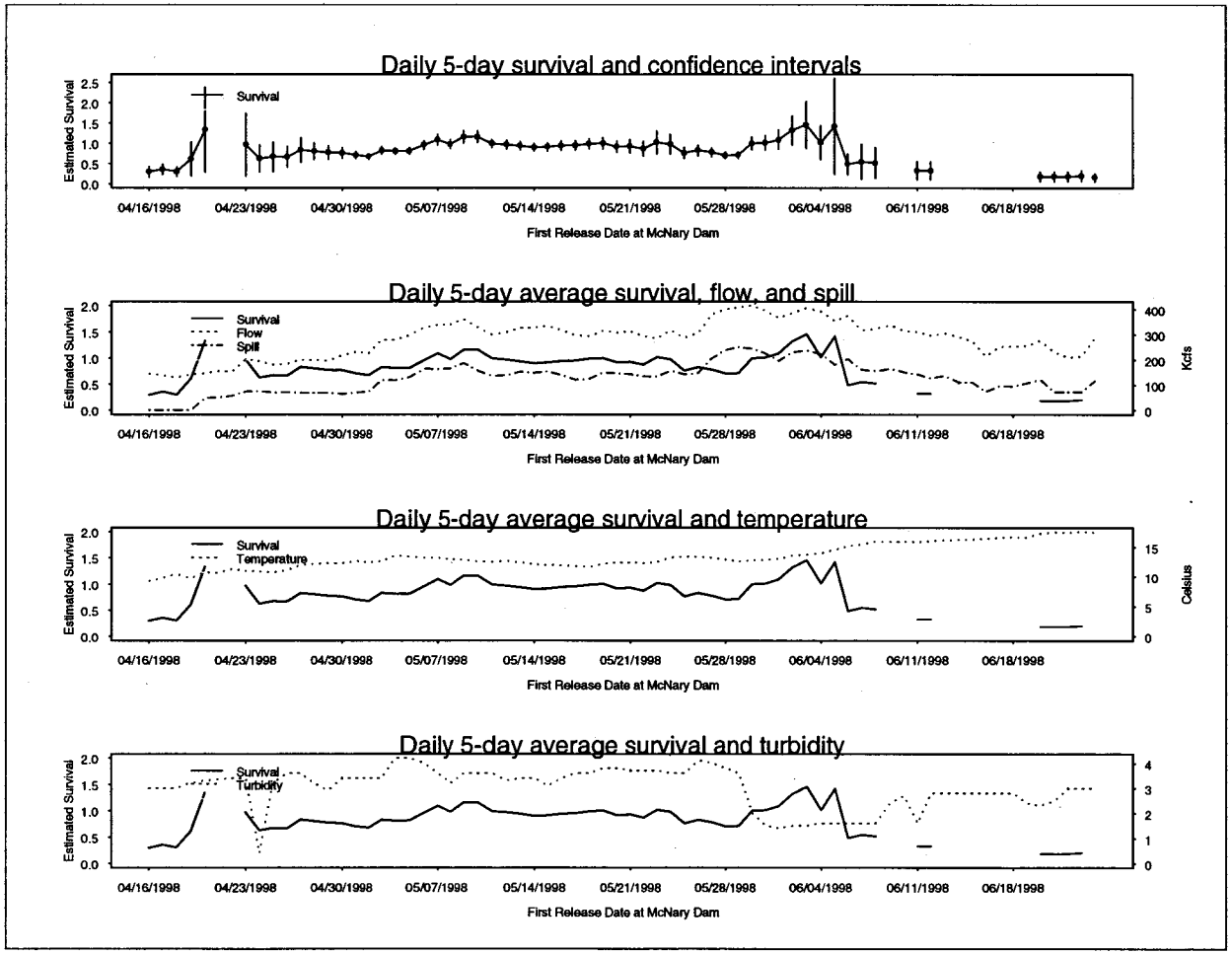


Figure 5. (Continued)

(d) Steelhead



C. Survival Relationships

The relationship between smolt survival and ambient river conditions was investigated using the available 1998 data. Five days of consecutive releases from McNary Dam bypass were pooled to estimate survival through the John Day project. These five-day pooled estimates were then regressed against the average ambient river conditions for the same period of time using weighted regressions. The survival estimates were weighted by their relative precision in the regression analyses. Table 7 summarizes the correlation coefficients between smolt survival and flow volume and spill volume at McNary and John Day Dams, mean water temperature, and turbidity.

Simple linear regression found no correlation between 5-day average survival probabilities and river operations or ambient conditions in 23 of 24 analyses (Table 7). Only spring chinook salmon smolt survival was found to be positively correlated with turbidity ($r = 0.2007$) at $\alpha = 0.10$. Appendix B presents x-y scatterplots of survival versus river conditions by stock. Again, the reader is reminded that these correlations represent only intraannual relationships found during 1998 and may not be representative of other years or between-year relationships.

V. Discussion

This report summarizes available information on salmonid smolt travel times and survival through the John Day project. This information was largely limited to PIT-tag detections in 1998. As such, only within-season trends for 1998 could be characterized in most cases. In the single case where travel times were available for steelhead in both 1997 and 1998, the between-year variability was large (1.69 versus 3.13 days, respectively, was significant at $P < 0.001$). This evidence suggests between-year variability is a significant component of overall variation in response that should not be ignored.

Table 7. Correlation coefficients between five-day pooled estimates and ambient river conditions during 1998.

Stock	McNary Dam				John Day Dam	
	Flow	Spill	Temperature	Turbidity	Flow	Spill
Chinook						
Spring	0.0003	0.0004	0.0351	0.2007*	0.0003	0.0004
Summer	-0.0055	-0.0036	-0.0826	0.3296	-0.0049	-0.0052
Fall	0.0008	0.0008	-0.0081	-0.0625	0.0009	0.0025
Steelhead	0.0002	-0.0001	-0.0123	0.0347	0.0001	-0.0006

* Denotes significance at $\alpha = 0.10$.

Table 8. Summary of mean* survival estimates reported by NMFS for Little Goose and Lower Monumental project reaches 1994-97 (standard error in parentheses).

Year	Stock	Little Goose	Lower Monumental	Reference
1994	Yearling chinook	0.794 (0.026)	0.891 (0.023)	Muir et al. 1985
	Yearling chinook	0.882 (0.010)	0.863 (0.022)	Muir et al. 1985
1995	Hatchery yearling chinook	0.883 (0.006)	0.928 (0.007)	Muir et al. 1996
	Wild yearling chinook	0.887 (0.012)	0.896 (0.017)	Muir et al. 1996
1996	Yearling chinook	0.887 (0.030)	0.879 (0.041)	Smith et al. 1998
	Steelhead	0.912 (0.024)	0.982 (0.030)	Smith et al. 1998
1997	Yearling chinook	0.954 (0.015)	0.825 (0.023)	Hockersmith et al. 1998
	Yearling chinook	0.969 (0.007)	0.897 (0.020)	Hockersmith et al. 1998
	Steelhead	0.949 (0.021)	0.879 (0.032)	Hockersmith et al. 1998

* Weighted mean based on weighting independent survival estimates inversely proportional to sample variance. Revised weighting scheme now used by NMFS of weighting inversely proportional to CV^2 would result in higher mean survival estimates than reported here in this table.

Travel speeds through the John Day project (i.e., McNary Dam PIT-tag detector to John Day Dam PIT-tag detector) were quite fast, ranging from 72.8 km/day for steelhead in 1997 to 32 km/day for fall chinook salmon in 1998. Steelhead mean travel time was significantly shorter in the John Day project than chinook stocks.

Mean yearling chinook salmon survival [i.e., 0.826 (SE = 0.030)] through the John Day project in 1998 was generally less than mean survival rates reported by the National Marine Fisheries Service (NMFS) for yearling chinook salmon at upper Snake River hydroprojects during 1994-97 (Table 8). Mean steelhead survival [1.043 (SE = 0.038)], however, was higher than mean values reported by NMFS for Snake River projects (Table 8).

The survival rates for fall subyearling chinook salmon at the John Day project in 1998 averaged 0.832 (0.0033). However, these hatchery fall chinook were predominately spring releases from the Mid-Columbia and may not be representative of wild stocks (i.e., larger, more mature smolt than wild subyearlings). Across the board, insufficient data currently exist to estimate survival rates for wild stocks of salmonids in the John Day project.

A careful assessment of smolt outmigration dynamics at John Day project will require additional years of tagging data before mean responses can be characterized and relationships to ambient river conditions can be thoroughly explored. However, based on currently available information from the 1998 PIT-tag database, smolt survival through the John Day project ranged from 83 to 100 percent depending upon salmonid stock.

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Attachment E - Appendix A

Smolt survival estimates by PIT-tag release group in 1998 for the John Day project.

Table A1. Summary of estimates of John Day project survival by release groups in 1998 for spring yearling chinook salmon smolt.

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
HLB98040.R05	Clearwater River NF	437	189	158	0.673 (0.137)
HLB98042.R07	Clearwater River NF	485	166	172	0.948 (0.212)
HLB98043.R08	Clearwater River NF	459	164	184	0.747 (0.155)
HLB98049.R09	Clearwater River NF	456	150	173	0.913 (0.196)
HLB98050.R10	Clearwater River NF	464	124	164	1.499 (0.480)
HLB98054.R11	Clearwater River NF	494	119	196	0.751 (0.155)
HLB98056.R12	Clearwater River NF	562	147	174	1.120 (0.283)
HLB98057.R16	Clearwater River NF	609	133	208	0.584 (0.104)
HLB98061.R17	Clearwater River NF	666	132	178	0.859 (0.207)
HLB98063.R18	Clearwater River NF	686	141	170	0.491 (0.085)
HLB98064.R19	Clearwater River NF	657	147	170	1.122 (0.308)
HLB98068.C20	Clearwater River NF	478	105	124	0.673 (0.146)*
HLB98068.R20	Clearwater River NF	99	21	31	
KEP98062.CR1	Crooked River P	57	38	29	0.535 (0.164)
KEP98062.P3B	Powell Rearing P	39	27	20	0.558 (0.350)
KEP98062.PW1	Powell Rearing P	39	21	24	1.225 (0.854)
KEP98062.RRL	Red River Rearing P	57	34	21	No estimate**
KEP98063.RRH	Red River Rearing P	43	38	30	0.752 (0.253)
LRB98012.WI1	Winthrop Hatchery	148	71	79	0.675 (0.252)
LRB98013.WI1	Winthrop Hatchery	127	62	91	0.626 (0.194)
LRB98013.WI2	Winthrop Hatchery	69	47	45	
LRB98014.WI1	Winthrop Hatchery	51	48	48	1.120 (0.388)
LRB98014.WI2	Winthrop Hatchery	175	77	89	
LRB98027.LV1	Leavenworth Hatchery	67	80	30	1.116 (0.216)
LRB98027.LV2	Leavenworth Hatchery	95	78	43	
LRB98027.LV3	Leavenworth Hatchery	126	138	63	
LRB98028.LV1	Leavenworth Hatchery	172	173	74	1.223 (0.240)
LRB98028.LV2	Leavenworth Hatchery	162	102	59	
LRB98029.LV1	Leavenworth Hatchery	151	134	57	0.810 (0.150)
LRB98029.LV2	Leavenworth Hatchery	118	125	56	

*Bracket indicates pooling of capture data used in estimating survival.
**No estimate was given by the CJS model for this reach.

Table A1. (Continued)

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
LRB98040.R11	Rapid River Hatchery	78	31	26	
LRB98040.R12	Rapid River Hatchery	86	41	30	
LRB98040.R13	Rapid River Hatchery	60	40	32	
LRB98040.R21	Rapid River Hatchery	69	35	25	
LRB98040.R22	Rapid River Hatchery	68	43	47	
LRB98040.R23	Rapid River Hatchery	88	47	50	
LRB98040.R32	Rapid River Hatchery	72	45	24	0.776 (0.091)
LRB98040.R33	Rapid River Hatchery	89	48	43	
LRB98040.R3B	Rapid River Hatchery	84	42	32	
LRB98040.R41	Rapid River Hatchery	60	42	32	
LRB98040.R42	Rapid River Hatchery	82	34	34	
LRB98040.R43	Rapid River Hatchery	95	42	46	
LRB98041.R11	Rapid River Hatchery	65	38	38	
LRB98041.R12	Rapid River Hatchery	88	44	32	
LRB98041.R13	Rapid River Hatchery	64	47	41	
LRB98041.R14	Rapid River Hatchery	77	46	40	
LRB98041.R22	Rapid River Hatchery	83	44	37	
LRB98041.R23	Rapid River Hatchery	69	39	43	
LRB98041.R24	Rapid River Hatchery	35	25	24	
LRB98041.R2A	Rapid River Hatchery	84	60	28	
LRB98041.R31	Rapid River Hatchery	73	36	40	0.745 (0.072)
LRB98041.R32	Rapid River Hatchery	69	44	40	
LRB98041.R34	Rapid River Hatchery	76	61	38	
LRB98041.R35	Rapid River Hatchery	102	38	38	
LRB98041.R41	Rapid River Hatchery	73	50	37	
LRB98041.R42	Rapid River Hatchery	73	47	33	
LRB98041.R44	Rapid River Hatchery	81	36	39	
LRB98041.R45	Rapid River Hatchery	69	47	42	

Table A1. (Continued)

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
LRB98040.R11	Rapid River Hatchery	78	31	26	
LRB98042.R12	Rapid River Hatchery	74	40	37	
LRB98042.R13	Rapid River Hatchery	85	37	37	
LRB98042.R14	Rapid River Hatchery	75	41	39	
LRB98042.R15	Rapid River Hatchery	75	36	31	
LRB98042.R21	Rapid River Hatchery	70	42	36	
LRB98042.R22	Rapid River Hatchery	62	46	23	
LRB98042.R23	Rapid River Hatchery	75	49	38	
LRB98042.R24	Rapid River Hatchery	74	37	35	
LRB98042.R31	Rapid River Hatchery	66	47	48	0.959 (0.103)
LRB98042.R32	Rapid River Hatchery	83	60	33	
LRB98042.R33	Rapid River Hatchery	76	43	35	
LRB98042.R34	Rapid River Hatchery	67	43	33	
LRB98042.R35	Rapid River Hatchery	93	34	37	
LRB98042.R41	Rapid River Hatchery	78	48	33	
LRB98042.R42	Rapid River Hatchery	69	36	32	
LRB98042.R43	Rapid River Hatchery	80	35	35	
LRB98042.R44	Rapid River Hatchery	72	38	46	
LRB98042.R45	Rapid River Hatchery	66	31	25	
PMS98048.01A	Imnaha River W	65	46	29	
PMS98048.01B	Imnaha River W	68	45	38	0.886 (0.153)
PMS98048.02A	Imnaha River W	160	71	84	
PMS98048.02B	Imnaha River W	101	45	60	
PMS98049.05A	Imnaha River W	186	111	94	0.698 (0.106)
PMS98049.05D	Imnaha River W	163	110	80	
PMS98050.04A	Imnaha River W	94	70	47	
PMS98050.04D	Imnaha River W	77	40	37	0.887 (0.150)
PMS98050.06A	Imnaha River W	77	48	49	
PMS98050.06B	Imnaha River W	73	48	47	
PMS98051.07C	Imnaha River W	256	128	100	0.933 (0.191)
PMS98051.07D	Imnaha River W	162	94	40	
PMS98054.03A	Imnaha River W	167	103	83	0.933 (0.170)
PMS98054.03D	Imnaha River W	91	62	54	
PMS98055.15B	Looking Glass Hatchery	62	37	30	0.721 (0.205)
PMS98055.15C	Looking Glass Hatchery	85	38	34	
PMS98056.15A	Looking Glass Hatchery	219	91	84	0.692 (0.129)
PMS98056.15B	Looking Glass Hatchery	220	120	70	
PMS98057.15A	Looking Glass Hatchery	216	104	87	0.722 (0.134)
PMS98057.15B	Looking Glass Hatchery	214	124	84	
PMS98058.15A	Looking Glass Hatchery	209	89	83	0.840 (0.184)
PMS98058.15D	Looking Glass Hatchery	185	95	69	
PMS98061.15B	Looking Glass Hatchery	247	121	95	1.157 (0.232)
PMS98061.15C	Looking Glass Hatchery	240	118	93	

Table A1. (Continued)

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
PMS98062.15A	Looking Glass Hatchery	190	81	66	0.638 (0.126)
PMS98062.15D	Looking Glass Hatchery	194	130	89	
PMS98063.15A	Looking Glass Hatchery	142	72	48	0.875 (0.224)
PMS98063.15B	Looking Glass Hatchery	122	59	31	
PMS98064.16C	Looking Glass Hatchery	208	136	100	0.782 (0.151)
PMS98065.16A	Looking Glass Hatchery	141	81	56	0.436 (0.122)
				Weighted average of available survival ests.	**0.826 (0.030)**

Table A2. Summary of estimates of John Day project survival by release group in 1998 for summer chinook salmon smolt.

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
CSM98105.WW*	Columbia River	169	133	106	1.103 (0.339)
CSM98107.WW*	Columbia River	194	138	89	0.880 (0.304)
CSM98109.WW*	Columbia River	156	154	92	2.146 (0.795)
CSM98111.WW*	Columbia River	125	140	73	1.534 (0.637)
CSM98113.WW*	Columbia River	126	160	70	0.938 (0.330)
CSM98105.WP*	Columbia River	135	135	78	1.438 (0.547)
CSM98107.WP*	Columbia River	124	143	77	0.819 (0.266)
CSM98109.WP*	Columbia River	96	119	66	0.718 (0.217)
CSM98111.WP*	Columbia River	114	130	75	0.659 (0.237)
CSM98113.WP*	Columbia River	134	127	71	1.673 (1.009)
LRB98048.M11	Knox Bridge	55	40	40	
LRB98048.M12	Knox Bridge	70	31	26	
LRB98048.M13	Knox Bridge	61	40	33	
LRB98048.M14	Knox Bridge	66	29	22	
LRB98048.M21	Knox Bridge	71	51	47	
LRB98048.M22	Knox Bridge	60	34	32	
LRB98048.M23	Knox Bridge	43	38	33	0.829 (0.094)
LRB98048.M31	Knox Bridge	53	47	32	
LRB98048.M32	Knox Bridge	55	41	33	
LRB98048.M33	Knox Bridge	51	36	31	
LRB98048.M41	Knox Bridge	74	49	44	
LRB98048.M42	Knox Bridge	76	52	49	
LRB98048.M43	Knox Bridge	59	48	40	
LRB98048.M44	Knox Bridge	56	50	22	

Table A2. (Continued)

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
LRB98049.M11	Knox Bridge	56	37	30	
LRB98049.M12	Knox Bridge	73	36	40	
LRB98049.M13	Knox Bridge	59	45	22	
LRB98049.M14	Knox Bridge	50	38	40	
LRB98049.M21	Knox Bridge	67	45	41	
LRB98049.M22	Knox Bridge	71	51	34	
LRB98049.M23	Knox Bridge	58	47	32	
LRB98049.M31	Knox Bridge	60	32	25	
LRB98049.M32	Knox Bridge	51	48	33	0.946 (0.104)
LRB98049.M33	Knox Bridge	48	44	41	
LRB98049.M34	Knox Bridge	59	50	32	
LRB98049.M41	Knox Bridge	59	33	34	
LRB98049.M42	Knox Bridge	68	48	33	
LRB98049.M43	Knox Bridge	51	32	37	
LRB98049.M44	Knox Bridge	70	47	34	
LRB98049.M45	Knox Bridge	60	42	26	
LRB98050.M11	Knox Bridge	70	56	37	
LRB98050.M12	Knox Bridge	75	46	33	
LRB98050.M13	Knox Bridge	59	39	31	
LRB98050.M14	Knox Bridge	49	35	32	
LRB98050.M21	Knox Bridge	49	29	30	
LRB98050.M22	Knox Bridge	69	36	35	
LRB98050.M23	Knox Bridge	70	47	34	
LRB98050.M24	Knox Bridge	64	51	31	0.711 (0.073)
LRB98050.M25	Knox Bridge	44	29	28	
LRB98050.M32	Knox Bridge	63	55	30	
LRB98050.M33	Knox Bridge	66	41	32	
LRB98050.M3B	Knox Bridge	71	35	33	
LRB98050.M42	Knox Bridge	60	45	27	
LRB98050.M43	Knox Bridge	63	35	27	
LRB98050.M44	Knox Bridge	62	39	37	
LRB98050.M45	Knox Bridge	49	37	28	
		Weighted average of available survival ests.			**0.890 (0.075)**

Table A3. Summary of estimates of John Day project survival by release groups in 1998 for fall subyearling chinook salmon smolt.

File ID	Release Site	Total smolt observed at:			John Day Survival
		McNary	John Day	Bonneville or below	
BDA98097.BC1	Clearwater River	225	149	80	1.058 (0.297)
BDA98098.BC4	Clearwater River	242	136	92	1.249 (0.389)
LRB98147.PR1	Priest Rapids Hatchery	261	55	22	0.463 (0.254)
MBE98104.RIT	Rock Island Dam	232	214	89	0.685 (0.152)
MBE98104.RRT	Rock Reach Dam	195	177	101	1.129 (0.267)
MBE98104.WEL	Wells Hatchery	139	157	88	1.152 (0.269)
MBE98105.RIT	Rock Island Dam	214	199	112	0.770 (0.163)
MBE98105.RRT	Rock Reach Dam	181	162	78	0.556 (0.130)
MBE98105.WEL	Wells Hatchery	134	165	76	0.978 (0.298)
MBE98106.RIT	Rock Island Dam	169	182	82	0.769 (0.183)
MBE98106.RRF	Rock Reach Dam	167	160	66	1.127 (0.299)
MBE98106.RRT	Rock Reach Dam	184	173	76	0.928 (0.221)
MBE98107.RIT	Rock Island Dam	200	211	88	1.077 (0.238)
MBE98107.RRT	Rock Reach Dam	176	155	87	0.789 (0.227)
MBE98107.WEL	Wells Hatchery	118	124	77	1.339 (0.471)
MBE98110.RIT	Rock Island Dam	181	199	99	0.813 (0.163)
MBE98110.RRT	Rock Reach Dam	198	189	90	1.055 (0.244)
MBE98110.WEL	Wells Hatchery	89	113	45	0.391 (0.152)
MBE98111.RIT	Rock Island Dam	208	215	99	0.623 (0.136)
MBE98111.RRT	Rock Reach Dam	184	162	77	0.754 (0.179)
MBE98112.RIT	Rock Island Dam	175	195	80	0.845 (0.204)
MBE98112.RRT	Rock Reach Dam	148	196	77	0.761 (0.168)
MBE98112.WEL	Wells Hatchery	122	126	59	1.453 (0.581)
MBE98113.RIT	Rock Island Dam	192	218	109	0.602 (0.105)
MBE98113.RRT	Rock Reach Dam	147	141	75	0.843 (0.243)
MBE98113.T5A	Wells Hatchery	112	126	59	0.871 (0.316)
MBE98114.RIT	Rock Island Dam	189	177	96	0.479 (0.126)
MBE98114.RRF	Rock Reach Dam	119	153	67	1.185 (0.455)
MBE98114.RRT	Rock Reach Dam	134	183	92	0.893 (0.228)
MBE98117.RIT	Rock Island Dam	157	203	71	0.693 (0.158)
MBE98117.RRT	Rock Reach Dam	151	159	85	0.696 (0.171)
MBE98117.WEL	Wells Hatchery	114	123	69	0.998 (0.392)
MBE98118.RIT	Rock Island Dam	151	203	80	0.783 (0.197)
MBE98118.RRT	Rock Reach Dam	151	191	73	0.533 (0.117)
MBE98118.WEL	Wells Hatchery	97	136	62	0.736 (0.255)
MBE98119.RIT	Rock Island Dam	168	180	81	0.537 (0.130)
MBE98119.RRF	Rock Reach Dam	140	159	87	0.932 (0.210)

Table A3. (Continued)

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
MBE98119.RRT	Rock Reach Dam	129	133	69	0.445 (0.181)
MBE98120.RIT	Rock Island Dam	154	162	62	0.763 (0.228)
MBE98120.RRT	Rock Reach Dam	132	157	67	1.038 (0.370)
MBE98120.WEL	Wells Hatchery	91	135	55	0.624 (0.181)
MBE98121.RIT	Rock Island Dam	157	187	77	0.639 (0.170)
MBE98121.RRT	Rock Reach Dam	149	160	65	2.308 (1.131)
MBE98121.WEL	Wells Hatchery	75	144	64	0.506 (0.175)
MBE98124.RIT	Rock Island Dam	173	155	62	1.400 (0.616)
MBE98124.RRF	Rock Reach Dam	133	107	49	0.844 (0.341)
MBE98124.RRT	Rock Reach Dam	142	139	67	0.859 (0.237)
MLS98096.LFH	Lyons Ferry Hatchery	301	100	64	0.986 (0.355)
MLS98104.LFH	Lyons Ferry Hatchery	256	107	72	0.970 (0.358)
WDM98155.PD1	Snake River	685	144	22	1.002 (0.549)
WPC98096.T10	Snake River	319	164	115	1.114 (0.332)
WPC98097.T09	Snake River	350	148	110	0.899 (0.268)
WPC98098.T07	Snake River	295	133	106	0.688 (0.175)
WPC98099.T08	Snake River	330	126	132	0.866 (0.216)
WPC98105.CJ4	Snake River	63	32	23	0.884 (0.441)
WPC98134.DR1	Snake River	333	98	31	0.540 (0.203)
				Weighted average of available survival ests.	**0.832 (0.033)**

Table A4. Summary of estimates of John Day project survival by release groups in 1998 for steelhead.

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
AAB98064.RR1	Red River	40	85	39	
AAB98064.RR2	Red River	38	74	26	
AAB98064.RR3	Red River	40	70	22	1.012 (0.159)
AAB98064.RR4	Red River	12	25	10	
AAB98064.RR7	Red River	38	74	28	
EWB98114.SNK	Snake River Trap	20	43	36	0.792 (0.287)
EWB98118.SNK	Snake River Trap	22	36	27	2.514 (1.771)
EWB98124.SNK	Snake River Trap	21	40	34	0.678 (0.248)
HLB98112.P16	Dworshak H.	39	60	42	
HLB98112.P20	Dworshak H	29	39	24	
HLB98112.P24	Dworshak H	27	45	19	1.160 (0.187)
HLB98112.P28	Dworshak H	19	58	25	
HLB98112.P32	Dworshak H	34	53	30	
LRB98107.P02	Dworshak H	15	27	20	
LRB98107.P03	Dworshak H	21	38	17	
LRB98107.P39	Dworshak H	13	30	9	0.868 (0.122)
LRB98107.P48	Dworshak H	23	32	11	
LRB98107.P59	Dworshak H	23	51	21	
LRB98107.P62	Dworshak H	33	50	28	
LRB98124.GR1	Grande Ronde	24	30	33	1.384 (0.701)
LRB98125.RI1	Rock Island D.	27	50	31	0.885 (0.326)
LRB98126.GR1	Grande Ronde	31	48	32	1.337 (0.375)
LRB98130.RI1	Rock Island D.	25	43	34	0.848 (0.326)
MLS98082.DAY	Dayton Acclim. P.	29	34	25	0.607 (0.206)
MLS98083.COT	COTP(?)	30	38	25	1.630 (0.893)
MLS98099.STH	Lyons Ferry H.	38	76	34	0.685 (0.195)
MLS98100.ENR	Tucannon River	33	35	29	1.238 (0.346)
MLS98100.MAR	Tucannon River	38	50	29	
PAK98116.NT2	Imnaha River Trap	22	24	25	2.913 (2.965)
PAK98118.NT1	Imnaha River Trap	34	39	35	2.184 (1.006)
PAK98140.NT1	Imnaha River Trap	41	91	39	0.705 (0.177)
PAK98140.NT2	Imnaha River Trap	14	28	10	
TDR98054.12E	Hazard C.	23	28	23	1.340 (0.767)
TDR98054.13E	Salmn R. E.F. W.	26	33	25	0.740 (0.321)
TDR98054.M4W	Salmon R.	22	35	31	1.233 (0.511)
TDR98054.M8W	Salmon R.	20	35	25	
TDR98055.N16	Pahsimeroi R. T.	20	32	23	0.948 (0.351)
TDR98056.H41	Sawtooth H.	27	27	20	
TDR98056.H43	Sawtooth H.	24	24	34	1.110 (0.308)
TDR98056.H51	Sawtooth H.	24	21	24	
TDR98056.H88	Hazard C.	24	55	33	1.597 (0.550)
WDM98110.TAL	Lower Granite D.	30	21	31	1.052 (0.509)
WDM98111.TAL	Lower Granite D.	39	53	44	0.829 (0.231)
WDM98112.TAL	Lower Granite D.	30	50	47	1.242 (0.348)
WDM98113.TAL	Lower Granite D.	32	38	24	1.076 (0.390)

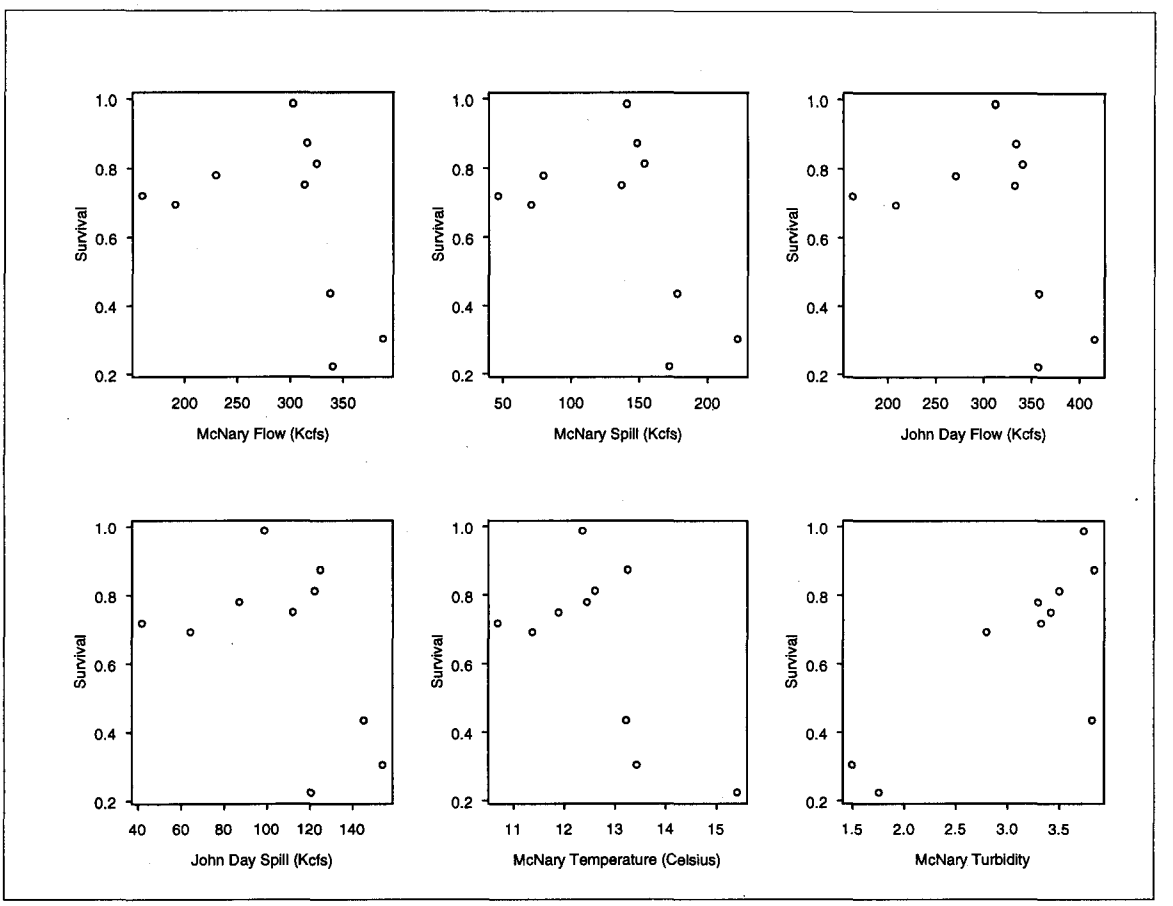
Table A4. (Continued)

File ID	Release Site	Total smolt observed at: McNary	John Day	Bonneville or below	John Day Survival
WDM98114.TAL	Lower Granite D.	28	45	31	1.713 (0.621)
WDM98115.TAL	Lower Granite D.	37	52	29	0.912 (0.228)
WDM98116.TAL	Lower Granite D.	35	63	27	0.755 (0.248)
WDM98117.TAL	Lower Granite D.	63	127	73	0.721 (0.131)
WDM98118.TAL	Lower Granite D.	54	117	58	0.852 (0.141)
WDM98119.TAL	Lower Granite D.	57	121	75	1.133 (0.267)
WDM98120.TAL	Lower Granite D.	45	84	49	1.115 (0.275)
WDM98121.TA2	Lower Granite D.	50	99	63	1.243 (0.179)
WDM98121.TAL	Lower Granite D.	54	119	70	
WDM98122.TA2	Lower Granite D.	50	84	56	1.377 (0.218)
WDM98122.TAL	Lower Granite D.	84	133	63	
WDM98124.TA2	Lower Granite D.	46	87	56	1.369 (0.202)
WDM98124.TAL	Lower Granite D.	75	109	60	
WDM98125.TA2	Lower Granite D.	36	72	36	0.846 (0.175)
WDM98125.TAL	Lower Granite D.	51	71	54	
WDM98126.TA2	Lower Granite D.	34	47	35	0.985 (0.194)
WDM98126.TAL	Lower Granite D.	74	72	65	
WDM98127.TA2	Lower Granite D.	44	72	43	0.999 (0.181)
WDM98127.TAL	Lower Granite D.	53	89	47	
WDM98128.TA2	Lower Granite D.	61	74	31	0.777 (0.136)
WDM98128.TAL	Lower Granite D.	62	86	55	
WDM98129.TA2	Lower Granite D.	62	68	38	0.877 (0.159)
WDM98129.TAL	Lower Granite D.	44	75	42	
WDM98131.TA2	Lower Granite D.	47	56	33	0.896 (0.186)
WDM98131.TAL	Lower Granite D.	58	66	46	
WDM98132.TA2	Lower Granite D.	56	91	40	1.489 (0.331)
WDM98132.TAL	Lower Granite D.	52	81	38	
WDM98133.TA2	Lower Granite D.	52	91	30	1.372 (0.285)
WDM98133.TAL	Lower Granite D.	53	91	48	
WDM98134.TA2	Lower Granite D.	46	73	42	1.005 (0.211)
WDM98134.TAL	Lower Granite D.	31	73	41	
WDM98135.TA2	Lower Granite D.	47	87	34	0.778 (0.165)
WDM98135.TAL	Lower Granite D.	48	60	25	
WDM98138.TAL	Lower Granite D.	21	41	21	0.944 (0.386)
WDM98142.TA2	Lower Granite D.	22	59	31	1.382 (0.581)
WDM98146.TAL	Lower Granite D.	55	127	32	0.667 (0.133)
WDM98147.TAL	Lower Granite D.	53	110	37	1.117 (0.196)
WDM98148.TAL	Lower Granite D.	41	106	39	0.893 (0.205)
WDM98149.TAL	Lower Granite D.	60	76	35	1.078 (0.260)
				Weighted average of available survival ests.	**1.043 (0.038)**

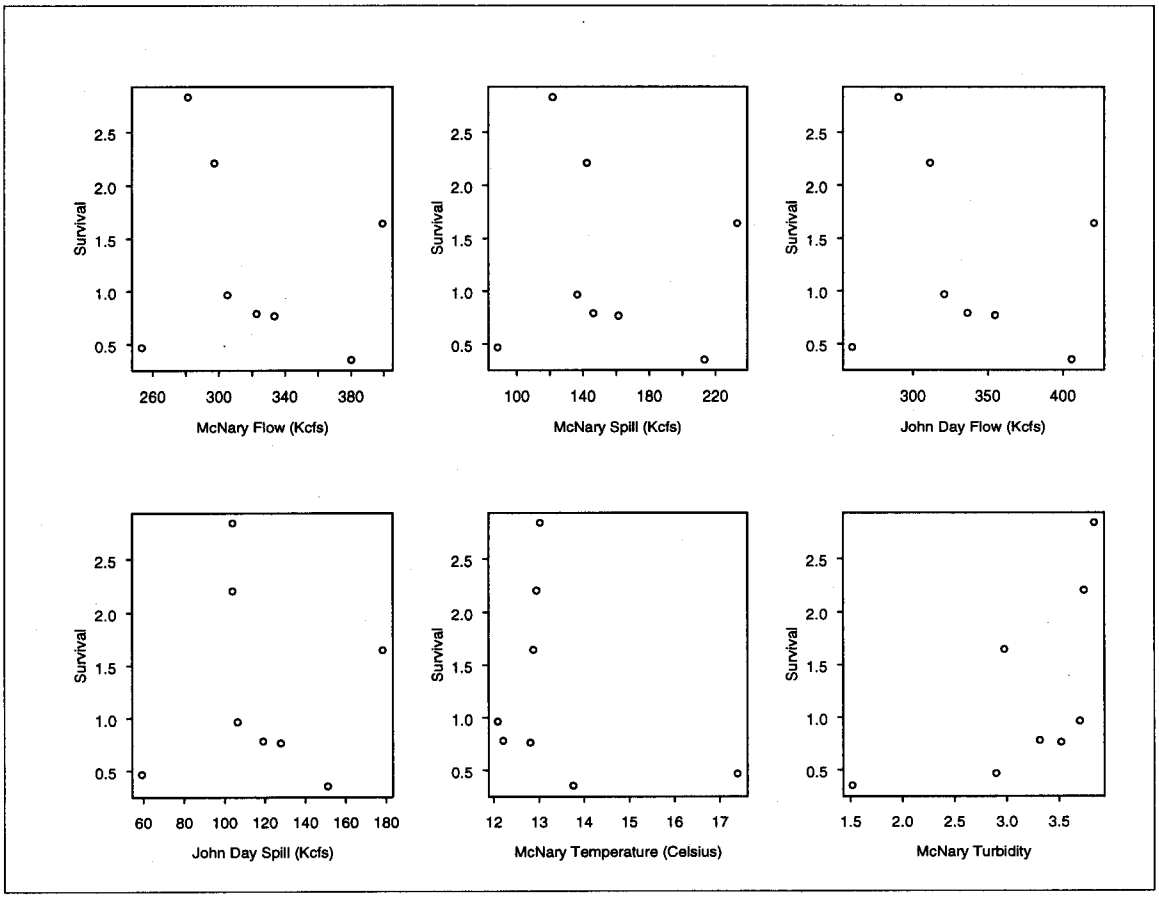
Attachment E - Appendix B

Scatterplots of survival estimates versus ambient river conditions for different salmonid stocks in the John Day project for 1998.

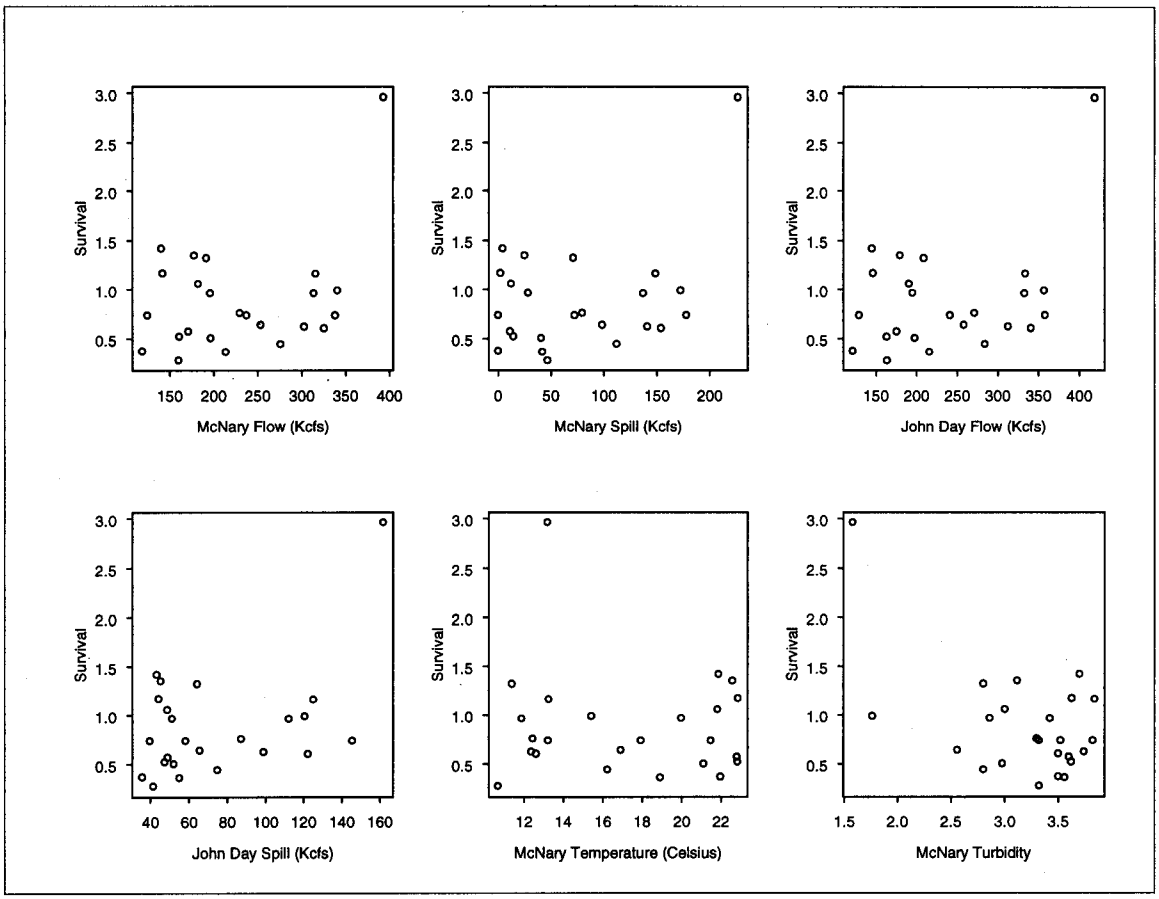
a. Spring chinook salmon



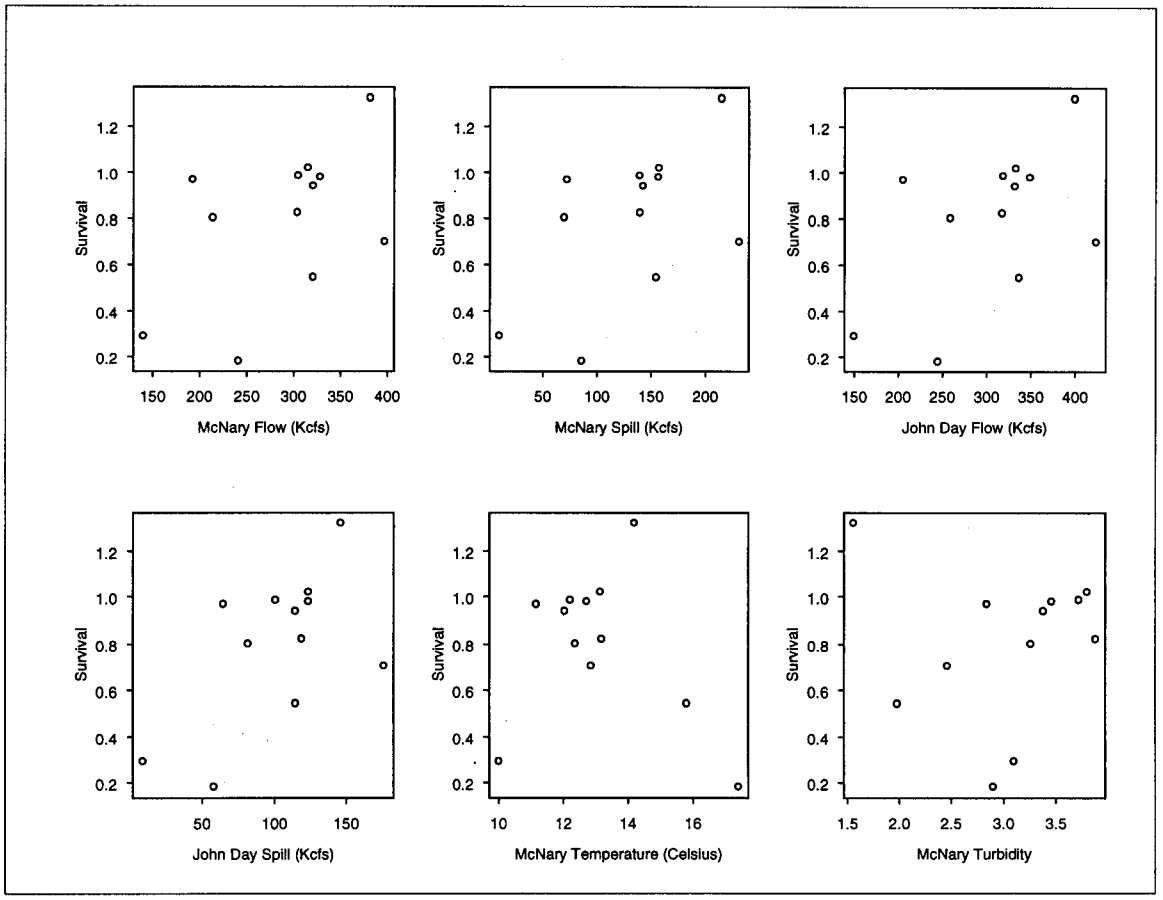
b. Summer chinook salmon



c. Fall chinook salmon



d. Steelhead


Survival
McNary Flow (Kcfs)
McNary Spill (Kcfs)
John Day Flow (Kcfs)
John Day Spill (Kcfs)
McNary Temperature (Celsius)
McNary Turbidity

Attachment E: *Modeling the Impacts of John Day Drawdown on the Survival of Salmonid Stocks.*

Modeling the impacts of John Day drawdown on the survival of salmonid stocks

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1 Introduction

The purpose of this study is to model the effect of a John Day Reservoir drawdown on anadromous salmonid populations, particularly populations listed under the Endangered Species Act. The approach utilizes passage models to characterize smolt survival through the hydrosystem and incorporates the passage model results into life-cycle models to characterize the effects of John Day actions on adult population levels. Because significant uncertainties exist on the effect of mitigation actions on fish survival and on how observed survivals are partitioned throughout the life cycle, a number of hypotheses are included in the analysis. The goal is to characterize the average effects over a range of hypotheses and to demonstrate the range of effects resulting from different hypotheses.

To produce estimates of the impacts of John Day mitigation actions on adult population levels, this analysis has used three methods. First of all, we utilized methods and results produced by PATH (Plan for Analyzing Testable Hypotheses, a group of approximately 25 scientists from state, tribal and federal agencies). The outputs from the PATH analysis are probabilities of meeting survival and recovery standards, and results relevant to this study are reported. Second, we simplified the PATH analysis (by removing the Bayesian decision analysis framework) to produce mean equilibrium spawner levels for particular actions under a range of hypotheses. This method produces intuitive results and can be used to estimate the gain or loss of spawning adults when analyzing one action compared to another. Third, for the more detailed analyses of actions at the John Day project, we further simplified the life-cycle analyses to produce only the difference in spawner levels under two actions. This simplification arises from the assumption that actions taken at the John Day project will not affect survivals in other life stages (e.g., ocean survival or egg to smolt survival) with the result that these survivals will cancel out when comparing two actions.

The specific actions considered at the John Day project were reservoir drawdown to natural river level, reservoir drawdown to the spillway crest, and drawdown to natural river level but using

John Day pool as a storage reservoir for flood control under high flow conditions. The analysis of the direct impacts of these actions on the survival of migrating smolts was conducted using the Columbia River Salmon Passage (CRiSP) model, developed at the University of Washington. In addition results from the FLUSH model (Fish Leaving under Several Hypotheses, developed by state and tribal agencies) were incorporated into life-cycle analyses where available.

For this report, Snake River spring and fall chinook were analyzed. Both these stocks are listed as threatened under the Endangered Species Act and were the focus of the PATH analysis. In addition, Hanford Reach fall chinook and Upper Columbia spring chinook were evaluated..

To summarize, three model systems were applied in the analysis. The PATH Bayesian life-cycle model to estimate the probabilities of survival and recovery, a deterministic model to determine the equilibrium and maximum sustainable spawner populations, and the CRiSP passage model to evaluate the impacts of drawdown on smolt survival and fish travel time.

2 Actions

To model John Day Reservoir drawdown two conditions are evaluated: spillway crest and natural river. The spillway crest draws the reservoir down to the crest of the John Day Dam spillway at 210 ft. Fish would pass through the spillway and plunge 50 ft. down into the tailrace at an elevation of about 160 ft. The full pool elevation is between 257 and 268 ft. giving a spillway crest drawdown level of approximately 50 ft., assuming a typical operating pool elevation of 265 ft. and a forebay elevation 5 ft. above the crest.

Under natural river drawdown, the reservoir elevation is taken to the level of the Dalles reservoir at the John Day tailrace. The natural channel of tailrace is at an elevation of 139 ft. Current minimum tailwater elevation is 155 ft. During a 2-yr flood the tailwater elevation is 166 ft. and under the 20 yr. flood it is 172-ft. Under these conditions, the natural river elevation would vary between 155 and 172-ft. Taking the typical elevation of the natural river as 165-ft., the natural river elevation drop is 100 ft.

John Day reservoir is used for flood control and has a capacity under current operating conditions to store 534,000 AF. The temporary storage of this amount of water requires lowering the elevation in anticipation of a flood event and then raising the level to approximately full pool. The net elevation change is approximately 10 ft. This level of flood control was sufficient to manage the 1997 spring runoff, which was one of the largest on record. In the analysis conducted here the same level of flood control is assumed and the resulting elevation change for a spillway crest and natural river control are estimated.

Table 1 lists the actions analyzed in this report. Some of the actions include a John Day drawdown in addition to a drawdown of the four lower Snake River projects; other actions treat a John Day drawdown without a Snake drawdown.

Table 1. Alternative actions evaluated

Action	Description
A0:	Base conditions as the hydrosystem without transportation
A1:	Base conditions as the hydrosystem is currently operated
A2:	Improved transportation with full transport of fish
A3:	Drawdown of the 4 lower Snake River dams
B1:	A3 with John Day drawdown to natural river level
B2:	A3 with John Day drawdown to spillway crest
B3:	A3 with John Day drawdown to natural river with flood control
C1:	Base, no transport, John Day drawdown to natural river level
C2:	Base, no transport, John Day drawdown to spillway crest
C3:	Base, no transport, natural river John Day drawdown with flood control
D1:	Natural river level John Day drawdown and Snake transport
D2:	Spillway crest John Day drawdown with flood control and Snake transport
D3:	Natural river level John Day drawdown with flood control and Snake transport

3 Measures of fish performance

To assess the performance of the drawdown and full transportation actions relative to the current hydrosystem operations, three measures of population performance were used. The first two are probabilities of meeting survival and recovery goals as defined by NMFS Jeopardy Standards. These were the measures used by PATH. The third measure is the equilibrium level of spawners for each recovery action. In addition to these measures, the absolute difference between pairwise comparisons of alternative actions for each measure is reported.

3.1 Survival Standard: 24-year

This measure was developed by PATH and was selected by NMFS as a primary survival standard for the A-Fish Appendix of the Biological Opinion. It is the fraction of simulation runs for which the average spawner abundance over a 24 year time period exceeds a predefined threshold for each index stock. For spring chinook, the survival threshold is 150 or 300 spawners depending on the river. For fall chinook, two survival thresholds have been proposed 300 and 700 spawners (Marmorek et al. 1998, Peters et al. 1999).

3.2 Recovery Standard: 48-year

This measure was developed by PATH and was selected by NMFS as a primary recovery standard. It is the fraction of simulation runs for which the average spawner abundance over the last 8 years of a 48-year simulation is greater than a specified level, which is 60% of the pre-1971 brood-year average spawner counts in each of the index streams (Marmorek et al. 1998). For fall chinook two recovery thresholds have been proposed: 2500 and 5100 spawners (Peters et al. 1999).

3.3 Equilibrium spawners

The equilibrium measure of the population is the level at which the spawning recruits of a brood are exactly sufficient to replace their parental brood. With typical salmon life-cycle models, in the absence of environmental variations and a constant harvest rate, the equilibrium population level is a stable point that a stock will approach over time. Simply put, the equilibrium is a measure of the number of fish a habitat can maintain with a specific set of management actions including hydro operations and fisheries regulations.

3.4 Smolt Passage Measures

The smolt passage measures provide a quantitative description of the direct effects of drawdown actions on smolt passage. These are valuable because they are not complicated by hypotheses on the linkage between effects of passage on ocean survival. Passage measures are defined for migration from the face of lower Granite Dam to the tailrace of Bonneville Dam (used in fall chinook analysis) or from the top of Lower Granite Pool to the tailrace of Bonneville Dam (used in the spring chinook analysis). The measures include fish travel time (FTT) in-river survival (Vn) and the total hydrosystem survival of both transported and in-river passing smolts (Sm). In addition, reported are the fractions of smolts in Bonneville tailrace that arrived through transportation (Pb) and in-river passage (1-Pb).

4 Life-cycle Framework

The models used in PATH, by the National Marine Fisheries Service, and the analysis in this report all are based on a salmon life-cycle model with low number of life history stages. Generally four important stages are identified (Fig. 1). The first stage is a freshwater spawning stage that in this report extends from the adult spawners laying eggs in redds to the beginning of smolt migration. This first freshwater stage characterizes the intrinsic freshwater production of a stock in terms of the number of progeny (per spawner) that survive through the stage. The second stage characterizes the migration of the smolts through the hydrosystem from the Lower Granite

project to the tailrace of Bonneville Dam. The third stage characterizes the ocean and estuary survival and ends with the adults at the entrance of the adult bypass channels of Bonneville Dam. The fourth stage begins with the adults entering the upstream bypass channels and ends just prior to the spawning event. These four stages describe a complete salmon life cycle. Further divisions can be made to characterize other sub-stages within each stage, but for the purpose of comparing the impacts of the drawdown of John Day reservoir to other actions on the hydrosystem, these four elements are sufficient.

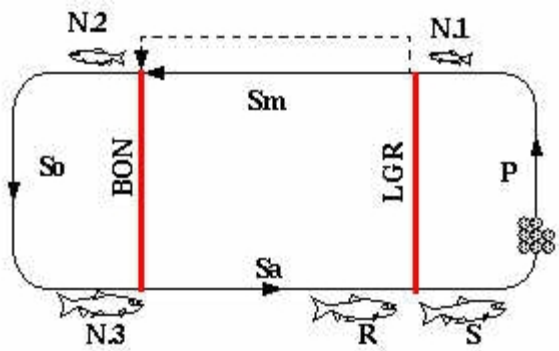

Figure 1. Life cycle of salmon extending from freshwater production stage, P, to hydrosystem survival, Sm, from Lower Granite Dam (LGR) to Bonneville Dam (BON), which includes in river and transport passage, to ocean survival, So, to upriver adult migration survival, Sa. S spawners produce R recruits.

The equation related to Figure 1 can be expressed

eq (1) $$R = S \cdot P \cdot Sm \cdot So \cdot Sa \cdot Ho \cdot Hr$$

where P is the production of smolts per spawner and may contain some form of density dependence, Sm is the survival of smolts through the hydrosystem by both in-river and transportation passage routes, and So is the survival of fish passing through the estuary. Sa is the survival of the returning adults as they migrate through the hydrosystem, with the inclusion of prespawning survival and river harvest. Ocean and river harvest mortality are defined as (1- Ho), and (1- Hr).

4.1 Freshwater production (P)

The freshwater production stage describes now many smolts are produced per spawner. The productivity depends on the number of spawners, with productivity decreasing as the number of spawners increases. In the PATH analysis, the density effects equation included the possibility of depensation in which productivity declines at low numbers of spawners. The spawner recruit data did not reveal any depensation, so functionally PATH used a Ricker density compensation equation. Here we use the functional form of the Ricker equation to express density effects in freshwater production:

eq (2) $$P = \exp(a_{fw} - b \cdot S)$$

The a_{fw} parameter defines only recruits to the smolt stage so the term is different from the Ricker "a" coefficient, which defines recruits to the spawning stage. The "b" parameter is the same as in the Ricker equation and is a measure of the decline in productivity with increasing spawner numbers.

4.1.1 Density dependence parameter (b)

The density dependent factor b, used in eq(2), is derived from the regression of natural log(recruits/spawner) versus spawners for spring and fall chinook from the Snake River basin. Table 2 presents estimates for this parameter and carrying capacity (a/b) for Snake River spring and fall chinook for the post-1974 period. For spring chinook, the b is taken as the average of the six Snake River index stocks. For the fall chinook, a single stock is represented, and the estimates of recruits take harvest into account. For the Snake River fall chinook, the regression is statistically insignificant ($R^2 = 0.01$). The resulting b is only useful for giving a ball park estimate of the fish numbers in all of the analyses.

Table 2: Ricker coefficients for spring and fall chinook from the Columbia/Snake River system stocks. Spawners were on redds and recruits were estimated to mouth of Columbia River. The Snake River spring chinook estimates are the mean values of 7 index stocks.

Chinook	Region	Ricker a	Ricker b	Reference
spring	Snake River	0.73	0.00174	Schaller et al. 1999
spring	Upper Columbia	1.04	0.00098	Schaller et al. 1999
spring	Lower Columbia	1.48	0.00195	Schaller et al. 1999
spring	Upriver Aggregate	0.41	0.0000016	Schaller et al. 1999
fall	Snake River	1.96	0.00027	Peters et al. 1999
fall	Columbia R. URB	2.65	0.00002	Peters et al. 1999
fall	Deschutes R.	2.84	0.00037	Peters et al. 1999

4.1.2 Stock equilibrium numbers

To extrapolate from the representative index stocks to the basin-wide impact on the species, estimates of the number of individual demes or stocks is required. The endangered stocks in the Snake River Basin are designated as wild and natural stocks. Wild stocks have genetic makeup unlikely to have been altered by hatchery fish. Natural stocks are naturally spawning fish that have genetically mixed with hatchery fish. In the Snake River Basin 23 natural and wild spring chinook and 9 summer chinook stocks were identified by Chapman et al. 1991. Stocks of hatchery origin include 12 spring chinook stocks and 2 summer chinook stocks. One wild-natural population of fall chinook has been identified (Chapman et al. 1991). The total number of natural and wild spring and summer chinook stocks is 32. Members of the Plan for PATH group suggested a more representative number of stocks is 38. This larger estimate was used in this report.

4.2 Passage survival (Sm)

Passage survival in eq(1) must be characterized in drawdown and transportation alternatives. In PATH, a general passage survival equation was developed that accounts for survival from above Lower Granite Dam to the tailrace of Bonneville Dam. The general model includes both direct transportation survival and in-river survival. Here we express smolt survival in a simplified heuristic form, with one transport dam:

eq (3) $$Sm = P1 \cdot Vt + (1 - P1) \cdot Vn$$

where P1 is the percent of the run passing the dam that are transported, Vt is the direct transportation survival, and Vn is the survival of the in-river passing fish. The actual passage models account for transport at a number of dams, adjusting survival of fish down to each transport dam.

4.2.1 In-river survival (Vn)

The in-river smolt passage survival is an important assumption in determining the relative effectiveness of the different actions. In PATH two smolt passage models were used, CRiSP and FLUSH, and subsequent to these model NMFS has developed a simple model for its Anadromous Fish Appendix. Although the models have varying degrees of complexity, CRiSP and FLUSH treat dam passage mortality in similar ways. The differences are in they way they formulate reservoir mortality. The FLUSH spring chinook model has an explicit travel-time /survival relationship in which the rate of mortality increases with time of migration through the reservoirs. This causes the greatest mortality to occur in the lower river, and it makes the survival sensitive to total time in passage and to flow. In the CRiSP fall and spring chinook models and the FLUSH fall chinook model, survival estimates are produced by more mechanistic models where reservoir survival rate is related to factors such as predator abundance. In their general form, the reservoir mortality rate increases with temperature, and for CRiSP it also increases for elevated levels of supersaturation. In these models the mortality rate is not directly dependent on

the time of passage, although longer fish travel times will result in lower survivals, all other factors being held constant. The NMFS model assumes no flow/survival relationship, and reservoir and dam mortality are not distinguished. Reservoir survival is essentially constant for each reservoir. The three classes of model can be expressed in general forms as

eq (4)	Spring FLUSH	$Vn = Vdam^{X} \cdot FTT^{-B}/(\exp(A \cdot FTT) - 1) + 1)$
	CRiSP and fall FLUSH	$Vn = Vdam^{X} \exp(f(T) \cdot FTT)$
	NMFS	$Vn = Vproj^{X}$

where Vdam is the survival of dam passage, X is the number of dams the fish pass, FTT is the fish travel time through the hydrosystem, and T is water temperature. In the NMFS model, the average reservoir and dam mortality components are combined in a single term, Vproj. In the spring FLUSH model the reservoir mortality depends on FTT as described by A and B, which are constants obtained by fitting the model to survival data. In CRiSP and fall FLUSH, the reservoir mortality is described by a mortality rate function f(T), which depends on a predator temperature response function, predator consumption rates, and a predator abundance index over the reservoirs. In CRiSP, mortality under high gas levels is also taken into account although this is generally a minor source of mortality.

To estimate the in-river survival of fish under drawdown conditions, a number of auxiliary hypotheses were used. In general, the drawdown survival was estimated independently giving a two part equation: the first part being survival of the drawdown section, the second part being the survival through the impounded sections estimated by the passage models. The survival of in-river fish is modified to

eq (5) $Vn = Vd \cdot Vi$

where Vd is the survival through a drawdown section of the river under a specific drawdown alternative, and Vi is the survival through the impounded sections.

4.2.2 Natural River Drawdown survival (Vd)

A great deal of uncertainty exists over what survivals will be in free-flowing river segments after reservoirs are drawn-down to natural river levels. In PATH, upper and lower bound survivals were used for the drawn-down Snake reservoirs in an effort to characterize the range of uncertainty. These estimates were developed by applying direct or indirect estimates of survival through existing free-flowing reaches to future drawn-down reaches on a per km basis. For fall chinook PIT tag survivals from 1995-1998 were used. These survival estimates encompass both free-flowing and impounded segments, and two methods were used to extract survivals through the free-flowing segment. (Peters et al. 1999). For spring chinook, free-flowing survival estimates were based on survival estimates from the Whitebird trap in the Salmon River to the uppermost dam, either Ice Harbor (1966-1969) or Lower Granite (1993-1996) (Marmorek and Peters 1998). For spring chinook, the John Day estimates were derived by taking survival per km from the Snake River studies and adjusting to the length of the John Day reservoir. For fall chinook, the John Day estimates were derived from the passage models under drawdown conditions. Upper and Lower bound survival estimates are provided in Table 3.

Table 3. Drawdown survivals Vd through free-flowing reaches of the Snake River and John Day reservoir.

Chinook type	river segment	Lower estimate	Upper estimate
spring	Snake R.	0.85	0.96
fall	Snake R.	0.61	0.90
spring	John Day	0.90	0.98
fall	John Day	0.87	0.87

Total system survival Sm generated from CRiSP for the different alternatives are given in Tables 21 and 22.

4.2.3 Spill Crest and Food Control Drawdown survival (Vd)

To estimate survival in John Day pool under a spillway crest drawdown and under natural river with flood control, the CRiSP passage model (Anderson et al. 1996) was used. In this model system reservoir elevation and river flow are used to estimate water velocity. The water velocity in turn is used to predict fish velocity, and this in turn is used to predict fish reservoir survival. Estimating the impacts of spillway crest drawdown and natural river drawdown under flood control then results in estimating the impact on fish velocity which effects survival. These capabilities are integral to the CRiSP model and so estimating these special conditions involved only defining reservoir elevations. Estimating the change in reservoir elevation with flood control involved additional calculations, which are detailed below.

The CRiSP model represents reservoirs through a number of rectangles giving the reservoir side and thalweg slopes as illustrated in Figure 2. The tailwater and forebay ends of the reservoir are set at the actual elevations giving an average thalweg gradient. In addition the angle of sides are set to approximate the slopes of the reservoir banks. As the reservoir is drawn down the upstream portion enters a free-flowing stage where the velocity is constant determined by drag properties of the streambed. In these calculations the free-flowing velocity, Ufree, was set at 5 ft/s.

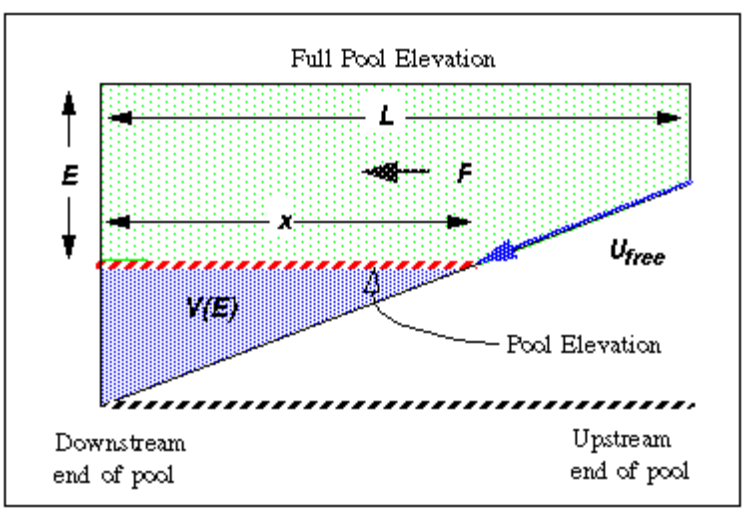

Figure 2. Reservoir with free-flowing and impounded portions. The terms are reservoir elevation E, length L, volume V(E), flow F, and stream velocity Ufree.

Under these model conditions, the reservoir water velocity increases in an approximately linear manner with elevation drawdown. At natural river drawdown levels, though, the river velocity reaches a maximum as velocity is determined by the drag of the channel (Table 4).

Table 4: Water velocities (miles/day) for different flows and pool elevations

Action	Elevation drop (ft)	Normal flow (236 kcfs)	High flow (400 kcfs)
Full pool	0	16	25
Spillway crest	50	53	80
Natural River	100	90	95

To determine the effects of flood control on river elevation and velocity the relationship between elevation and velocity is used. The average river velocity, U, at elevation z is equal to the flow, F, divided by the cross-sectional area, A:

eq (6) $U(z) = F/A.$

The velocity under flood control, in which the elevation is raised to absorb the flood control water, can be expressed as

eq (7) $U(z1) = L \cdot F/(V + L \cdot F/U(z0)).$

Where z0 and z1 are the base and flood control elevations, L = 76.4 miles is the John Day reservoir length, F = 500 kcfs is the flow at which flood control is typically required and V = 534,000 acre-feet is the flood control volume. If the high flow at a natural river elevation gives a velocity of 100 miles/day or approximately 6 ft/s then the velocity after absorbing the flood control volume becomes about 50 miles/day or 3 ft/sec. Since velocity is approximately linearly related to elevation, the change in reservoir elevation (from natural river conditions) with flood control can be estimated. The elevation under natural river would rise to about the spill crest elevation, and under a spill crest drawdown flood control would raise the elevation another 10 to 15 feet above the spillway crest. These estimates are approximate since they are developed on the assumption of simplified reservoir geometry, and the natural river segment velocity is fixed. The

hydraulics is sufficient to estimate the impacts of natural river level flood control on smolt passage.

4.3 Ocean/estuary survival (So)

The survival of fish, between the time they leave the tailrace of Bonneville dam as smolts and return to the fish ladders of Bonneville dam as spawning adults, is an important life stage that exhibits a large range of variability from year to year. A number of assumptions in PATH were developed to characterize the possible factors that determine survival during the ocean residence life stage. Of particular importance, are the effects of hydrosystem passage route on ocean survival. Because fish pass through the hydrosystem in transportation and as run of the river fish, there is the possibility that the ocean survival is different for fish from each passage route. The basic equation for survival in the ocean life history stage (which in this definition includes the segment form Bonneville to the estuary), accounting for the two passage routes, is

eq (8) $$So^* = (\lambda t \cdot Pb + \lambda n \cdot (1-Pb)) \cdot \lambda o$$

where Pb is the proportion of fish that entered below Bonneville via transportation, (1-Pb) is the proportion of fish entering below Bonneville via in-river passage, λo is the base ocean survival common to both groups, $\lambda t \cdot \lambda o$ is the ocean survival of transported fish and $\lambda n \cdot \lambda o$ is the ocean survival of non-transported fish. The passage route specific survivals λt and λn may change from year to year depending on hydrosystem operations, ocean and climate conditions, and any changes in the fish condition prior to, or during, migration. The common survival λo is constant and typically, in a life-cycle analysis, it is absorbed into the density independent term of the stock recruitment function. That is, λo is contained within exp(a) of the stock recruitment equation mused in this formulation, which is $R = S \exp(a - b \cdot S)$.

Since there are insufficient data to characterize the time-varying ocean survivals of transported and non-transported fish, the equation is rewritten to express the time varying survival of the non-transported fish only. The ocean survival of the transported fish is then characterized relative

to the non-transported fish survival. Also the base term λo can be ignored in the deterministic model because in the pairwise comparisons of actions, ratios of productivity are taken and so λo cancels. With these simplifications the ocean survival equation becomes

eq (9) $$So = \lambda n \cdot (D \cdot Pb + 1 - Pb)$$

where the ratio of the ocean survivals of the transport to the non-transported is

eq (10) $$D = \lambda t / \lambda n.$$

The estimation of D and λn can be derived in various ways depending on the types of data available. The resulting values of these terms, and how they have changed over time, are extremely significant to the conclusions on the effectiveness of fish transportation as a fish recovery action. Therefore these sources of mortality are discussed further in the sections below.

4.3.1 Extra mortality

Extra mortality is defined as the differences in the mortality estimated from the spawner/recruit data and the mortality that can be accounted for by the smolt passage models and assumptions on adult upstream mortality. Extra mortality for in-river fish is defined as $1- \lambda n$; in other words, if there was no extra mortality in a given year, the ocean survival would be equal to the base ocean survival, λo. For spring chinook, an increasing trend in ocean mortality corresponds with the development of the Snake River dams in the 1970s, the increase in hatchery production, and shift in the ocean climate conditions in 1977. As a result, a number of factors could contribute to the trend in mortality and it is uncertain as to the significance of any individual factor, or others not yet considered. In PATH these possible factors were considered individually and a combination hypothesis was not considered. In particular, the trend in ocean mortality was hypothesized to be the result of either degraded freshwater conditions, increased stress in hydrosystem passage, or changes in the ocean ecosystem. These were designated the BKD, the HYDRO and the CLIMATE hypotheses. Functionally the three hypotheses attribute the cause of ocean mortality to different life stages as described below:

BKD hypothesis states that the extra mortality is associated with a change in the wild fish condition, possibly from disease such as bacterial kidney disease (BKD) resulting from increased hatchery production beginning in the late 1970s. Under this hypothesis the extra mortality is endemic to the wild Snake River chinook and is here to stay under changes in the hydrosystem operations.

HYDRO hypothesis postulates that the extra mortality is associated with the hydrosystem and specifically the Snake River portion of the hydrosystem. Under this hypothesis it could be associated with the cumulative stress in hydrosystem passage. Consequentially, in this hypothesis removing dams removes the stress and results in higher survival below the hydrosystem.

CLIMATE hypothesis postulates that the extra mortality is associated with a climate/ocean regime shift that occurred in the late 1970s. Under this hypothesis the extra mortality only disappears if the climate shifts back to a fish-favorable ocean regime and the effect is independent of any changes made to the hydrosystem.

Some details of the linkages between life-stage survivals were developed in PATH, but are not important to explore here. What is important though, is that specific mechanisms have not been identified for any of the hypotheses, nor has significant correlation between of variables related to the hypotheses and ocean survival of non-transported fish been demonstrated. As a result of this inability to clarify the mechanisms, the results from PATH should be considered as exploratory of the range of possible consequences.

The range of ocean survivals, expressed as the extra mortality factor as $1- \lambda n$, was derived in PATH from the combination of a life-cycle model with a passage model for the spring and fall chinook. The essential survivals are given in Table 5. Note the examples in the table characterize ocean survival as affected by the extra mortality factor. The possible levels of extra mortality depend on mortality assumptions in the retrospective analysis of stock dynamics. These details

are beyond the scope of the deterministic analysis here. Table 5 is intended to illustrate the general ranges over which extra mortality contributes to ocean survival of fish.

Table 5: Characteristic ocean survival factor λn, as determined by extra mortality, under different passage model hypotheses using the Delta model developed in PATH (Hinrichsen and Paulsen 1998) with ranges (min-max). Spring λn estimates from regressions of Vn vs. λn with Vn = 0.2 for 1975-1990 and Vn = 0.4 for 1952-1990 period. Fall chinook estimates from spawner/recruit analysis (Peters et al. 1999).

Chinook	years	CRiSP	FLUSH
spring	1952-1990	0.7 (0.1-1.5)	0.75 (0.1-1.5)
spring	1975-1990	0.4 (0.1-1.8)	0.75 (0.2-1.31)
fall	1964-1991	1	1

4.3.2 Delayed mortality

The transportation efficiency factor D, which is the ratio of ocean survival of transported fish to the ocean survival of non-transported fish, is critical in determining the relative effectiveness of recovery actions. In PATH, this factor has been referred to as the delayed mortality because mortality is thought to occur somewhere below Bonneville Dam as a result of smolts being transported. If D equals one, then the survival from in-river and transport hydrosystem passage routes are equal and fish experience no delayed mortality as a result of being transported. If D is less than one, then the transported fish suffer a delayed mortality relative to the in-river fish.

Exactly where this mortality occurs is unknown though, and the mechanisms resulting in higher mortality of transported fish are unknown. Observed survivals of fish held after transportation range between about 80% and 100% (Reference). Furthermore, radio tracking studies of transported and in-river fish tagged at Bonneville Dam indicate equal survivals in the two groups down to the estuary where the salt water makes the radio tag inoperative.

Estimating D has been problematic for both spring and fall chinook. For spring chinook, D is calculated from estimates of the in-river smolt survival times the ratio of returning adults marked as smolts for transportation and in-river passage groups. As a result, estimates of D have several

critical assumptions that increase the uncertainty, especially in the early years of the transportation program, prior to the development of the PIT tag technology. In the early years of the transportation program, a large fraction of the transport studies control fish were transported at a lower dam so the transport to control ratio of the returning adults was actually a comparison of returns of fish from two transport sites. The use of these data is problematic for assessing the difference of transport and in-river fish because further assumptions are required to correct for the transported control fish. Furthermore, recent transport studies indicate that the timing of arrival of fish to the estuary has a significant impact on their ocean survival (Hinrichsen et al. 1996). PATH did not fully address or resolve these issues and so the estimates of D are highly uncertain. Two basic approaches were taken to estimate D in PATH, and it was determined the most important factor was the choice of smolt passage model, FLUSH or CRiSP. Additionally, in the NMFS A-fish Appendix a high value of D was explored, based on the recent PIT tag studies.

Although these details are important in evaluating the historical D, the most important hypotheses concern the D current and future levels. The possible ways that D could have changed from the early years of fish transportation is summarized in the hypotheses listed in Table 6. For spring chinook, the dividing year between the early and the current levels of D is taken as 1980. Prior to 1980 the transportation system was experimental and significant handling problems that stressed the fish were evident at the transport dams (reference). The geometric averages of D for spring chinook transportation for the three hypotheses are given in Table 6.

Table 6: Spring chinook geometric average estimates of D for early experimental period (pre-1980) and current/prospective period (post-1980).

Hypotheses	pre-1980	post-1980
FLUSH	0.476	0.351
CRiSP	0.18	0.65
NMFS	---	0.80

Estimates of D for fall chinook are even more uncertain than for spring chinook because there were no fall chinook transport experiments on which to estimate D independent of the spawner recruit data. In the PATH fall chinook analysis, a single, fixed value of D was estimated as the

fitting parameter in the life-cycle analysis based on the spawner/recruit data and the modeled in-river passage. The estimated value of D ranged from 0.03 to 0.52. In addition, a D value for fall chinook transported from McNary Dam was estimated from both a life-cycle analysis and transport-to-control studies at this dam. These values were considerably higher than the estimates for Snake River fall chinook obtained from the life-cycle model. McNary Dam D values ranged between 0.6 and 6, with a geometric mean of 1.7. In PATH, five sets were considered for the change D from the retrospective to the prospective periods (Table 7). In effect, these hypotheses explored the transportation effectiveness in the past and what might be obtained in the future.

Table 7: Five fall chinook hypotheses of D for the existing operations (Retrospective) future period (Prospective).

Scenario	Retrospective	Prospective
D1	0.05	0.24
D2	0.05	1.00
D3	0.05	0.05
D4	0.20	0.20
D5	1.00	1.00

4.4 Adult upstream migration survival (Sa)

The estimates of the number of fish lost during upstream migration are based on comparative dam counts recorded by species and age category, either jack or adults. In this formulation the upstream loss was corrected for in-river harvest, the loss from turnoff to other streams, and natural mortality. The conversion rates for full pool and drawdown conditions as used in PATH (Table 4.5-5 in Peters et al. 1999) are illustrated in Table 8 for A1, A2, A3 and B1. Increased survivals through John Day reservoir drawdown only were set at 5% for natural river as applied in Peters et al. (1999), and 1.5% for spillway crest and 1.3% for natural river with flood control.

Table 8: Conversion rate of adult migration survival (Sa) from Bonneville Dam tailrace to the spawning grounds (Marmorek et al. 1996, Peters et al. 1999).

	A1	A2	A3	B1	B2	B3	C1	C2	C3
Spring	0.67	0.67	0.77	0.81	0.79	0.80	0.70	0.69	0.69
Fall	0.42	0.42	0.83	0.87	0.87	NA	0.44	0.43	NA

5 Bayesian life-cycle analysis

To model the survival and recovery probabilities, a detailed life-cycle model adapted from the PATH analysis has been used. The methods of the PATH analysis are described in Marmorek, Peters and Parnell (eds.) (1998) and Peters Marmorek and Parnell (eds) (1999). The model uses the basic life-cycle dynamics expressed by eq (1) with a Ricker density dependence similar to eq (13). The PATH analysis was set up to explore the consequences of different assumptions on life stages, and in PATH two basic passage models were explored along with different assumptions on how life stages were connected and represented.

A retrospective analysis of spring and fall chinook was conducted in PATH using the historical spawner recruit and passage data to characterize detailed hypotheses on the life stages. In addition, in PATH a prospective analysis was developed to project the time evolution of stocks under differing assumptions about the effects of actions. Using a Bayesian analysis, the different hypotheses could be weighted with output of the probabilities of meeting survival and recovery goals.

Selected results from the PATH analysis are used in this report. Specifically, the retrospective analysis is used to characterize the life stage parameters for the deterministic analysis presented in this report. In addition, the prospective analysis has been applied to produce survival and recovery probabilities and equilibrium spawner levels under different weightings of the hypotheses.

The results of the life-cycle analyses depend on hypotheses used and the weightings applied to each hypothesis. In PATH a large number of hypotheses on life stage parameter values and functional forms of the linkages of ocean survival to the passage survival and the freshwater production life stage were evaluated. In this analysis a reduced set of the most influential hypotheses are included in evaluating survival and recovery probabilities and equilibrium population levels.

5.1 Actions evaluated for spring and fall chinook

The PATH Bayesian Simulation Model (BSM) was only used to evaluate action A1, A2, A3, and B1. In addition, for assessing probabilities of recovery over time, Actions A3 and B1 were evaluated under different delays of implementing the actions (Table 9).

Table 9: Actions evaluated with the PATH Bayesian model

Action	Description
A1:	Uses the existing transportation rules
A2:	Maximizes transportation using current system configuration
A3(3yr):	Drawdown of four Snake River dams (3-year delay)
A3(8yr):	Drawdown of four Snake River dams (8-year delay)
B1(10yr):	Drawdown of four Snake River dams (3-year delay) and drawdown of John Day Dam (10-year delay)
B1(15yr):	Drawdown of four Snake River dams (8-year delay) and drawdown of John Day Dam (15-year delay)

5.2 Hypotheses

The most important hypotheses concerned the survival of smolts through the hydrosystem and the survival of smolts after departing the hydrosystem. Because some smolts migrate through the river while others are collected at dams and transported, survival through both hydrosystem passage routes, and the associated survivals below the hydrosystem, must be considered. Scenarios to evaluate different factors controlling these hypotheses are listed in Table 10 for spring chinook and Table 11 for fall chinook.

5.2.1 Spring chinook hypotheses

The smolt passage models applied in the Bayesian life-cycle model are detailed in Section 4.2. Each passage model is grouped with an assumption on D that characterizes delayed mortality in transportation. CRiSP is paired with midrange D values and FLUSH is paired with low D values. In addition, model runs were conducted with the assumption that D was high. The values for D are described in Section 4.3.2. Three hypotheses on the source of the extra mortality were considered in the Bayesian analysis, the BKD, CLIMATE and HYDRO hypotheses. These are described in Section 4.3.1.

Two life-cycle models were considered in this analysis: the Alpha and Delta models, which differed primarily in the characterization of climatic/ocean change. The Delta model assumed that decadal scale climate/ocean changes in Snake River spring have the same pattern as observed in the mid- and lower Columbia spring chinook. The Alpha model characterized ocean variation through decadal climate indices, the PAPA drift index, and river flow at Astoria.

A lower level hypothesis in the modeling system involves the estimated time required to implement drawdown actions. This affects the success of the drawdown as a recovery action. In the analysis two periods were considered: 3 and 8 year delays for drawing down the four Snake River dams and 10 and 15 year delays for drawing down the four Snake River reservoirs plus the John Day reservoir. Assumptions were also included to characterize the amount of time before the drawdown reservoirs reach equilibrium in terms of the riverine habitat. Two periods were assumed: 2 and 10 years.

Table 10: PATH Hypotheses for spring chinook analysis

Model Group	Hypothesis	Weighting applied to the particular choice in the model group others in group have weight 0 unless otherwise noted.
	EQUAL	Equal weights on all hypotheses associated with particular action
Passage models	FLUSH	Weight of 1 on FLUSH passage/D-values.
	CRISP	Weight of 1 on CRISP passage/D-values.
	NMFS	Weight of 1 on CRISP passage with NMFS D values of 0.8
Extra mortality models	BKD	Weight of 1 on BKD extra mortality hypothesis.
	HYDRO	Weight of 1 on HYDRO extra mortality hypothesis.
	REGIME	Weight of 1 on REGIME shift extra mortality hypothesis.
Life- Cycle models	ALPHA	Weight of 1 on ALPHA life-cycle model.
	DELTA	Weight of 1 on DELTA cycle model.
Equilibrium Times	2 YEAR TRANSITION	Weight of 1 on 2-yr transition to reach equilibrium drawdown survival.
	10 YEAR TRANSITION	Weight of 1 on 10-yr transition to reach equilibrium drawdown survival.
Passage optimism	OPT.PASS	Weight of 1 on optimistic passage survival estimates.
	PESS.PASS	Weight of 1 on pessimistic passage survival estimates.

5.2.2 Hypotheses Evaluated for Fall Chinook

For fall chinook, hypotheses involved different harvest rates during recovery, different assumptions on the transportation effectiveness, D, two passage models, factors controlling the extra mortality, the length of the time required for the drawdown to reach equilibrium conditions, and upper and lower bounds on fall chinook smolt passage survival (Table 11).

A number of harvest-rate actions were considered, including one that increases harvest rates in the ocean by 15% as the stocks recover, and a number of actions that decrease the harvest rate in the ocean and in the river. The CRiSP and FLUSH fall chinook passage models were used in the analysis to define the in-river survival of fish. These models are defined in Section 4.2.1. Five values of D were evaluated. In three cases, the present day level of D was fit as a free parameter in the Bayesian life-cycle model. For projecting future stock levels in the prospective analysis, three different D hypotheses were applied. In the other two hypotheses, the D parameter in the

retrospective and prospective analyses were specified. The three extra mortality hypotheses were evaluated as in the spring chinook. It should be noted though that under low values of D, as are derived from the MLE estimation of D, the extra mortality is essentially zero. Only when D is large (~ 1) is an extra mortality factor required to account for the decline in the fall chinook. Two transition periods were evaluated for the time for each drawn-down reservoir to reach a functioning state that stabilizes survival. Finally, the models were run with combinations of juvenile passage survival representing low and high levels of survival.

Table 11: Hypotheses used in the fall chinook analysis

Hypothesis	Weights
EQUAL	All hypotheses weighted equally
Base(-/-)	Base ocean and in-river harvest
+15%/-	(% increase in ocean harvest/% increase in in-river harvest)
-15%/-	(% increase in ocean harvest/% increase in in-river harvest)
-50%/-	(% increase in ocean harvest/% increase in in-river harvest)
-75%/-	(% increase in ocean harvest/% increase in in-river harvest)
-50%/-50%	(% increase in ocean harvest/% increase in in-river harvest)
-75%/-50%	(% increase in ocean harvest/% increase in in-river harvest)
CRISP	CRISP passage model
FLUSH	FLUSH passage model
D1	Retro D value is MLE, prospective D=0.24
D2	Retro D values is MLE, prospective D=1.0
D3	Retro D value is MLE, prospective from posterior distribution
D4	Retro D=.2, prospective D=.2
D5	Retro D=1.0, prospective D=1.0
REGIME	Regime shift extra mortality hypothesis
BKD	BKD extra mortality hypothesis
HYDRO	HYDRO extra mortality hypothesis
2YR.TRANSITION	2-year transition to equilibrium juvenile survival under drawdown
10YR.TRANSITION	10-year transition to equilibrium juvenile survival under drawdown
LOW.EJUV	juvenile survival lower bound in Snake and John Day drawdown
HIGH.EJUV	juvenile survival upper bound in Snake and John Day drawdown

5.2.3 Weighting hypotheses

The Bayesian life-cycle model combined competing, and sometimes mutually exclusive, hypotheses giving probabilities of meeting recovery goals and levels of escapement under equilibrium conditions. To investigate the influence of particular hypotheses on a life stage component of survival, or for a set of hypotheses for several life stages, any of the hypotheses in the life-cycle model can be given different weights. For example, by giving the FLUSH model a weight of one and the CRiSP model a weight of zero the patterns of the stocks over a 48 year future is modeled under the FLUSH passage model hypothesis only. The neutral case gives equal weighting to each competing hypothesis. Decision analysis generally is not designed to give a single answer on the response of a stock to an action, such as drawdown. By combining all the hypotheses with, or without, equal weighting, it is designed to identify which actions are the most robust to the uncertainties in the models.

In the decision analysis approach, differing hypotheses were combined to evaluate actions. The numbers of hypotheses for the spring chinook BSM analysis are given in Table 12.

Table 12: Number of hypotheses under each action

Action	Total Hypotheses
A1	36
A2	36
A3	144
B1	144

5.3 BSM Results

5.3.1 Probability of Survival and Recovery

Probabilities of survival and recovery for spring and fall chinook under each of the major alternatives and under different weightings of the important hypotheses are detailed in Table 13 through Table 16. The results are based on the Bayesian life-cycle model. Also provided in Table 17 and 18 are the relative risk expressed as the change in probability in taking Action A3 instead of B1 and loss of probability in taking A1 instead of A3. Table 19 gives the equilibrium stock levels for the six Snake River index stocks under the different actions and under different weightings of the hypotheses.

Table 13: Spring chinook 24-year survival probability mean values.

Hypothesis weighting	A1	A2	A3(3yr)	A3(8yr)	B1(10yr)	B1(15yr)
EQUAL	0.673	0.668	0.725	0.698	0.723	0.702
FLUSH	0.582	0.563	0.689	0.632	0.685	0.640
CRISP	0.730	0.732	0.760	0.741	0.755	0.746
NMFS	0.707	0.709	0.725	0.720	0.728	0.721
BKD	0.607	0.606	0.674	0.642	0.670	0.644
HYD	0.758	0.746	0.794	0.773	0.794	0.780
REGIME	0.654	0.653	0.705	0.677	0.704	0.683
ALPHA	0.647	0.647	0.712	0.682	0.706	0.684
DELTA	0.699	0.690	0.737	0.713	0.740	0.720
2YR.TRANSITION	0.673	0.668	0.739	0.708	0.739	0.711
10YR.TRANSITION	0.673	0.668	0.710	0.687	0.706	0.694
OPT.PASS	0.693	0.693	0.751	0.727	0.749	0.728
PESS.PASS	0.654	0.644	0.698	0.668	0.697	0.677

Table 14: Spring chinook 48-year recovery probability mean values.

Hypothesis weighting	A1	A2	A3(3yr)	A3(8yr)	B1(10yr)	B1(15yr)
EQUAL	0.533	0.514	0.787	0.795	0.822	0.811
FLUSH	0.368	0.328	0.863	0.880	0.911	0.897
CRISP	0.615	0.614	0.789	0.786	0.807	0.815
NMFS	0.616	0.599	0.708	0.719	0.749	0.721
BKD	0.375	0.364	0.668	0.677	0.709	0.699
HYD	0.624	0.594	0.858	0.864	0.895	0.884
REGIME	0.600	0.582	0.834	0.845	0.863	0.850
ALPHA	0.468	0.451	0.728	0.747	0.779	0.760
DELTA	0.598	0.577	0.846	0.843	0.865	0.863
2YR.TRANSITION	0.533	0.514	0.790	0.793	0.822	0.812
10YR.TRANSITION	0.533	0.514	0.784	0.798	0.822	0.811
OPT.PASS	0.568	0.552	0.834	0.844	0.862	0.861
PESS.PASS	0.498	0.475	0.739	0.747	0.783	0.762

Table 15: Fall chinook probability of meeting a 24-year survival standard.

Hypothesis weighting	A2	A3(3yr)	A3(8yr)	B1(10yr)	B1(15yr)
EQUAL	0.940	0.955	0.946	0.952	0.949
Base(-/-)	0.924	0.948	0.935	0.944	0.939
+15%/-	0.918	0.947	0.932	0.942	0.937
-15%/-	0.929	0.950	0.938	0.946	0.942
-50%/-	0.941	0.954	0.946	0.951	0.948
-75%/-	0.946	0.955	0.949	0.952	0.950
-50%/-50%	0.961	0.966	0.963	0.965	0.964
-75%/-50%	0.964	0.967	0.965	0.966	0.965
CRISP	0.915	0.944	0.930	0.940	0.935
FLUSH	0.964	0.966	0.963	0.965	0.964
D1	0.993	0.991	0.993	0.991	0.992
D2	0.999	0.997	0.997	0.998	0.998
D3	0.830	0.918	0.885	0.907	0.895
D4	0.922	0.942	0.929	0.938	0.933
D5	0.956	0.955	0.952	0.953	0.952
REGIME	0.945	0.961	0.952	0.958	0.955
BKD	0.938	0.958	0.949	0.955	0.952
HYDRO	0.938	0.963	0.952	0.959	0.955
2YR.TRANSITION	0.940	0.956	0.948	0.953	0.950
10YR.TRANSITION	0.940	0.954	0.945	0.951	0.949
LOW.EJUV	0.940	0.952	0.943	0.948	0.945
HIGH.EJUV	0.940	0.958	0.950	0.956	0.953

Table 16: Fall chinook probability of meeting the 48-yr recover standard.

Hypothesis weighting	A2	A3(3yr)	A3(8yr)	B1(10yr)	B1(15yr)
EQUAL	0.643	0.955	0.954	0.969	0.969
Base(-/-)	0.578	0.945	0.944	0.961	0.961
+15%/-	0.555	0.94	0.939	0.958	0.958
-15%/-	0.601	0.949	0.948	0.964	0.964
-50%/-	0.672	0.96	0.959	0.973	0.973
-75%/-	0.698	0.964	0.963	0.976	0.976
-50%/-50%	0.723	0.969	0.968	0.979	0.979
-75%/-50%	0.749	0.972	0.971	0.982	0.982
CRiSP	0.557	0.956	0.956	0.967	0.967
FLUSH	0.729	0.954	0.952	0.971	0.971
D1	0.917	1	1	1	1
D2	1	1	1	1	1
D3	0.351	1	1	1	1
D4	0.437	0.999	0.999	0.999	0.999
D5	0.737	0.867	0.864	0.908	0.909
REG	0.735	0.984	0.984	0.991	0.991
BKD	0.665	0.936	0.934	0.954	0.954
HYD	0.665	1	1	1	1
2YR.TRANS	0.643	0.955	0.956	0.97	0.97
10YR.TRANS	0.643	0.955	0.953	0.969	0.969
LOW.EJUV	0.643	0.938	0.937	0.955	0.955
HIGH.EJUV	0.643	0.973	0.972	0.984	0.984

5.3.2 Relative risks

Relative risks are the difference in probabilities of survival and recovery between two action for a given set of weightings of the hypotheses. A positive value indicates the probability of meeting a measure increases by the amount. Relative risk for a Snake River drawdown compared to a Snake River drawdown with a John Day drawdown to natural river level is defined by the difference in the probabilities for A3 and A1, i.e. B1 - A3. The relative risk of Snake River drawdown compared to transportation is defined by the difference in probabilities for A3 and A1, i.e. A3 - A1. Table (17) gives relative risk for spring chinook and Table (18) gives relative risk for fall chinook.

Table 17: Relative risk for spring chinook. B1-A3 is gain in probability of meeting standard due to taking action B1 over action A3. A3-A1 is gain in probability of meeting standard due to taking action A3 over action A1.

	24-Year Survival		**48-Year Recovery**	
Hypothesis weighting	**B1- A3**	**A3- A1**	**B1- A3**	**A3- A1**
EQUAL	0.0010	0.0385	0.0255	0.2580
FLUSH	0.0020	0.0785	0.0325	0.5035
CRISP	0	0.0205	0.0235	0.1725
NMFS	0.0020	0.0155	0.0215	0.0975
BKD	-0.0010	0.0510	0.0315	0.2975
HYD	0.0035	0.0255	0.0285	0.2370
REGIME	0.0025	0.0370	0.0170	0.2395
ALPHA	-0.0020	0.0500	0.0320	0.2695
DELTA	0.0050	0.0260	0.0195	0.2465
2YR.TRANSITION	0.0015	0.0505	0.0255	0.2585
10YR.TRANSITION	0.0015	0.0255	0.0255	0.2580
OPT.PASS	-0.0005	0.0460	0.0225	0.2710
PESS.PASS	0.0040	0.0290	0.0295	0.2450

Table 18: Relative risk for fall chinook. B1-A3 is gain in probability of meeting standard due to taking action B1 over action A3. A3-A2 is gain in probability of meeting standard due to taking action A3 over Action A2.

	24 year survival		48 year recovery	
Hypothesis weighting	**B1-A3**	**A3-A2**	**B1-A3**	**A3-A2**
EQUAL	0	0.0105	0.0145	0.3115
Base(-/-)	0	0.0175	0.0165	0.3665
+15%/-	0	0.0215	0.0185	0.3845
-15%/-	0	0.0150	0.0155	0.3475
-50%/-	-0.0005	0.0090	0.0135	0.2875
-75%/-	-0.001	0.0060	0.0125	0.2655
-50%/-50%	0	0.0035	0.0105	0.2455
-75%/-50%	-0.0005	0.0020	0.0105	0.2225
CRiSP	0.0005	0.0220	0.0110	0.3990
FLUSH	0	0.0005	0.0180	0.2240
D1	-0.0005	-0.0010	0	0.0830
D2	0.001	-0.0020	0	0
D3	-0.0005	0.0715	0	0.6490
D4	0	0.0135	0	0.5620
D5	-0.001	-0.0025	0.0430	0.1285
REG	0	0.0115	0.0070	0.2490
BKD	0	0.0155	0.0190	0.2700
HYD	-0.0005	0.0195	0	0.3350
2YR.TRANS	-0.0005	0.0120	0.0145	0.3125
10YR.TRANS	0.0005	0.0095	0.0150	0.3110
LOW.EJUV	-0.001	0.0075	0.0175	0.2945
HIGH.EJUV	0.0005	0.014	0.0115	0.3295

Equilibrium stock levels

Equilibrium stock levels are the population numbers that stocks at equilibrium. The equilibrium levels are estimated from eq(14) in Section (6.2). For each action, weighting parameters are selected, and the equilibrium levels are determined as the geometric means of all alternative hypotheses. The spring chinook equilibrium levels in Table (19) are given for each ESU index stock for each action and each hypothesis-weighting scheme. Also given are the gains in equilibrium numbers for action B1 relative to action A3 and for action A3 relative to action A1.

The fall chinook equilibrium stock level under different actions and hypotheses are given in Table (20) along with the difference in populations level of action B1 relative to A3 and action A3 relative to A2.

Table 19: Spring chinook equilibrium stock levels for the index Snake River ESU stocks under different actions and hypotheses weightings. B1- A3 and A3-A1 are the relative gains in numbers of spawners between two actions.

Imnaha

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	1127	1115	1694	1882	188	567
FLUSH	991	954	2249	2587	338	1258
CRISP	1346	1344	1681	1793	112	335
NMFS	1046	1048	1152	1264	112	106
OPT.PASS	1172	1156	1890	2092	202	718
PESS.PASS	1083	1074	1499	1671	172	416

Minam

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	687	681	1003	1108	105	316
FLUSH	612	590	1313	1502	189	701
CRISP	809	808	996	1058	62	187
NMFS	642	644	702	764	62	60
OPT.PASS	712	704	1112	1225	113	400
PESS.PASS	663	658	895	991	96	232

Bear Valley

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	1288	1274	1942	2159	217	654
FLUSH	1131	1087	2583	2974	391	1452
CRISP	1540	1538	1927	2056	129	387
NMFS	1193	1196	1318	1446	128	125
OPT.PASS	1339	1321	2168	2402	234	829
PESS.PASS	1237	1227	1717	1916	199	480

March Creek

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	681	672	1060	1185	125	379
FLUSH	590	564	1430	1656	226	840
CRISP	826	826	1051	1126	75	225
NMFS	626	628	698	772	74	72
OPT.PASS	711	700	1191	1325	134	480
PESS.PASS	651	645	929	1044	115	278

Sulphur CR

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	486	481	678	741	63	192
FLUSH	440	426	866	980	114	426
CRISP	560	559	673	711	38	113
NMFS	458	458	494	532	38	36
OPT.PASS	501	495	744	813	69	243
PESS.PASS	470	468	612	670	58	142

Poverty Flat

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	1001	989	1541	1719	178	540
FLUSH	871	835	2070	2391	321	1199
CRISP	1208	1207	1528	1635	107	320
NMFS	922	925	1025	1132	107	103
OPT.PASS	1043	1028	1727	1920	193	684
PESS.PASS	958	950	1355	1519	164	397

Johnson Cr.

Hypotheses	A1	A2	A3	B1	B1- A3	A3- A1
EQUAL	367	363	560	623	63	193
FLUSH	320	308	748	862	114	428
CRISP	441	440	555	593	38	114
NMFS	339	340	376	414	38	37
OPT.PASS	382	377	626	694	68	244
PESS.PASS	352	349	493	552	59	141

Table 20: Fall chinook equilibrium stock level for the index Snake River ESU stocks under different actions and hypotheses weightings. B1- A3 and A3-A2 are the relative gains in spawners for between two actions.

Weighting	A2	A3 3yr	A3 8yr	B1 (10yr)	B1 (15yr)	B1-A3	A3-A2
EQUAL	5396	15751	15326	17875	17677	2238	10143
Base(-/-)	5021	15338	14913	17440	17241	2215	10105
+15%/-	4876	15149	14724	17244	17049	2210	10061
-15%/-	5160	15495	15067	17601	17404	2222	10121
-50%/-	5628	16190	15750	18338	18137	2268	10342
-75%/-	5764	16210	15781	18365	18165	2270	10232
-50%/-50%	5848	16287	15862	18437	18240	2264	10227
-75%/-50%	5976	16303	15889	18460	18263	2266	10120
CRISP	4733	14938	14520	16319	16145	1503	9996
FLUSH	6058	16563	16132	19430	19209	2972	10290
D1	5377	22012	21425	24641	24330	2767	16342
D2	15577	23201	23077	25984	25822	2764	7562
D3	2448	21638	20897	24194	23814	2737	18820
D4	2718	16093	15522	18221	17982	2294	13090
D5	5668	8875	8655	10463	10394	1664	3097
REGIME	6856	18543	18124	20916	20691	2470	11478
BKD	6108	17192	16773	19390	19173	2299	10875
HYDRO	6108	19356	18848	21796	21540	2566	12994
2YR.TRAN	5396	15879	15472	18184	17963	2398	10280
10YR.TRAN	5396	15622	15179	17565	17391	2078	10005
LOW.EJUV	5396	14647	14293	16719	16521	2150	9074
HIGH.EJUV	5396	16854	16358	19030	18833	2326	11210

6 Deterministic Life-cycle analysis

As an alternative to the BSM model analysis, a deterministic life-cycle model was developed to compare the difference in performance measures between two actions. The measures are the difference in equilibrium stock levels, the difference between maximum sustainable yield (MSY) and the difference in MSY plus escapement. This analysis provides a relative comparison of the merits of one action relative to another. It assumes no environmental or intrinsic variability and as such the estimates of productivity and mortality represent long term time averages. In this approach the environmental variations common to the pair of actions do not have to be considered.

The deterministic model is based on the life-cycle model given by eq (1). First, number of recruits to the spawning grounds (R) per spawner (S) is defined from eq (1) as

eq (11) $$R/S = P \cdot Sm \cdot So \cdot Sa \cdot H$$

where H is the combined ocean and in-river harvest rate. At equilibrium, the number of spawners equals the number of recruits so R = S, and

eq (12) $$1/P^* = Sm \cdot So \cdot Sa \cdot H$$

where P* is the freshwater productivity at equilibrium conditions. Applying a Ricker type density dependence to the freshwater production term P* yields

eq (13) $$P^* = \exp(a_{fr} - b \cdot S^*)$$

where S* is the equilibrium spawner level and a_{fr} is the productivity component of the freshwater life stage. Then the equilibrium stock level in terms of spawners is

eq (14) $$S^* = (a_{fr} + \log(Sm) + \log(So) + \log(Sa) + \log(H))/b$$

The ratio of equilibrium freshwater productivity of two actions can be expressed

eq (15) $$G_{x,y} = P^*_y / P^*_x = (Sm_x \cdot So_x \cdot Sa_x \cdot H_x)/(Sm_y \cdot So_y \cdot Sa_y \cdot H_y)$$

where the subscripts refer to alternatives x and y. Applying Ricker type density dependence to the freshwater production term from eq (13) the difference in population numbers between two alternatives is

eq (16) $$S^*_x - (b_y/b_x)S^*_y = (1/b_x)\ \ln(G_{x,y}) + (a_x - a_y)/b_x$$

Since in the hydrosystem passage corridor actions x and y are not expected to directly alter the spawning habitat productivity rate parameters, $a_x = a_y$ and $b_x = b_y$, so at population equilibrium the difference in population numbers for actions x and y is

eq (17) $$E_{x,y} = S^*_x - S^*_y = (1/b)\ \ln(G_{x,y})$$

The b factor is the Ricker density-dependent term, which describes how quickly the stock productivity decreases as S increases. It can be estimated from stock recruitment data and is insensitive to assumption to the hypotheses discussed above.

The impact of actions on the maximum sustainable yield (MSY) from a stock, $M_{x,}$ under actions x can be estimated from the intrinsic Ricker a and b parameters and the G factor given by eq (15). No explicit function can be defined for MSY for a Ricker stock recruitment function, but M, and the difference in MSY between actions,

eq (18) $$\Delta M = M_x - M_y,$$

can be obtained numerically. Following Ricker (1978) the MSY spawning level at is obtained from the explicit equation

eq (19) $$(1-b_x \cdot S_{msy,\,x})\exp(a_y - b \cdot S_{msy,\,x}) = 1$$

The MSY for an action x is then obtained with $S_{msy,\,x}$ in

eq (20) $$M_x = S_{msy,\,x}\,(\exp(a_x - b_x \cdot S_{msy,\,x}) - 1)$$

where a_x is the density-independent productivity term under action x. The total spawning population under the MSY conditions is

eq (21) $$R = M + S_{msy}$$

The difference in R between two actions is

eq (22) $$\Delta R = R_x - R_y$$

These MSY measures can be related to G by the equation

eq (23) $$a_x = a_0 - \log G_{0,x}$$

where a_0 is the base, or current conditions, Ricker at term referenced to the spawning grounds and where the Ricker term b_x may or may not change between the two actions. The same equation can be used to identify M_y with the substitution of parameters for action y. For the estimation of MSY and recruitment at MSY the results are referenced to the spawning population. Therefore, since the Ricker density independent parameter, a, given in Table 2 is referenced to the Bonneville dam, it is related to the spawning ground term as

eq (24) $$a_0 = a + \log(Sa_{A1})$$

where Sa_{A1} is the current upriver survival of adults.

Because the MSY estimates depend on the Ricker a parameter as references to the base level according to eq(23) assumptions on the distribution of factors in the base level will affect the pairwise comparison between actions. Therefore, pairwise comparisons between actions not involving transportation, such as A3 and B1, will depend on the assumption made on transportation. This complication does not affect the comparisons of equilibrium levels though.

The ratio of return adults to the number of smolt outmigration (SAR) is an important measure of the performance of the stocks. The ratio of the SAR of one action to another is in turn a relative measure of the performance of two actions. This ratio of ratios can be expressed through G_{xy}. Several measures can be defined depending on the collection point for the adults. The smolt population is defined as the point at which the smolts enter the hydrosystem. Adults can be defined in terms of the return to collection in the fisheries or return to the spawning ground or hatchery. For return to the fisheries we define

eq (25) $$SAR_x/SAR_y = (Sm_x \cdot So_x)/(Sm_y \cdot So_y)$$

For return to the spawning grounds or hatcheries the ratio is defined

eq (26) $$SAR_x/SAR_y = G_{x,y}$$

The difference in equilibrium population and maximum sustainable yields can be expressed in terms of the ratio of life stage survivals, G. The estimates of the life stage terms involve hypotheses on how the stages are coupled and the life stage parameter values under the various actions. In this analysis a wider range of actions are evaluated (see Table 1). Expressing differences in equilibrium population measures in this simple analytical form illustrates the significance of the hypotheses that come into play to produce a specific result.

6.1 Deterministic Life Stage Parameters

The life stage parameters used in the deterministic analysis are given in the Tables below. Four stocks from subbasins are considered. Parameters for a representative Snake River spring chinook are given in Table 21 and Snake River fall chinook parameters are given in Table 22. Parameters for the Upper Columbia are given in Table 23 for Upper Columbia spring chinook and the Hanford Reach fall chinook in Table 24. Most parameters were obtained from the PATH analysis and are described in Section 4. Details for the Upper Columbia chinook, which were not covered in PATH analysis, are discussed below.

The survivals of spring chinook smolts (*Sm*) were taken from PATH analysis as the means of passage models. Specifically the means were taken of CRiSP with TURB 4 hypothesis and FLUSH TURB 5 hypothesis (The results were compiled by NMFS in a spread sheet designated, all98.xls). By taking the means of the model results a middle point estimate of direct hydrosystem survival was obtained. In general, the CRiSP inriver passage survival estimates were higher than the FLUSH estimates by about a factor of two. The mean CRiSP inriver survival was about 0.44, while the FLUSH estimate was about 0.2 giving a mean of 0.32. Although these estimates are significantly different, the direct survival estimates of transported plus inriver passing fish are similar for the two models because most of the fish are transported and the two models have approximately the same transport percentages and used the same direct transport survival of 0.98.

To estimate the effect of John Day drawdown under natural river, spillway crest and natural river with flood control the CRiSP 1.6 passage model was used (See Table 44). These CRiSP derived estimates for the C and D actions were then adjusted to approximate the mean of *Sm* by adjusting the CRiSP A1 base in river survival, designated the A1 inriver survival in Table 44) according to the mean *Sm* of the A1 inriver survival estimate.

To estimate the smolt migration survival, *Sm*, for the Hanford Reach upriver bright fall chinook, first the survival per kilometer of inriver passing fish was estimated from recent PIT tag studies of fall chinook from McNary Dam to John Day Dam. The survival from MCN tailrace to BON

tailrace was then expressed over the travel distance plus accounting for passage at three dams. The equation for in-river passage survival is then

eq (27) $$V_{mcn} = (\text{dam passage survival})^3 \, R^{145.6}$$

where R = 0.99309 is the reach survival per km and was estimated from the MCN to JDA reach as the geometric mean of 0.53 over 76.4 km length (Smith and Achord, 1999). Using a dam passage survival of 0.9 and a Hanford Reach (from Ringold Hatchery) to BON migration distance of 146 km the in-river survival is $V_{mcn} = 0.37$. The PIT tag determined geometric mean of survival from Hanford Reach to MCN dam tailrace was 0.72. Assuming that the collection transportation survival is 1, the direct smolt passage survival can be approximated as

eq (28) $$Sm = V_{mcn} (1 - FGE) + FGE$$

where FGE is the fish guidance efficiency at MCN Dam. We use a representative historical (Action A1) FGE (24%) for the base period from 1974 to 1995, which is the period over which the parameters a and b are estimated in Schaller et al. (1999). For prospective analysis (Action A2) we used the recent estimates of FGE = 68% (Peters et al. 1999).

The D factor for transport from McNary Dam is set at 1, which is the estimate obtained from two methods: 1) T/C ratios and in-river survival and 2) from a life-cycle analysis (Peters et al. 1999). To estimate the impacts of John Day drawdown we assume the drawdown survival in JDA is 0.95 and note that smolts cross 3 dams. Upstream conversion rates are taken from estimates for the stock provided in Table 3.1.2-3 in Marmorek, Peters and Parnell (1998).

For upper Columbia spring chinook life-cycle parameter estimates, we used the Snake River spring chinook estimates with changes reflecting the fact that these fish are not affected by the Snake River drawdown.

Table 21: Snake River spring chinook survival information for the calculations.

	A0	A1	A2	A3	B1	B2	B3	C1	C2	C3	D1	D2	D3	comments
Sa	0.67	0.67	0.67	0.77	0.81	0.79	0.80	0.70	0.68	0.69	0.70	0.68	0.69	
Sa_{low}	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	0.67	lower estimate
Sm	0.34	0.81	0.84	0.61	0.64	0.60	0.63	0.363	0.342	0.358	0.84	0.84	0.84	
λn_c	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	0.40	bkd /climate
λn_h	0.40	0.40	0.40	0.70	0.70	0.70	0.70	0.40	0.400	0.40	0.40	0.40	0.40	hydro
Pb	0.00	0.93	0.96	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.96	0.96	0.96	

Table 22: Snake River fall chinook survival information for the calculations.

	A0	A1	A2	A3	B1	B2	C1	C2	D1	D2	comments
Sa	0.42	0.42	0.42	0.83	0.87	0.87	0.44	0.43	0.43	0.43	
Sm_{high}	0.28	0.78	0.78	0.51	0.58	0.56	0.31	0.30	0.78	0.78	high est.
Sm_{low}	0.28	0.78	0.78	0.35	0.40	0.39	0.31	0.30	0.78	0.78	low est.
Sm_{mean}	0.28	0.78	0.78	0.44	0.50	0.49	0.31	0.30	0.78	0.78	mean est.
λn	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	
Pb	0.00	0.97	0.97	0.00	0.00	0.00	0.00	0.00	0.97	0.97	

Table 23: Upper Columbia spring chinook survival information for calculations.

	A0	A1	A2	A3	B1	B2	B3	C1/D1	C2/D2	C3/D3
Sa	0.67	0.67	0.67	0.67	0.70	0.68	0.69	0.70	0.68	0.69
Sm	0.34	0.34	0.34	0.34	0.36	0.34	0.36	0.39	0.37	0.38
λn	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Pb	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table 24: Hanford Reach fall chinook survival information for calculations.

	A0	A1	A2	A3	B1	B2	C1	C2	D1/D2
Sa	0.75	0.75	0.75	0.75	0.79	0.79	0.79	0.79	0.79
Sm	0.31	0.31	0.54	0.19	0.21	0.20	0.21	0.20	0.54
λn	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
Pb	0.00	0.55	0.89	0.00	0.00	0.00	0.00	0.00	0.89

6.2 Equations for comparison of actions

6.2.1 Equilibrium Population and MSY Differences in Drawdown Actions

The pairwise difference between actions x and y equilibrium populations, $E_{x,y}$, described by eq (17) and the maximum sustained yield $\Delta M_{x,y}$ using eq (19), eq (20) and eq (23), both use $G_{x,y}$ from eq (15). For John Day drawdown actions relative to Snake River drawdown the growth ratio equation reduces to

eq (29) $$G_{x,A3} = (Sa_x/Sa_{A3})(Sm_x/Sm_{A3})$$

where the actions are x = B1, B2, B3, C1, C2, C3 as detailed in Table 1, and the adult and smolt migration survival values are given in Table 21 and 22. In these calculations the ocean survival factors are assumed equal under all alternatives. That is, the effects of hydrosystem passage under Snake River drawdown, (A3), the Snake and John Day (B1, B2, B3) and a John Day drawdown only (C1, C2, C3) are set equal, so the ratio of post-Bonneville survivals, $\lambda n_x/\lambda n_y$, is always 1.

6.2.2 Equilibrium and MSY Differences of Transport vs. Drawdown Actions

For comparing transportation and drawdown actions the population growth index for spring chinook takes the form

eq (30) $$G_{A1,y} = [Sa_{A1}/Sa_y][(D_{A1}Pb_{A1}+1-Pb_{A1})][\lambda n_{A1}/\lambda n_y][Sm_{A1}/Sm_y]$$

The difference in equilibrium populations between A1 and alternative y is estimated by using $G_{A1,y}$ in eq (17) and the difference in MSY is estimated with eq (19), eq (20) and eq (23).

The for comparing transportation to drawdown actions population growth index for Snake River fall chinook is calculated by the equation

eq (31) $$G_{A1,y} = [Sa_{A1}/Sa_y][(D_{A1}\cdot Pb_{A1}+1-Pb_{A1})][Sm_{A1}/Sm_y].$$

Here the ocean survival term under actions A1 and y are assumed equal. This result comes from the PATH fall chinook retrospective analysis, which indicated that under low levels of D that extra mortality was negligible. That is, under an assumption of low D, the decline in the fall chinook is principally the result of their transportation. The result does not hold if D is high though. Then an extra mortality term is required to account for the decline in the fall chinook. In this case eq (31) is still appropriate under CLIMATE and BKD hypotheses. The description of the extra mortality for fall chinook, thus, has not been resolved. This issue is not critical when evaluating the impact of the addition of a John Day drawdown, since ocean survival under this action is not expected to be significantly different from ocean survival under a drawdown of both the Snake River system and John Day.

6.2.3 Equivalence points

An important consideration is the mix of life-stage survivals that make drawdown and transportation actions equivalent in terms of the equilibrium population levels. Note that the different drawdown actions are always different in that additional drawdown produces higher survival under the assumptions of this analysis. The equivalence point at which A2 and A3 given equal population levels can be expressed by setting $G_{A2,A3} = 1$. The resulting equivalence point between transportation and drawdown for Snake River spring chinook is

eq (32) $$Sa_{A3} \cdot Sm_{A3} = D_{A2} \cdot \lambda n_{A2} / \lambda n_y / 2$$

where the factor 1/2 is approximately the product of smolt and adult upstream survivals in the existing system. The Snake River fall chinook equivalence point between transportation and drawdown actions is

eq (33) $$Sa_{A3} \cdot Sm_{A3} \sim D_{A2}/3$$

where the factor 1/3 is the approximate product of the smolt and adult river migration survival under the existing passage conditions.

6.3 Results

6.3.1 Pairwise comparison of drawdown alternatives

The typical equilibrium population of an index stock under action A1 is 419 for spring chinook and 7259 for fall chinook. The resulting difference in spawners at population equilibrium and MSY for a pairwise comparison of drawdown alternatives is given in Tables 25 and 26 for Snake River spring chinook and in Tables 27 and 28 for Snake River fall chinook. For example, comparing action A3 to action B1 in Table 25, at equilibrium, action B1 results in 58 more spawners than action A3. These estimates are based on b given in Table (2). The number of spring chinook stocks in the Snake River Basin is on the order of 38 making a total equilibrium population of about 16000. Thus we expect a 58x38 = 2204 fish increase in the Snake River Basin index spawning population by including the John Day drawdown with a Snake River drawdown action (B1). For fall chinook, the equilibrium population size in the Snake River basin is about 7000 making the maximum benefit of a John Day action in addition to a Snake River drawdown of about 650 spawners. For the fall chinook analysis, the effect of uncertainties in the smolt survival with Snake River drawdown are illustrated by providing equilibrium levels and MSY for high and low smolt survivals (Tables 27 and 28).

The ΔM estimates in Table 26 depends on the assumption on D. To a good approximation the relationship is linear with ΔM decreasing as D increases. For a comparison of A3 to B1 the influence in D can be expressed by the equation

eq (34) $$\Delta M_{A3,B1} = 157 - 147\ D$$

Table 25: Spring chinook equilibrium population differences $E_{row,col}$. Equilibrium number of spring chinook under A1 is 420 fish per stock.

	B1	B2	B3	C1	C2	C3
A3	56	5	40	-354	-404	-370
B1	0	-52	-17	-410	-461	-427
B2		0	35	-359	-410	-375
B3			0	-394	-445	-410
C1				0	-51	-17
C2					0	34

Table 26: Spring chinook MSY difference population differences $\Delta M_{row,col}$). MSY for spring chinook under A1 is 93 fish per stock.

	B1	B2	B3	C1	C2	C3
A3	108	10	76	-448	-488	-461
B1	0	-98	-32	-556	-596	-569
B2		0	66	-458	-498	-471
B3			0	-524	-564	-537
C1				0	-40	-13
C2					0	27

Table 26a: Snake River spring chinook $G_{row,col}$.

	B1	B2	B3	C1	C2	C3
A3	1.1	1.00	1.07	0.54	0.49	0.52
B1	1.0	0.91	0.97	0.49	0.44	0.47
B2		1.00	1.06	0.53	0.49	0.52
B3			1.00	0.50	0.46	0.49
C1				1.00	0.91	0.97
C2					1.00	1.06

Table 27: Snake river fall chinook equilibrium population differences $E_{row,col}$ under high (0.9) and low (0.6) estimates of free-flowing river smolt survival Vl. Equilibrium number of fall chinook under A1 is 7259 fish.

	Vl_{A3}	B1	B2	C1	C2
A3	0.9	650	520	-4195	-4401
B1	0.9		-130	-4846	-5052
B2	0.9			-4716	-4922
C1	0.9				-207
A3	0.6	668	575	-2801	-3007
B1	0.6		-94	-3469	-3676
B2	0.6			-3376	-3582
C1	0.6				-207

Table 28: Snake River fall chinook differences in MSY population $\Delta M_{row,col}$ under high (0.9) and low (0.6) estimates of free-flowing river smolt survival Vl. MSY for fall chinook under A1 is 6548 fish.

	Vl_{A3}	B1	B2	C1	C2
A3	0.9	12828	10082	-45136	-46290
B1	0.9		-2746	-57964	-59118
B2	0.9			-55218	-56372
C1	0.9				-1154
A3	0.6	9061	7690	-24199	-25353
B1	0.6		-1371	-33260	-34414
B2	0.6			-31889	-33043
C1	0.6				-1154

Table 28a: Snake River fall chinook $G_{row,col}$ with Sm set as the average of high and low estimates from Table 22.

	B1	B2	C1	C2
A3	1.19	1.16	0.37	0.35
B1	1.00	0.98	0.31	0.29
B2		1.00	0.31	0.30
C1			1.00	0.94

6.3.2 Pairwise comparison of transport vs. drawdown alternatives

Differences in equilibrium and MSY populations for Snake River and Upper Columbia stocks, for drawdown actions relative to the current operations conditions, A1, are illustrated in Tables 29 through 37. Positive values indicate the alternative action population is above the level from A1 population. Upper and lower estimates of the population level are illustrated for different assumptions on ocean survival and adult migration conversion rates. Also illustrated are the impacts of the three levels of D for spring chinook (see Table 6 for description) and fall chinook (see Table 7 for description).

For a Snake River spring chinook index stock, G is defined by eq (30). The life-cycle parameter values are given in Table 21, and the results are given in Tables 29, 30, 31, 31a and 33. The greatest increase above A1 occurs for B1 (828 spawners for E) under a low transportation efficiency in A1 (D = 0.35), and in changing from A1 to B1, the ocean survival increases by the factor 1.75 and adult upstream conversion (survival) increases by 1.15 (Table 29). If upstream conversion and ocean survival do not increase between A1 and B1 and transportation is effective, as expressed by D = 0.8, then changing from A1 to B1 causes a decreases in the equilibrium level by 18 spawners under equilibrium E. In Table 32 the effect of C actions are not computed for an increase in ocean survival because a drawdown of John Day reservoir without drawing down the Snake River reservoirs is not expected to increase ocean survival.

For a Snake River fall chinook index stock G is defined by eq (31). The life-cycle parameter values are given in Table 22. Pairwise comparisons of actions are given in Tables 31a, 33a, 34 and 34a, 35, 36 and 37. Comparing the effect of transportation, A1, with putting fish back in the river with no transportation, action A0, is illustrated in Table 33a. A0 is a preferred option except when transportation effectiveness is high (D = 1). Also, improving transportation, represented by D_{A2}, also improves fish stocks. Here, only the D hypotheses are a major determinant of the difference between A2 and A0. The balance between D and the passage survivals is illustrated in eq (33), showing the equivalence point where drawdown and

transportation are equal. Table 33a, 34 and 34a show the effects of D hypotheses and the high and low passage survivals on pairwise comparisons of fall chinook.

The Hanford reach fall chinook G is estimated with eq (29) using the life-cycle survival data in Table 24. The resulting differences in equilibrium and MSY levels from comparing different alternatives are illustrated in Tables 39, 40 and 41.

The Upper Columbia spring chinook G is estimated with eq (29) using the life-cycle survival data in Table 23. The resulting differences in equilibrium and MSY levels from comparing different alternatives are illustrated in Tables 42 and 43.

Table 29: Snake River index spring chinook stock pairwise difference in equilibrium population. Equilibrium level under A1 is 420 spawners per index stock.

D	$\lambda n_y / \lambda n_{A1}$	Sa_{A3} / Sa_{A1}	$E_{A1,A2}$	$E_{A1,A3}$	$E_{A1,B1}$	$E_{A1,B2}$	$E_{A1,B3}$	$E_{A1,D1}$	$E_{A1,D2}$	$E_{A1,D3}$
0.35	1.75	1.15	-9	771	828	776	812	17	0	8
0.65	1.75	1.15	11	464	521	470	505	37	20	28
0.80	1.75	1.15	16	356	413	362	397	41	25	33
0.35	1	1	-9	370	397	360	388	-9	-9	-9
0.65	1	1	11	63	90	53	81	11	11	11
0.80	1	1	16	-45	-18	-55	-27	16	16	16

Table 30: Snake River index spring chinook stock G for pairwise comparisons under different D, extra mortality and upstream conversion rates.

D	$\lambda n_y / \lambda n_{A1}$	Sa_{A3} / Sa_{A1}	$G_{A1,A2}$	$G_{A1,A3}$	$G_{A1,B1}$	$G_{A1,B2}$	$G_{A1,B3}$	$G_{A1,D1}$	$G_{A1,D2}$	$G_{A1,D3}$
0.35	1.75	1.15	0.98	3.82	4.22	3.86	4.10	1.03	1.00	1.01
0.65	1.75	1.15	1.02	2.24	2.47	2.26	2.40	1.06	1.03	1.05
0.80	1.75	1.15	1.02	1.86	2.05	1.87	1.99	1.07	1.04	1.06
0.35	1	1	0.98	1.90	1.99	1.87	1.96	0.98	0.98	0.98
0.65	1	1	1.02	1.11	1.17	1.09	1.15	1.02	1.02	1.02
0.80	1	1	1.02	0.92	0.97	0.91	0.95	1.02	1.02	1.02

Table 31: Snake River index spring chinook stock differences in maximum sustainable yield pairwise comparisons of actions.

D	$\lambda n_y / \lambda n_{A1}$	Sa_{A3} / Sa_{A1}	$\Delta M_{A1,A2}$	$\Delta M_{A1,A3}$	$\Delta M_{A1,B1}$	$\Delta M_{A1,B2}$	$\Delta M_{A1,B3}$	$\Delta M_{A1,D1}$	$\Delta M_{A1,D2}$	$\Delta M_{A1,D3}$
0.35	1.75	1.15	-1	648	756	658	724	4	1	2
0.65	1.75	1.15	3	241	297	245	280	8	5	6
0.80	1.75	1.15	4	152	196	156	183	9	6	7
0.35	1	1	-1	162	183	155	176	-1	-1	-1
0.65	1	1	3	15	22	12	20	3	3	3
0.80	1	1	4	-7	-3	-8	-4	4	4	4

Table 31a: Snake River index spring chinook stock differences ΔR pairwise comparisons of actions.

D	$\lambda n_y / \lambda n_{A1}$	Sa_{A3} / Sa_{A1}	$\Delta R_{A1,A2}$	$\Delta R_{A1,A3}$	$\Delta R_{A1,B1}$	$\Delta R_{A1,B2}$	$\Delta R_{A1,B3}$	$\Delta R_{A1,D1}$	$\Delta R_{A1,D2}$	$\Delta R_{A1,D3}$
0.35	1.75	1.15	-5	927	1049	938	1013	11	0	6
0.65	1.75	1.15	8	426	502	433	479	25	14	19
0.80	1.75	1.15	11	300	364	305	345	28	17	22
0.35	1	1	-5	314	345	304	335	-5	-5	-5
0.65	1	1	8	43	63	36	56	8	8	8
0.80	1	1	11	-28	-11	-33	-16	11	11	11

Table 32: Snake River index stock spring chinook levels for E, ΔM, G, and ΔR under different levels of D for comparison of A1 to the C alternatives. Note that with John Day Drawdown no benefit on ocean survival is assumed to occur.

D	$E_{A1,C1}$	$E_{A1,C2}$	$E_{A1,C3}$	$\Delta M_{A1,C1}$	$\Delta M_{A1,C2}$	$\Delta M_{A1,C3}$	$G_{A1,C1}$	$G_{A1,C2}$	$G_{A1,C3}$	$\Delta R_{A1,C1}$	$\Delta R_{A1,C2}$	$\Delta R_{A1,C3}$
0.35	96	46	80	24	10	19	1.18	1.08	1.15	67	31	55
0.65	-210	-261	-227	-16	-14	-16	0.69	0.63	0.67	-118	-142	-125
0.80	-318	-369	-335	-10	-4	-8	0.57	0.52	0.55	-167	-188	-174

Table 33: Snake River spring chinook difference in equilibrium E, maximum sustainable yield ΔM, and total population under MSY ΔR under different D hypotheses

D	$E_{A1,A0}$	$E_{A1,A2}$	$\Delta M_{A1,A0}$	$\Delta M_{A1,A2}$	$\Delta R_{A1,A0}$	$\Delta R_{A1,A2}$
0.35	35	-9	8	-1	23	-5
0.65	-272	11	-14	3	-147	8
0.80	-380	16	-2	4	-192	11

Table 33a: Snake River fall chinook difference in equilibrium E, maximum sustainable yield ΔM, and total population under MSY ΔR under different D hypotheses

D_{A1}	D_{A2}	$E_{A1,A0}$	$E_{A1,A2}$	$\Delta M_{A1,A0}$	$\Delta M_{A1,A2}$	$\Delta R_{A1,A0}$	$\Delta R_{A1,A2}$
0.05	0.24	5631	4475	13700	8827	15054	9995
0.05	1.00	5631	9424	13700	46673	15054	48404
0.05	0.05	5631	0	13700	0	15054	0
0.20	0.20	1747	0	2103	0	2650	0
1.00	1.00	-3794	0	-1499	0	-3099	0

Table 34: Snake River fall chinook difference in equilibrium populations for passage survivals and D assumptions. Note equilibrium population under A1 is 7259 spawners.

Passage	D_{A1}	$E_{A1,A3}$	$E_{A1,B1}$	$E_{A1,B2}$	$E_{A1,D1}$	$E_{A1,D2}$
	0.05	10373	11024	10894	172	87
Sa_{A3} high	0.20	6490	7141	7011	172	87
Sm_{A3} high	1.00	949	1599	1469	172	87
Sa_{A3} low	0.05	6456	6951	6857	0	0
Sm_{A3} low	0.20	2573	3067	2973	0	0
	1.00	-2969	-2474	-2568	0	0

Table 34a: Snake River fall chinook difference in ΔR for passage survivals and D assumptions.

Passage	D_{A1}	$\Delta R_{A1,A3}$	$\Delta R_{A1,B1}$	$\Delta R_{A1,B2}$	$\Delta R_{A1,D1}$	$\Delta R_{A1,D2}$
	0.05	63548	76406	73654	214	107
Sa_{A3} high	0.20	19963	24498	23528	214	107
Sm_{A3} high	1.00	1300	2381	2151	214	107
Sa_{A3} low	0.05	19749	24365	23665	214	107
Sm_{A3} low	0.20	4364	6033	5781	214	107
	1.00	-2625	-2177	-2244	214	107

Table 35: Snake River fall chinook difference in maximum sustained yield for different passage survivals and D assumptions.

Passage	D_{A1}	$\Delta M_{A1,A3}$	$\Delta M_{A1,B1}$	$\Delta M_{A1,B2}$	$\Delta M_{A1,D1}$	$\Delta M_{A1,D2}$
	0.05	61764	74592	71846	155	77
Sa_{A3} high	0.20	18494	22955	21999	155	77
Sm_{A3} high	1.00	987	1876	1683	155	77
	0.05	18284	22824	22135	155	77
Sa_{A3} low	0.20	3600	5113	4881	155	77
Sm_{A3} low	1.00	-1419	-1271	-1297	155	77

Table 36: Snake River fall chinook G for passage survivals and D assumptions.

Passage	D_{A1}	$G_{A1,A3}$	$G_{A1,B1}$	$G_{A1,B2}$	$G_{A1,D1}$	$G_{A1,D2}$
	0.05	16.46	19.62	18.94	1.04	1.02
Sa_{A3} high	0.20	5.76	6.87	6.63	1.04	1.02
Sm_{A3} high	1.00	1.29	1.54	1.48	1.04	1.02
	0.05	5.71	6.53	6.36	1.00	1.00
Sa_{A3} low	0.20	2.00	2.28	2.23	1.00	1.00
Sm_{A3} low	1.00	0.44	0.51	0.50	1.00	1.00

Table 37: Snake River fall chinook levels for E, ΔM and G under different levels of D and passage survivals for comparison of A1 to the C alternatives.

Passage	D_{A1}	$E_{A1,C1}$	$E_{A1,C2}$	$\Delta M_{A1,C1}$	$\Delta M_{A1,C2}$	$G_{A1,C1}$	$G_{A1,C2}$	$\Delta R_{A1,C1}$	$\Delta R_{A1,C2}$
	0.05	6179	5972	16628	15474	5.30	5.01	18058	16877
Sa_{A3} high	0.20	2295	2089	3055	2678	1.85	1.75	3749	3318
Sm_{A3} high	1.00	-3246	-3452	-1458	-1479	0.41	0.39	-2793	-2912
	0.05	6179	5972	16628	15474	5.30	5.01	18058	15877
Sa_{A3} low	0.20	2295	2089	3055	2678	1.85	1.75	2749	3318
Sm_{A3} low	1.00	-3246	-3452	-1458	-1479	0.41	0.39	-2793	-2912

Table 38. Hanford Reach fall chinook difference in equilibrium populations, $E_{row,col}$, between actions. Equilibrium population under A1 is 132500 spawners.

	A2	A3	B1	B2	C1	C2	D1/D2
A1	27749	-24478	-16876	-19315	-16876	-19315	30347
A2		-52228	-44626	-47065	-44626	-47065	2597
A3			7602	5162	7602	5162	54825
B1				-2440	0	-2440	47223
B2					2439	0	49662
C1						-2440	47224
C2							49663

Table 39 Hanford Reach fall chinook difference in MSY for actions, $\Delta M_{row,col}$. Note MSY under A1 is 214640 spawners.

	A2	A3	B1	B2	C1	C2	D1/D2
A1	142694	-72715	-54066	-60391	-54066	-60391	160677
A2		-215409	-196760	-203085	-196760	-203085	17983
A3			18649	12324	18649	12324	233392
B1				-6325	0	-6325	214743
B2					6325	0	221068
C1						-6325	214743
C2						-6325	221068

Table 40 Hanford Reach fall chinook $G_{row,col}$.

	A2	A3	B1	B2	C1	C2	D1/D2
A1	1.74	0.61	0.71	0.67	0.71	0.67	1.83
A2		0.35	0.40	0.39	0.40	0.39	1.05
A3			1.16	1.10	1.16	1.10	2.99
B1				0.95	1.00	0.95	2.57
B2					1.04	1.00	2.70
C1						0.95	2.63
C2						0.95	2.70

Table 41. Upper Columbia spring chinook difference in equilibrium populations, $E_{row,col}$, between actions. Note equilibrium population under A1 is 1061 spawners.

	A2	A3	B1	B2	B3	C1/D1	C2/D2	C3/D3
A1	0	0	184	101	143	184	101	143
A2		0	184	101	143	184	101	143
A3			184	101	143	184	101	143
B1				-84	-42	0	-84	-42
B2					42	83	0	42
B3						41	-43	0
C1/D1							-84	-42
C2/D2								42

Table 42 Upper Columbia spring chinook difference in maximum sustained yield, $\Delta M_{row,col}$ for actions. Note MSY under A1 is 369 spawners.

	A2	A3	B1	B2	B3	C1/D1	C2/D2	C3/D3
A1	0	0	91	47	68	91	47	68
A2		0	91	47	68	91	47	68
A3			91	47	68	91	47	68
B1				-44	-23	0	-44	-23
B2					21	44	0	21
B3						23	-21	0
C1/D1							-44	-23
C2/D2								21

Table 43. Upper Columbia spring chinook $G_{row,col}$.

	A2	A3	B1	B2	B3	C1/D1	C2/D2	C3/D3
A1	1	1	1.19	1.10	1.15	1.19	1.10	1.15
A2		1	1.19	1.10	1.15	1.19	1.10	1.15
A3			1.19	1.10	1.15	1.19	1.10	1.15
B1				0.92	0.96	1.00	0.92	0.96
B2					1.04	1.08	1.00	1.04
B3						1.04	0.95	1.00
C1/D1							0.92	0.96
C2/D2								1.04

7 Downstream passage model

The CRiSP model (Anderson et al. 1999) was used for the detailed analysis of actions at the John Day project. The model tracks release groups of juvenile salmonids as they migrate through the hydrosystem. Mortality is attributed to direct dam mortality (as fish pass through the turbines, bypass, or spillways of dams), predation (in the forebays, tailraces and main reservoirs), and gas bubble disease resulting from nitrogen supersaturation. Fish migration rate is modeled in terms of river velocity date in the season, and length of time in migration (Zabel and Anderson 1997, Zabel et al. 1998). In addition, fish are collected at several dams and transported to below Bonneville Dam.

For the life-cycle modeling, a combination of CRiSP v1.5 and v1.6 was utilized. CRiSP v1.5 was used in the PATH spring chinook analysis, and the results from these runs were used in the life-cycle analysis, with the following exception. For the more detailed analysis of explicit actions at the John Day project (results presented below) we utilized CRiSP v1.6. The previous v1.5 results were then scaled to reflect the proportional survival increases under the various actions. CRiSP v1.6 was used for all fall chinook analyses since this was the version used in PATH.

The primary difference between v1.5 and v1.6 is the data used for survival calibration. CRiSP v1.6 relies on NMFS survival estimates from PIT tag data (details in Anderson et al. 1999). CRiSP v1.5 was calibrated to predator consumption and abundance indices (Anderson et al. 1996). Comparisons between the two versions show that they produce similar results. All dam passage parameters (FGE, spill effectiveness, dam passage mortality) for both versions were taken from PATH reports (e.g., Marmorek et al. 1998) and were common to both CRiSP and FLUSH models.

7.1 Configuration for John Day drawdown

For spring chinook, survival through John Day was modeled with both reservoir improvement and John Day Dam passage improvements. Reservoir survival was modeled to reflect a decrease

in reservoir passage time resulting from the increased water velocity through the reservoir. In natural river drawdown, John Day Dam passage survival was set at 100%. With spillway crest drawdown, passage survival was set at 98%.

For fall chinook, upper and lower bounds were modeled for John Day drawdown. Both bounds assumed that with natural drawdown the survival in dam passage and the forebay would be removed. An upper bound of reservoir survival was modeled to increase in proportion with the increase in water velocity, which reflects a decrease in reservoir residence. For a lower bound the fish travel time was not decreased with drawdown. For spillway crest drawdown these same scenarios were applied for reservoir survival but dam passage survival was set to 98% reflecting a small mortality in spillway passage.

7.2 Passage Model Results

For CRiSP model runs presented in this analysis, fish were released at the forebay of Lower Granite Dam. The release distributions were based on passage index data for wild spring and fall chinook. Survivals and travel times were computed from release point to Bonneville tailrace. The survivals reported were a weighted average based on release size. The travel times were the median travel times for all fish in a given year.

The model runs utilized historical flows and temperatures with current dam operations and survivals. Water years modeled, including flow and temperature conditions, were 1975-1998 for spring chinook and 1975-1992 for fall chinook.

7.2.1 Mortality associated with the John Day project

The improvement in survival as a result of an action at the John Day project can be calculated as

eq (35) $$\Delta S = S_Y / S_X - 1.0$$

where S_X is the total in-river survival with John Day project at full pool and S_Y is the total in-river survival under an action X at John Day projects. This can be expressed as a percentage by multiplying the result by 100.

7.2.2 Changes in travel time under John Day operations

The change in travel time associated with different John Day operations can be expressed

eq (36) $$\Delta tt = tt_X - tt_X$$

where tt_X is the travel time through John Day reservoir with full pool operation and tt_Y is the travel time through John Day pool under various actions.

7.2.3 Spring Chinook Results

The results of the spring chinook passage modeling analysis are presented in Tables 44 to 47. The flood control scenario was performed for 1997 only, which was a high flow year.

Table 44: In-river survival for Snake River spring chinook under various management actions. Survivals are from the forebay of Lower Granite Dam to the tailrace of Bonneville Dam. High survival uses drawdown survival through Snake River of 0.95, low survival used drawdown survival of 0.85.

	min	mean	max	S.D.
A1 (inriver only)	0.327	0.437	0.551	0.062
A3 (high survival)	0.533	0.619	0.719	0.042
A3 (low survival)	0.477	0.554	0.635	0.038
B1 (high survival)	0.640	0.705	0.777	0.042
B1 (low survival)	0.570	0.631	0.695	0.038
C1 (JD draw down)	0.391	0.498	0.603	0.062
C2 (JD spill crest)	0.360	0.471	0.580	0.064

Table 45: Travel times in days for Snake River spring chinook under various management actions. Travel times are from the forebay of Lower Granite Dam to the tailrace of Bonneville Dam.

	min	mean	max	S.D.
A1 (inriver only)	15.3	18.0	20.8	1.1
A3	10.6	12.7	15.1	0.9
B1 (JD draw down)	9	10.6	12.7	0.8
C1 (JD draw down)	13.7	15.9	18.4	1.0
C2 (JD spill crest)	14.3	17.0	19.8	1.1

Table 46: Proportional changes in survival between various management actions as change in survival.

	min	mean	max
C2/no transport	0.053	0.079	0.105
C1/no transport	0.094	0.145	0.196
C1/C2	0.040	0.061	0.086
C1/C3 (97 only)		0.01	
B1/A3	0.09	0.14	0.19

Table 47: Changes in travel time of Snake River spring chinook between various management scenarios as change in travel time for in river fish.

	min	mean	max
A1 - A3	4.7	5.3	5.7
A1 - B1	6.3	7.4	8.1
A3 - B1	1.6	2.1	2.4
A1 (inriver only) - C2	1.0	1.1	1.0
A1 (inriver only) - C1	1.6	2.1	2.4
C2 - C1	0.6	1.0	1.4
C3 - C1 (97 only)		0.5	

Mean survival under John Day drawdown was estimated to improve by 14.5 percent as compared to a full river/no transport option. The drawdown to spillway crest option improved survival by

7.9 percent. In the flood control model run, survival decreased by 1.3 percent as compared to the full river drawdown with no flood control.

The full river drawdown decreased travel times by 2.1 days as compared to the full pool option, and the spill crest option decreased travel time by 1 day. The flood control option increased median travel time by half a day as compared to full drawdown with no flood control.

7.2.4 Fall Chinook results

The results of passage modeling for fall chinook are presented in Tables 48 to 51. For the A3 and B1 scenarios, we modeled both a lower bound (0.61) and upper bound (0.89) survival through the free-flowing Snake River based on the latest PATH fall chinook report (Marmorek et al. 1999).

We did not run a "C3" scenario (flood control) for fall chinook because the water was released prior to the onset of fall chinook migration. If flood-control water was stored in John Day pool until the migration, fall chinook could receive a potential benefit. The tables below show fall chinook survivals from LGR forebay to BON tailrace with modifications to John Day pool only.

Table 48: Inriver survival for Snake River fall chinook under various management actions. Survivals are from the forebay of Lower Granite Dam to the tailrace of Bonneville Dam.

	min	mean	max	S.D.
A2	0.263	0.292	0.319	0.019
A3 (lower bound)	0.341	0.352	0.363	0.006
A3 (upper bound)	0.498	0.513	0.530	0.009
B1 (lower bound)	0.385	0.397	0.409	0.007
B1 (upper bound)	0.562	0.579	0.596	0.011
C1 (JD draw down)	0.283	0.312	0.342	0.021
C2 (JD spill crest)	0.273	0.301	0.329	0.020
No transport	0.254	0.279	0.305	0.018

Table 49: Travel times in days for Snake River fall chinook under various management actions. Travel times are from the forebay of Lower Granite Dam to the tailrace of Bonneville Dam.

	min	mean	max	S.D.
A2	21.7	24.2	26.6	1.6
A3	13.8	17.1	21.1	2.0
B1	10.4	13.2	17.7	2.0
A1 (in-river only)	21.5	24.0	26.6	1.6
C2 (JD spill crest)	20.3	22.9	25.8	1.7
C1 (JD draw down)	19.0	21.5	24.6	1.7

Table 50: Proportional changes in Snake River fall chinook survival between various management actions.

	min	mean	max
C2/no transport	0.071	0.078	0.085
C1/no transport	0.110	0.120	0.126
C1/C2	0.036	0.039	0.043
B1/A3	0.115	0.127	0.137

Table 51: Changes in Snake River fall chinook inriver passage travel time between as a result of various management actions.

	min	mean	max
A2-A3	5.4	7.2	8.5
A2-B1	8.8	11.0	12.1
A3-B1	3.4	3.9	4.2
A1 (inriver only) - C2	0.8	1.0	1.2
A1 (inriver only) - C1	2.0	2.4	2.7
C2-C1	1.2	1.4	1.5

The improvement in survival conferred by drawing down John Day pool to a natural river level as compared to full pool operations is estimated to be 12-13 per cent. This is based on comparing survival under A3 (drawdown of all four Snake River projects) to B1 (drawdown of Snake projects and John Day) and by comparing C1 (John Day drawdown only) to the full river/no transport option. Note that this improvement in survival results from both reduced reservoir and dam mortality. Approximately 2/3 of this improvement could be realized by only drawing the river down to the spill crest level.

Travel times were reduced by 2-4 days with John Day drawdown.

8 Discussion

The impact of John Day drawdown action on the Snake River and Upper Columbia spring and fall chinook was evaluated using several response measures and three modeling approaches. The analysis was conducted with the PATH Bayesian life-cycle analysis, which provided probabilities of meeting survival and recovery goals. In addition, the analysis was applied to determine the equilibrium spawner levels. This analysis was conducted for Snake River listed spring and fall chinook. A second approach used a deterministic life-cycle analysis under equilibrium conditions. This approach provided the equilibrium number of spawners, the MSY and the total adult population under MSY of Snake River spring and fall chinook, Hanford Reach fall chinook, and Upper Columbia spring chinook. A third analysis evaluated the impact of the drawdown actions on the smolt passage survival and travel time using the CRiSP smolt passage model.

Actions evaluated include transportation, Snake drawdown only, John Day drawdown only, with and without transportation from the Snake River, and a drawdown including the Snake River and John Day reservoir. Three variations of John Day drawdown were evaluated: natural river drawdown, spillway crest drawdown, and natural river drawdown with flood control measures in the spring that allowed for partial refilling of the reservoir during times of high flow.

Six of stock performance measures were evaluated: probability of survival over 24 years, probability of recovery over 48 years, equilibrium population level, maximum sustainable yields, smolt passage survival, and smolt passage travel time. A number of hypotheses were explored on in-river survival of smolts and the linkages between freshwater survival, smolt passage stages, and ocean survival. In addition, several hypotheses were explored on the expected levels of survival under drawdown and on the survival of adults migrating up river.

The Bayesian analysis and the deterministic analysis both provide pairwise estimates of the difference in equilibrium population levels between two alternatives. The Bayesian results for Snake River spring chinook are given in Table 19 and the deterministic results are given in Table 25. The Bayesian analysis gave larger values. For example, the difference in equilibrium population levels between alternatives B1 and A3 for the Bayesian analysis were between 68 and 217 spawners under equal weighting of hypotheses, while the deterministic value was 58. Although these estimates are different the ratio of the equilibrium levels defined (B1-A3)/A1 is nearly identical for the two methods. From the Bayesian analysis (Table 19) the geometric mean of this ratio, across the 7 index stocks, is 0.163. For the deterministic model the ratio is 0.138. Noting from eq (17) that the equilibrium level in the deterministic model is scaled by the Ricker parameter b, it follows that the value of a pairwise comparison depends on b, but in the ratio measure, (B1-A3)/A1, the Ricker b terms cancel. This is also true of the Bayesian model.

We note that the equilibrium ratio (B1-A3)/A1 is very close in the two methods while the measure B1-A3 is different. Therefore, the main difference in pairwise comparisons from the two methods involves the choice of the Ricker b used in each method. For the Bayesian analysis the mean value of the Ricker b is 0.00122 while the mean value for the deterministic method was 0.00174. The difference in the estimates of b accounts for over 80% of the difference in the results of the two methods. The remaining difference in the methods is a result of slightly different passage assumptions. Mathematically the two approaches are functionally equivalent, so using the same input in the two methods should give very similar results.

The question of which set of Ricker b estimates are better can not be resolved. In this work the estimates used in the Bayesian analysis and the estimates used in the deterministic method were both extracted from work by the same researchers developed at different times. The Bayesian analysis used b values prepared by Schaller, Petrosky and Langness for the PATH analysis (Marmorek and Peters 1998) and the deterministic estimate used b values published by Schaller et al (1999).

8.1 Bayesian Analysis

The analysis showed a range of probabilities of survival and recovery depending on the action and hypotheses. Irrespective of the details, general trends emerge that are illustrated by taking the upper and lower estimates of the absolute values and differences in survival and recovery probabilities of transportation relative to drawdown actions

The ranges in survival probabilities with different hypotheses were small and indicated that in the BSM analysis the hypotheses projecting 24 year survival probabilities for spring or fall chinook did not give significantly different results. The second feature is that all three actions, transportation (A2), Snake River drawdown (A3) and Snake River plus John Day drawdown (B1) gave essentially the same chance of survival, which was estimated to be high. The third result was that assumptions were not important in distinguishing actions A3 vs. B1 for recovery. The model predicted a high chance of recovery for both actions. The fourth result was that hypotheses were important in determining the effectiveness of transportation actions. The range of recovery probabilities was large and under some hypotheses drawdown actions were significantly better than transport actions. The assumptions making drawdown better than transportation were explored with the deterministic model.

Table 52: Range of Snake River spring and fall chinook survival and recovery probability means under actions A1, A2, A3 and B1.

Standard	Action	Range	A3 - A1	B1 - A3
	A1, A2	0.56-0.76	0.01 to 0.08	
24-year survival	A3	0.63-0.79		-0.002 to 0.005
spring chinook	B1	0.64-0.79		
	A1, A2	0.33-0.60	0.10 to 0.50	
48-year recovery	A3	0.67-0.88		0.02 to 0.03
spring chinook	B1	0.70-0.91		

Standard	Action	Range	A3 – A2	B1 - A3
	A1, A2	0.83-0.99	-0.002 to 0.07	
24-year survival	A3	0.88-1.00		-0.001 to 0
fall chinook	B1	0.89-1.00		
	A1, A2	0.35-0.92	0 to 0.65	
48-year recovery	A3	0.86-0.92		0 to 0.043
fall chinook	B1	0.91-1.00		

The equilibrium level, which is also a measure of how well an action can lead to recovery, was estimated individually for each of the index stocks for both spring chinook and fall chinook. The total equilibrium values of the index stocks for spring chinook, as the sum of individual stocks from Table 19, are given in the Table 53 below. The analysis used seven index stocks. The Snake River Basin has about 38 index stocks, so the modeled stocks may represent about one quarter of the total spring chinook population in the Snake River Basin. For spring chinook the equilibrium level under drawdown actions were about 50% higher than under the transportation actions. John Day drawdown, with Snake River drawdown, added an additional 10% to the equilibrium population level beyond just the Snake River drawdown. For the fall chinook (See Table 20), the drawdown actions increase the equilibrium level of stocks by a factor of three, and adding John Day drawdown to the Snake River drawdown increased the stocks by 10%.

Table 53: Total equilibrium spring chinook population (index stocks) for each action and hypotheses plus the difference in index stocks comparing actions B1 to A3 and A3 to A1.

Hypotheses	A1	A2	A3	B1	B1-A3	A3-A1
EQUAL	4956	4903	7418	8232	814	2462
FLUSH	4365	4200	9829	11296	1467	5464
CRISP	5904	5896	7360	7846	486	1456
NMFS	4600	4611	5067	5552	485	467
OPT.PASS	5149	5081	8267	9146	879	3118
PESS.PASS	4763	4726	6571	7319	748	1808

8.2 Deterministic life-cycle analysis

Using the deterministic model, pairwise comparisons were made between the drawdown alternatives (i.e. A3, B and C) and between the transportation and drawdown alternatives (i.e. A vs. B, C, and D). A John Day drawdown, along with a Snake River drawdown, always improved the fish population measures (A3 vs. B1). Drawing down John Day only, and ending transportation (A3 vs. C1), was significantly worse for Snake River spring and fall chinook than if the Snake River system were drawn down only (A3 vs. C2) (Table 54).

Table 54. Pair-wise comparison of differences in Snake River drawdown equilibrium level, MSY and R with various combinations of John Day drawdown.

	Snake spring			Snake fall		
	E	ΔM	ΔR	E	ΔM	ΔR
A3 vs. B1	2128	4104	4636	668	9061	9104
A3 vs. C1	-13452	-17024	-21166	-2801	-24199	-24471
A3 vs. C2	-15352	-18544	-23370	-3007	-25353	-25652
A3 vs. C3	-14060	-17518	-21888			

A comparison of the base action (A1) to a natural river drawdown of John Day produced mixed results for stocks depending on the assumptions made and the stocks being considered (Table 55). High transportation efficiency assumptions produced negative numbers in the A1 vs. C1 comparisons. That is, compared to the current operating conditions, equilibrium numbers and MSY decreased with a John Day drawdown that also terminated the transportation program. Using an assumption of low transportation effectiveness, C1 produced positive numbers for Snake River spring and fall chinook relative to A1. For Hanford Reach fall chinook, John Day drawdown was detrimental. Studies on the effectiveness of transporting Hanford Reach fish from McNary dam suggest a high transport effectiveness for this system. Thus, transportation was always better than passing the fish through the lower Columbia River hydrosystem. Upper Columbia spring chinook, which are currently not transported from McNary dam, experienced a small increased in equilibrium numbers and MSY with John Day drawdown.

Table 55: Difference in equilibrium population levels and MSY comparing base action A1 to drawdown of John Day reservoir, C1.

	$E_{A1,C1}$	$\Delta M_{A1,C1}$	$\Delta R_{A1,C1}$
Snake River spring chinook	-11791 to 2336	-2976 to 1280	2546 to -6346
Snake River fall chinook	-3428 to 6179	-1487 to 16791	18058 to -2793
Hanford Reach fall chinook	-16876	-54066	-57322
Upper Columbia spring chinook	184	91	165

8.3 Smolt passage analysis

Smolt passage information derived from the CRiSP passage model show small changes in travel time for spring chinook and relatively large changes for fall chinook. For spring chinook, the comparison was made only between full-pool travel time and the drawdown options. The difference was two days between full pool and natural drawdown of John Day reservoir. Flood control delayed fish an additional half-day. The spillway crest option gave results between full pool and the natural river drawdown. There was little difference in the survivals between the different C alternatives. Fall chinook exhibited a larger response to John Day drawdown with the benefit of a decreased travel time up to one week. In a comparison between full pool and

drawdown of the Snake River dams plus the John Day Dam, the average in-river survival changed from 0.29 to 0.51. The travel time of the fall chinook changed from 24 days to 13 days between the two configurations.

8.4 Summary

To develop an understanding of the relative benefits of John Day drawdown under the Bayesian model analysis, we compared results of a Snake River drawdown (A3) to the Snake River plus John Day drawdown to natural river level (B1). The results in Table 56 used passage assumptions that optimized the smolt and adult passage survival (OPT.PASS and HIGH.EJUV in Tables 13 through 20). The difference in the measures of A3 and B1 illustrates the individual contribution of a natural river drawdown of John Day reservoir. The drawdown contributed nothing to Snake River spring and fall chinook recovery and survival probabilities. Under current conditions (A1), the Bayesian analysis suggested equilibrium populations of about 5000 adults for both spring and fall chinook. The John Day drawdown increased the equilibrium population by approximately 20% for spring chinook and 50% for fall chinook.

Table 56: Difference in Snake River salmon survival and recovery probabilities and the difference in equilibrium populations between A3 and B1. Results are from the Bayesian model with weightings of hypotheses favoring optimum passage conditions.

	Δ 24-yr Surv. Prob.	Δ 48-yr Recov. Prob.	$E_{A3,B1}$
spring chinook	-0.0005	0.022	879
fall chinook	0.0005	0.011	2326

To characterize the effect of a John Day drawdown on the Upper Columbia and the Snake River stocks, the deterministic model was used in pairwise comparison of actions. For actions without transportation, the effect of John Day can be characterized by comparing action A3 to B1, B2 and B3. That is, to characterize the effects of the John Day drawdown on the equilibrium conditions, the difference in including a John Day drawdown to the Snake River drawdown is compared to a Snake River drawdown alone. The maximum improvement of adding the John Day drawdown was between 1 and 17%, depending on species (Table 57).

Table 57. Effects of John Day drawdowns are illustrated with pair-wise comparisons to Snake River drawdown equilibrium levels. Equilibrium population under current condition, A1, is Equ. Pop. The total Snake River spring chinook population is estimated assuming 38 stocks.

Stock	Equ. Pop.	$E_{A3,B1}$	$E_{A3,B2}$	$E_{A3,B3}$
Snake River spring chinook	15942	2128	190	1520
Snake River fall chinook	7259	647	572	
Hanford Reach fall chinook	132499	7602	5162	
Upper Columbia spring chinook	1061	184	101	143

The impact of the actions on the pairwise difference of MYS plus escapement is illustrated in Table 57a. The spring and fall chinook estimates were generated with high and low D values, which give the least and most benefit to action B1, B2 and B3.

Table 57a. Effect of John Day drawdowns are illustrated with pair-wise comparisons to Snake River drawdown equilibrium levels. Recruitment plus harvest at MSY population under current conditions, A1, is R_{A1}. Difference in R is ΔR. The total Snake River spring chinook population is estimated assuming 38 stocks.

Stock	R_{A1}	D	$\Delta R_{A3,B1}$	$\Delta R_{A3,B2}$	$\Delta R_{A3,B3}$
Snake River spring chinook	4006	0.35	4636	418	3268
		0.80	2432	190	1710
Snake River fall chinook	3288	0.05	11043	9667	
		1.00	942	825	
Hanford Reach fall chinook	190404	1.00	20358	13502	
Upper Columbia spring chinook	422	NA	165	88	126

The effect of John Day drawdown without transportation, compared to the current transportation of fish in A1, presents mixture benefits and detriments depending on the value of transportation effectiveness as characterized by the D factor (Table 58). The maximum and minimum effects are illustrated by comparing A1 to C1, C2 and C3. For Snake River stocks, when D is low, John Day drawdowns by themselves are beneficial, but if D is high, drawdown is detrimental to fish. For the Hanford Reach stock, a John Day drawdown that removes the transportation of these fish is always detrimental, while the drawdown has a small benefit for Upper Columbia Spring

chinook, since none of these fish are currently transported at McNary Dam. In Table 58 other factors are selected that give the maximum benefit for the low D calculation and maximum detriment for the high D calculation.

Table 58. Effect of John Day drawdowns relative to transportation actions are illustrated with pair-wise comparisons of John Day drawdowns to current conditions. Measures give differences in equilibrium levels relative to A1. Transportation effectiveness is D. Equilibrium population under current condition, A1, is Equ. Pop.

Stock	Equ. pop.	D	$E_{A1,C1}$	$E_{A1,C2}$	$E_{A1,C3}$
Snake River spring chinook	15942	0.35	3648	1748	3040
		0.80	-12084	-14022	-12730
Snake River fall chinook	7259	0.05	18058	16877	
		1.00	-2793	-2912	
Hanford Reach fall chinook	132500	1.00	-16876	-19315	
Upper Columbia spring chinook	1061	NA	184	101	143

Table 58a. Effect of John Day drawdowns relative to transportation actions are illustrated with pair-wise comparisons of John Day drawdowns to current conditions. Measures give difference in populations at MSY relative to A1. Transportation effectiveness is D. Recruitment plus harvest at the MSY population under current condition, A1, is R_{A1}. Difference in R is ΔR. The Snake River is assumed to have 38 spring chinook stocks.

Stock	R_{A1}	D	$\Delta R_{A1,C1}$	$\Delta R_{A1,C2}$	$\Delta R_{A1,C3}$
Snake River spring chinook	4006	0.35	2546	1178	2090
		0.80	-6346	-7144	-6612
Snake River fall chinook	3288	0.05	18058	16877	
		1.00	-2793	-2912	
Hanford Reach fall chinook	190404	1.00	-57322	-64178	
Upper Columbia spring chinook	422	NA	165	88	126

The final comparison shows the effects of John Day drawdown relative to current conditions, assuming that a fish transportation system can be implemented along with a drawdown. In this case the pairwise comparisons is between A1 vs. D1, D2 and D3 (Table 59 and 59a). The result

is that the effect of drawdown depends on the transportation effectiveness on Snake River spring chinook. If transportation is effective (D = 0.80) then John Day drawdowns are beneficial. If the transportation is ineffective (D= 0.35) then drawdown with transportation is detrimental. For Snake River fall chinook, drawdown is insignificant relative to current condition, but it does improve Hanford Reach fall chinook. The impact on Upper Columbia spring chinook is small.

Table 59. Effect of John Day drawdowns with transportation relative to the current operating conditions are illustrated with pairwise comparisons of equilibrium levels relative to A1. Parameters, selected to produce estimates, are given as Param. Equilibrium population under A1 is Equ. pop.

Stock	Equ. pop.	Param.	$E_{A1,D1}$	$E_{A1,D2}$	$E_{A1,D3}$
Snake River spring chinook	15942	D = 0.35	-342	-342	-342
		D = 0.80	608	608	608
Snake River fall chinook	7259	Sa,Sm high	172	87	
		Sa,Sm low	172	87	
Hanford Reach fall chinook	132500		30347	30347	
Upper Columbia spring chinook	1061		184	101	143

Table 59a. Effect of John Day drawdowns with transportation relative to the current operating conditions are illustrated with pairwise comparisons of population levels at MSY relative to A1. Parameters, selected to produce estimates, are given as Param. Recruitment plus harvest at MSY population under current conditions, A1, is R_{A1}. Difference in R is ΔR. Snake River spring chinook consists of 38 stocks.

Stock	R_{A1}	Param.	$\Delta R_{A1,D1}$	$\Delta R_{A1,D2}$	$\Delta R_{A1,D3}$
Snake River spring chinook	4006	D = 0.35	-190	--190	--190
		D = 0.80	418	418	418
Snake River fall chinook	3288	Sa,Sm high	214	107	
		Sa,Sm low	214	107	
Hanford Reach fall chinook	190404		164908	164908	
Upper Columbia spring chinook	422		165	88	126

This analysis indicates that a John Day drawdown can have a positive or negative impact on the stocks above the reservoir. The relative effects are clear when comparing the contribution of a John Day reservoir drawdown to the drawdown of the Snake River dams. Under this comparison

the effectiveness of transportation is not a large issue, since neither action has transportation. Then under the assumptions of the model, the contributions of John Day are incremental and small. As illustrated in Table 57, spring chinook equilibrium increases by up to 2000 fish, while the fall chinook increase by about 600 fish. The increase in the harvest plus escapement at MSY (Table 57a) is up to four thousand spring chinook adults and eleven thousand fall chinook adults. In comparison, the current equilibrium levels for the Snake River spring and fall chinook are about sixteen thousand and seven thousand respectively. Historically, the runs were considerably larger. In the 1950s, annual spring chinook returns to the Snake River were over one hundred thousand adults and in the 1960s they were about sixty thousand. The Snake River fall chinook spawning population, including escapement to the mouth of the Columbia and harvest, averaged about seventeen thousand adults over the period 1966-1991 (Peters et al 1999). This level could be obtained by fall chinook transportation, if the current transportation is ineffective (Table 58). Furthermore, under this assumption, the contribution of improved passage with a John Day drawdown is insignificant (Table 59). If fall chinook transportation is effective, then stopping transportation decreases the population (Table 58).

8.5 Final Remarks

This analysis suggests that benefits of a John Day drawdown are uncertain. In general, the effect of a John Day drawdown on Snake River fish depends on the assumptions about the extra mortality and delayed mortality in transportation. If transportation is effective, then drawdown is ineffective. If transportation is ineffective, then drawdown is effective. Hanford Reach fall chinook are not improved with John Day drawdown, unless transportation is continued with the drawdown. The Upper Columbia stocks generally are not affected by drawdown since the fish are currently not transported and the change in survival with and without drawdown is small. In all cases, a natural river drawdown is better than a spillway crest drawdown and flood control has an insignificant impact on smolts passing through John Day reservoir.

9 References

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10 Comments by PATH and Responses

10.1 Comments provided by Paul Wilson of CBWFW are included below.

Below are comments on the Sept. 13, 1999 draft of the paper. Many of these comments were made to two of the authors on a conference call on Sept. 14. (Note: the numbering does not refer to this draft of the document)

Pg. 17, Table 3. Lower bound (pessimistic) estimates of survival rate through the free-flowing John Day reach were used in PATH modeling, for both spring and fall chinook. The pessimistic estimate for spring chinook was a fixed value of 0.90 [EJUV1 – Marmorek et al. (1998), Table 2.2.1-1]. Note also that the optimistic survival rate value was 0.98, not .95, as indicated in Table 3. For fall chinook, modeled estimates of free-flowing survival rate, not fixed values, were used for both the optimistic and pessimistic values. The alternative assumptions differed in whether there would be a decrease in fish travel time through the reach due to drawdown (Peters et al. 1999, Section 5.2.2).

Pg. 20, Section 4.3. The term λo is not one that was used or defined within PATH. It appears the definition intended here is "the base ocean survival common to transported and non-transported groups." It's not clear what the criteria are for including a PATH-modeled factor influencing post-Bonneville survival in λo as opposed to λn or λt. If λo is "contained within the exp(a) of the equation (sic) stock recruitment equation $R = S \exp(a - bS)$" then the year effect (δy) of the Delta model in PATH is not contained in λo, and must somehow be applied to both λn and λt to be analogous to what was done in PATH

Pg. 22, Section 4.3.1. The three hypotheses about the trend in ocean mortality are not presented in an equitable manner. The claim is made that, only for the hydro hypothesis, "a direct link between fish survival and hydrosystem passage has not been identified", even though it's the only hypothesis of the three for which a credible, direct evidential link *has* been identified (see Schaller et al. 1999).

Pg. 23, Table 5. Description of λn as "characteristic ocean survival factor" is misleading, since it is really only the complement of the extra mortality experienced by Snake R. spring chinook. Ocean survival in the Delta model is composed of several modeled factors, including the year effect. By presenting λn as the ocean survival factor, it leads the reader to believe with FLUSH there is no apparent decline in ocean survival in the period from 1975 on. Inclusion of the year effect with FLUSH model estimates results in reduced ocean survival of spring chinook in the period 1975-90 compared to 1952-90.

Pg. 26, Table 8. Something seems to be amiss here: the conversion rates for B1 and B3 for spring chinook are lower than the A3 value. This makes no sense, unless a detriment to adult passage from dam removal is being assumed.

Pg. 41, Section 6. The text above eq. 17 indicates that hydrosystem passage corridor actions are never expected to alter the spawning habitat. However, in PATH an assumed increase in spawning habitat under Snake River drawdown was simulated for fall chinook, when escapements were very high (Peters et al. 1999).

Pg. 42, Eq 18. Not necessary to estimate S_{msy} numerically. Hilborn (1985) presents direct formulas approximating the spawning level at MSY and MSY for the Ricker curve, and the conditions under which they are useable.

Pg. 46, Eq. 25. This formulation of the benefit of John Day drawdown plus Snake R. drawdown vs. Snake R. drawdown alone illustrates one of the limitations of this analysis. This equation assumes that there are benefits of B1 over and above A3 only in the juvenile and adult hydrosystem passage stages. In PATH, under the hydro extra mortality hypothesis, there would be a benefit to post-Bonneville survival of actions that improve in-river conditions for smolts, such as John Day drawdown. We didn't explicitly consider or model the additional benefits of Snake River plus John Day drawdown compared to Snake River drawdown alone. Effectively, this analysis give a 100% weight to the hypothesis that there is no additional benefit to later life stages of a John Day drawdown, above that of

a Snake R. drawdown alone, and 0% to any other hypothesis. However, PIT tag studies indicate that post-hydrosystem survival of spring chinook smolts is influenced by hydrosystem passage experience (Sandford and Smith 1999, Schaller et al. 1999).

Pg. 48, Section 6.3.1. The number of spring chinook stocks in the Snake R. basin is indicated to be 24. This may be close to the truth, but the ESU consists of both spring and summer chinook stocks, which number 35-40 total. Thus, the increase in the "Snake River Basin index spawning population" is underestimated by an amount proportional to the underestimate in number of stocks in the ESU.

Pg. 52, first paragraph. The claim that "a drawdown of John Day reservoir without drawing down the Snake River reservoirs is not expected to increase ocean survival" is not a conclusion reached, or even discussed, in PATH. We did not do any analysis of scenarios where only John Day is drawn down. PIT tag data indicate that the higher the number of times a smolt passes a dam through bypass, the lower the SAR (Sandford and Smith 1999, Schaller et al. 1999), suggesting that ocean survival decreases with number of dams passed.

Pg. 52, last paragraph. Upper Columbia spring chinook "G" is estimated with eq 25, which includes no effect of transportation. Since, under A2, upper Columbia origin smolts would be transported at McNary dam, eq. 26 or 27 is more appropriate. Tables 41 and 42 treat A1, A2, and A3 as identical in their effect on upper Columbia fish. Because of the MCN transport under A2, they are not. Also, A3 flow timing and magnitude in the lower Columbia would be different from A1 and A2.

Pg. 60, Section 7.2.3. In tables 44 – 47, why weren't B1 vs. A3 comparisons done? C1, C2 and C3 are not necessarily good surrogates for gauging the effect of B1 or B2 vs. A3. Snake River drawdown would increase the water velocity in the Snake, accelerating the arrival time of smolts to McNary dam in either CRiSP or FLUSH. This would, for example, affect flows and temperatures, increasing flows and lowering temperatures experienced by smolts in the lower part of the hydrosystem, and hence affect survival in John Day and the lower reservoirs. Differences of C1, C2, or C3 from. A1 are therefore likely underestimates of the additional benefits to smolt survival provided by John Day drawdown with Snake River drawdown.

Pg. 71, Section 8.5, last paragraph. Earlier comments detail why this analysis does not represent "the expected range of positive and negative impacts of a John Day drawdown." The text about the Ricker equation here makes no sense in the context of this analysis. If Ricker equation is inappropriate, the biggest effect here would be to underestimate the number of spawners resulting from B1, relative to A3 or any other scenario. This is because of the descending right hand limb inherent in the Ricker curve. If productivity doesn't decline at higher abundances, but stays about the same, many more recruits per spawner would be produced at the higher abundances expected under A3 and B1 scenarios.

References

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Schaller, H., N. Bouwes, P. Budy, C. Petrosky, P. Wilson, O. Langness, E. Weber and E. Tinus. 1999. An analysis of differential delayed mortality experienced by stream-type chinook salmon of the Snake River. August 13, 1999 Draft.

10.2 Response to review.

The responses to the review of P. Wilson and other PATH comments communicated by D. Marmorek are listed below:

1. The differences in the Bayesian and deterministic methods was evaluated and resolved. The main result is the two approaches are mathematically equivalent and the differences in pairwise comparisons depend on inputs selected for each method.

2. Concerns were considered related to the interpretation and characterization of ocean mortality using information from the PATH analysis. The Bayesian analysis in this study was conducted with the programs used in PATH and so the approach to characterizing ocean factors is identical to the PATH treatment. For the deterministic method, the approaches are different. The deterministic approach looked at pairwise comparisons at equilibrium, allowing great simplification of the analysis. In this approach all factors that are common between two alternatives, whether they are time variable or not, cancel. Because of this simplification basic differences in ocean survival need only reflect the differences resulting from the compared alternatives. Factors such as the ocean delta factor are subsumed into the Ricker a parameter.

3. It was assumed in this analysis, as was done in PATH, that the John Day reservoir drawdown did not alter ocean survival of stocks. The evidence for a linkage between ocean survival and passage experience remains uncertain and contradictory. Studies suggesting ocean survival increases with passage through fewer bypass systems (Stanford and Smith 1999) are equivocal and more adult returns are needed to establish if a pattern exists. In PATH (Hinrichsen and Paulsen 1998) demonstrated there was no correlation between the post Bonneville survival and the juvenile inriver passage survival. In the analysis here, the effects of juvenile passage experience on ocean survival were encompassed through the Hydro hypotheses for Snake River drawdown, but the possible effects of John Day drawdown were not considered.

4. For spring chinook from the Snake River, the number of stocks in the September draft of this paper was 24. This was changed to 32 spring and summer wild and natural stocks from the Snake River basin. This increased all estimates of stock numbers for equilibrium and the MSY levels by 1/3.

5. No attempt was made to evaluate the expansion of Ricker b factors by a drawdown action. The expansion of the Snake River fall habitat was not considered because the possible competition of Hanford Reach fall chinook for any new habitat could not be resolved, in addition to this being beyond the scope of this study.

6. The MSY calculations were estimated exactly rather than through the Hilborn (1985) approximation. For this study the Hilborn method was invalid.

References

Hinrichsen, R. and C. Paulsen. 1998. Testing the hydro-related extra mortality hypotheses. PATH document.

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