APPENDIX F:

AQUATIC RESOURCES TECHNICAL INFORMATION AND ANALYSIS

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F.1 INTRODUCTION

This technical appendix provides information pertaining to analyses of effects on aquatic ecological resources for the Glen Canyon Dam Long-Term Experimental and Management Plan (LTEMP) Environmental Impact Statement (EIS), including the aquatic food base, native and nonnative fishes, and fish parasites. It is intended to supplement the information presented in Sections 3.5 and 4.5 of the EIS.

Methods used to evaluate resources, including modeling methods, are described and results regarding effects of alternatives and associated long-term strategies are presented. Analysis of effects on the aquatic food base is based upon a review of literature pertaining to past studies and extrapolation of those results to qualitatively evaluate effects of alternatives and the associated long-term strategies. The evaluation of impacts on rainbow trout (*Oncorhynchus mykiss*), endangered humpback chub (*Gila cypha*), other native fish, nonnative fish, and fish parasites is based upon reviews of the scientific literature and upon the evaluation of performance metrics that were developed for the LTEMP assessment process. The values for the performance metrics were calculated using models developed to examine effects of alternatives on the various aquatic resources.

The potential effects of six action alternatives are compared to the no-action alternative (Alternative A), which describes how the dam is currently operated. Operations under Alternative A employ a release pattern established in the 1996 Record of Decision (ROD) (Reclamation 1996) associated with the 1995 Environmental Impact Statement (EIS) on operations of Glen Canyon Dam (Reclamation 1995). This operational release pattern, referred to as Modified Low Fluctuating Flows, moderated the releases relative to operations practiced in the 1960s through 1980s. As described in Chapter 2 of the EIS, Alternative A also includes various operational decisions and non-flow actions that have been established since the 1996 ROD.

Some of the alternatives under consideration in the EIS (especially Alternatives C, D, and E) are complex experimental or adaptive designs. These alternatives prescribe different management interventions depending on resource conditions. Various condition-dependent triggers govern the implementation of experiments. To understand effects of alternatives that incorporate multiple adaptive components, especially components that might be considered experimental, the complex alternatives were decomposed into 19 versions referred to as long-term strategies, with specific experimental elements included or excluded in each long-term strategy. Table 4.1-1 identifies the experimental elements included in each of the long-term strategies associated with the LTEMP alternatives. Descriptions of each alternative, including the elements included in the long-term strategies, are presented in Sections 2.2 and 4.1 of the EIS. Modeling to evaluate potential effects on aquatic resources was conducted similarly for each long-term strategy and results were compared using various performance metrics to evaluate how inclusion of experimental elements as part of an alternative affected the modeled outcome for the resources of concern. As discussed in Section 4.1 of the EIS, the long-term strategies used to represent the alternatives in Section 4.5 are A, B1, C1, D4, E1, F, and G.

A full range of potential hydrologic and sediment conditions were modeled for a 20-year period (water years 2013–2033) that represented the 20 years of the LTEMP. Twenty-one potential Lake Powell inflow scenarios (known as hydrology traces) for the 20-year LTEMP period were sampled from the 105-year historic record (water years 1906 to 2010) using the Index Sequential Method and selecting every fifth sequence of 20 years. Using this approach, the first 20-year period considered was 1906–1925, the second was 1911–1930, and so forth. As the start of traces reach the end of the historic record, the years needed to complete a 20-year period are obtained by wrapping back to the beginning of the historical record. For instance, the trace beginning in 1996 consists of the years 1996–2010 and 1906–1910, in that order. This method produced 21 hydrology traces for analysis that represented a range of possible conditions from dry to wet.

In addition to these 21 hydrology traces, three 20-year sequences of sediment inputs from the Paria River sediment record (water years 1964–2013) were analyzed that represented low (water years 1982 to 2001), medium (water years 1996 to 1965), and high (water years 2012 to 1981) sediment input conditions. In combination, the 21 hydrology traces and 3 sediment traces resulted in an analysis that considered 63 possible hydrology-sediment conditions for each alternative and long-term strategy.

Section F.2 of this appendix describes analyses conducted to evaluate impacts of alternatives on the aquatic food base. Section F.3 presents methods, results, and conclusions from modeling conducted to evaluate population-level effects of alternatives on rainbow trout and humpback chub. Section F.4 presents methods, results, and conclusions for modeling conducted to evaluate how alternatives would affect the suitability of mainstem water temperatures for sustaining populations of humpback chub and other native fish species, nonnative fish species, and fish parasites.

F.2 AQUATIC FOOD BASE ASSESSMENT

This section provides information on flow and temperature effects of LTEMP alternatives on the aquatic food base. It serves as the basis for descriptions and conclusions provided in Sections 3.5.1 and 4.5 of this EIS.

F.2.1 Description of the Aquatic Food Base Downstream from Glen Canyon Dam

Determining the impacts of LTEMP alternatives on the aquatic food base requires an evaluation of changes in the aquatic food base from pre-dam years through various post-dam operations, changes in the food base that occur with increasing distance from the dam, and the effects of intentional and unintentional species introductions. The following discussion provides this information and supplements the aquatic food base information presented in Section 3.5.1.

F.2.1.1 The Aquatic Food Base Prior to Construction of Glen Canyon Dam

Prior to the construction of Glen Canyon Dam, the productivity of the Colorado River was low due to scouring and high turbidity levels that limited the colonization and growth of benthic macroalgae and invertebrates (Woodbury 1959; Stevens et al. 1997; Ward et al. 1986). Generally, the more productive habitats for algae and invertebrates occurred at the lower edge of deltas formed at the mouths of tributaries, on and behind boulders, and on woody debris carried by floodwaters (Woodbury 1959). A pre-dam survey of 171 mi of the Colorado River between Dirty Devil River, Utah, and Lees Ferry, Arizona (collections made along the banks of the Colorado River and in tributaries or side canyons), included 28 species of green algae, 5 species of cyanobacteria, 20 species of diatoms, and 91 species of insects including mayflies, dragonflies, true bugs, fishflies, caddisflies, aquatic snout moths, beetles, and true flies (Woodbury 1959). Sixteen insect species were collected from sites along the river bank while 77 species were collected from tributary streams. From a sample of fish stomachs, it appeared that organisms derived from tributaries and terrestrial habitats played an important role in the diet of river fishes (Woodbury 1959). Pre-dam reports of invasive aquatic food base species in the Grand Canyon are limited. In 1932, 50,000 amphipods (*Gammarus lacustris*) were introduced into Bright Angel Creek. They apparently washed into the mainstem of the Colorado River where they became abundant (Carothers and Minckley 1981), particularly within the Glen Canyon reach, where they are associated with *Cladophora* beds (Blinn and Cole 1991; Blinn et al. 1992; Hardwick et al. 1992).

Stanford and Ward (1986) suggested that the lower Green and Colorado Rivers in Canyonlands National Park, Utah, may provide the best examples of the pre-regulated Colorado River, as these reaches retain similar hydrographs to pre-dam conditions and are the farthest downstream from the large dams in the upper Colorado River basin. High suspended sediment concentrations limited the growth of primary producers; thus, the primary carbon source for benthic invertebrates was terrestrial organic matter. The invertebrate community was composed of 49 taxa, mostly mayflies, caddisflies, and true flies. Stoneflies and dragonflies comprised a smaller portion of the community (Haden et al. 2003).

F.2.1.2 The Aquatic Food Base of the Colorado River Downstream from Glen Canyon Dam

Section 3.5.1 of the EIS provides an overview of the aquatic food base of the mainstem of the Colorado River following installation of Glen Canyon Dam. The following supplements that information. Glen Canyon Dam altered the primary carbon source from terrestrial (e.g., leaf litter) to aquatic (e.g., algae and detritus), the temperature regime from seasonally warm to stenothermically cool, and discharge patterns from low daily variations to high daily variations (Benenati et al. 2002). Nevertheless, riparian and upland vegetation still contribute energy to the impounded river system, particularly during flood events (Blinn et al. 1998, 1999). The large quantity of driftwood that occurred in the pre-dam river is now replaced by lower quantities of woody debris derived from tributaries during floods, or from occasional scouring flows of the vegetated post-dam shoreline (Stevens et al. 1997). Benthic detrital standing mass is generally low and variable, increasing through the more turbid downstream reaches (Shannon et al. 1996).

The Paria and Little Colorado Rivers, the primary sediment delivery systems downstream from Glen Canyon Dam, divide the Colorado River into three distinct turbidity zones that have a significant impact on mainstem aquatic food base communities (Stevens et al. 1997). The first 16 mi downstream from Glen Canyon Dam account for 60% of the total phytobenthic standing biomass throughout the remaining 242 mi of the river corridor (Blinn et al. 1995). Algae production decreases from Glen to Marble Canyon and is even lower in the Grand Canyon (Hall et al. 2010) because of the increasing suspended sediment loads that reduced light availability (Kennedy et al. 2013). *Cladophora* grows best in continuously submerged clear-water stable habitats, whereas *Oscillatoria* forms dense mat-like matrices of filaments and sand in the varial zone and other habitats with high suspended sediments that are more typical of many southwestern streams (Shaver et al. 1997).

Oscillatoria tends to colonize relatively early in disturbed or newly inundated zones, while colonization by *Cladophora* is reduced or occurs more slowly (McKinney et al. 1997). As *Oscillatoria* supports tenfold fewer invertebrates than *Cladophora*, the input of terrestrially derived carbon has become vital to support the aquatic food base organisms. However, the leaves of the common nonnative *Tamarix ramosissima* along the river are an inferior food source for macroinvertebrates due to their high tannin content and slower decomposition rate compared to leaves of native cottonwoods and willows (Bailey et al. 2001).

Zooplankton is an important food resource for larval and juvenile native fish in the Colorado River system. The zooplankton found in regulated rivers is composed of both plankton derived from the reservoir (lentic species) and those derived from the streambed, backwaters, and tributaries of the river (lotic species) (Haury 1986). Lotic zooplankton and detritus are positively correlated with distance downriver from Glen Canyon Dam, increased discharge, and nearshore versus midchannel locations. Lentic zooplankton abundance also increases at higher discharges and in nearshore habitats, but is negatively correlated with distance downriver. It is possible that lentic zooplankton cannot survive and reproduce under the cold temperatures in the mainstem, although nearshore habitats provide a more stable environment than the mainstem, which may enhance rearing and development of lentic zooplankton (Benenati et al. 2001). Copepods are the most abundant zooplankton species in the Colorado River (AZGFD 1996). The biomass, productivity, and abundance of zooplankton (cladocerans, copepods, and ostracods originating primarily from Lake Powell) are highest in the Glen Canyon reach and drop sharply downstream (Tables F-2 through F-4).

There is evidence that Lake Powell zooplankton can survive downstream passage to Diamond Creek with only a small mortality rate due to abrasion. Thus, the zooplankton derived from Lake Powell has the potential of contributing to the aquatic food base throughout the river system to Lake Mead (Haury 1986). However, the Colorado River between Lake Powell and Lake Mead has a highly constricted channel and for the most part lacks backwaters of any significant area, which may account for the limited importance of zooplankton drift in the river (Blinn et al. 1995). Zooplankton may also have an affinity for *Cladophora* and other algae in the Glen Canyon reach, and are consumed by macroinvertebrates and fish in that reach (Benenati et al. 2001). These factors may also account for the diminished importance of zooplankton in the Marble and Grand Canyon reaches.

Generally, the responses of macroinvertebrates downstream of dams depend largely on the depth of the reservoir, the depth from which water is drawn, and on the ratio of low to high discharges (Jones 2013). Information on macroinvertebrates collected before and after closure of the Flaming Gorge Dam on the Green River in northeastern Utah is applicable to events that may have occurred in the Colorado River below Glen Canyon Dam (Blinn and Cole 1991; Pearson et al. 1968). Following closure of the Flaming Gorge Dam, macroinvertebrate genera declined from >70 to <30, while the mean macroinvertebrate abundance increased from 1,000 to 10,000/m^2 (Vinson 2001). Mayflies declined from 30 species to a single common species and two rare species. Midges and blackflies were the only other common post-dam insect taxa (Vinson 2001). Colonization of tailwaters by insects can be somewhat limited by lack of drift and small downstream insect population sizes that may limit recruitment from upstream flying adults (Vinson 2001).

River regulation by Glen Canyon Dam decreases turbidity in the tailwaters and permits increased algae growth on bottom substrates (Angradi 1994; Shannon et al. 1994), leading to an increased expansion of macroinvertebrate populations in the tailwater reach of Glen Canyon Dam (Blinn et al. 1992; Stevens et al. 1997). Algae biomass and production decrease downstream as water clarity decreases (Carothers and Brown 1991; Stevens et al. 1997; Hall et al. 2010). As is evident in Table F-1, this drives a downstream decrease in aquatic invertebrate biomass (e.g., *Gammarus*, midges, snails, and aquatic worms) (Carothers and Brown 1991; Stevens et al. 1997; Kennedy and Gloss 2005; Rosi-Marshall et al. 2010).

Various studies in the 1990s demonstrated that over 80% of the invertebrate biomass below Glen Canyon Dam was composed of *Gammarus*, midges, aquatic worms, and snails, many of which graze on epiphytes and other fine particulate matter (Blinn et al. 1998). Predation on insect eggs (e.g., by *Gammarus*) may contribute to the absence of mayflies and stoneflies below dams (Vinson 2001). In Glen Canyon, blackflies and midges support more than half of rainbow trout (*Oncorhynchus mykiss*) production but represent under 10% of total invertebrate production and abundance (Tables F-3 and F-4) (Kennedy et al. 2013). Midges and blackflies dominate invertebrate production in Marble and Grand Canyons (Table F-2); cobble bars are “hotspots” for midge and blackfly production (e.g., 2 to 10 times higher than other habitat types) (Kennedy et al. 2013). New Zealand mudsnails (*Potamopyrgus antipodarum*), *Gammarus*, aquatic worms, and midges dominate the current composition of the benthic community at Lees Ferry (Table F-3). In cobble substrates, New Zealand mudsnails and aquatic worms dominate the benthic biomass. They also dominate depositional habitats, although these areas tend to support lower benthic biomass (Cross et al. 2013). *Gammarus* dominates talus slopes and cliff faces, but these habitats generally have the lowest benthic biomass in the Lees Ferry reach. Blackflies (*Simulium arcticum*) are present in the Lees Ferry reach, but their biomass and abundance are generally low (Tables F-2 and F-4).

Cold water temperatures and daily fluctuations in discharge associated with hydropower production are likely responsible for the low diversity and abundance of aquatic insects downstream of the Paria River (Stevens et al. 1997; Kennedy et al. 2016). The decrease in stream clarity lowers primary production and favors the growth of the less nutritious *Oscillatoria* in the

TABLE F-1 Average Mean Habitat-Weighted Invertebrate Biomass at Select Sites in the Colorado River, July 2006–June 2009

	Habitat-Weighted Biomass (mg AFDM/m^2)[a]					
Taxon	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Acari (water mite)	<1	1	<1	<1	1	<1
Blephariceridae (net-winged midge)	0	0	0	2	<1	0
Ceratopogonidae (biting midge)	<1	<1	0	<1	<1	<1
Chironomidae (midge or chironomid)	163	113	58	30	43	45
Cladocera (water flea)	5	<1	<1	0	<1	0
Collembola (springtail)	0	<1	0	0	<1	0
Copepoda (copepod)	4	<1	<1	<1	<1	<1
Corixidae (water boatman)	0	0	0	<1	<1	0
Elmidae (riffle beetle)	0	<1	<1	6	3	3
Empididae (dagger or balloon fly)	0	<1	2	<1	1	<1
Ephemeroptera (mayfly)	0	0	0	0	<1	0
Gammarus (scud)	1,053	30	5	2	3	5
Hydropsychidae (net-spinning caddisfly)	0	0	2	3	2	<1
Hydroptilidae (microcaddisfly)	0	16	30	0	8	9
Molophilus (crane fly)	0	0	0	<1	<1	0
Nematoda (roundworm)	15	13	2	<1	<1	<1
New Zealand mudsnail	3,170	45	3	35	8	7
Oligochaetes (earthworm/bloodworm)	2,077	218	65	28	42	25
Ostracoda (seed shrimp)	25	<1	0	<1	<1	0
Physidae (bladder snail)	122	<1	0	1	<1	<1
Planariidae (planarian or flatworm)	114	9	<1	<1	<1	<1
Pyralidae (snout moth)	0	<1	0	<1	<1	3
Rhyacophilidae (free-living caddisfly)	0	42	0	0	<1	0
Simuliidae (blackfly)	100	858	35	50	49	44
Sphaeriidae (pea or fingernail clam)	14	<1	0	0	0	0
Zygoptera (damselfly)	0	0	0	0	<1	0
Total	6,862	1,345	202	155	160	141

[a] AFDM = ash-free dry mass. Site 1 is 16 mi downstream of Glen Canyon Dam (GCD, upstream of Paria River confluence), Site 2 is 45 mi downstream of GCD (downstream of Little Colorado River confluence), Site 3 is 78 mi downstream of GCD, Site 4 is 142 mi downstream of GCD, Site 5 is 180 mi downstream of GCD, and Site 6 is 240 mi downstream of GCD (upstream of Diamond Creek confluence).

[b] Biomass values <0.1 mg AFDM/m^2 for a taxon not included in total productivity value.

Source: Cross et al. (2011, 2013).

TABLE F-2 Average Mean Habitat-Weighted Invertebrate Production at Select Sites in the Colorado River, July 2006–June 2009

	Habitat-Weighted Productivity (mg AFDM/m^2/yr)[a]					
Taxon	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Acari (water mite)	0.3	5.0	4.9	3.7	7.0	4.5
Blephariceridae (net-winged midge)	0.0	0.0	0.0	12.4	1.6	0.0
Ceratopogonidae (biting midge)	0.3	4.1	1.4	0.2	1.1	3.6
Chironomidae (midge or chironomid)	717.6	1,103.2	549.0	437.8	634.6	575.7
Cladocera (water flea)	45.2	0.3	<0.1	0.0	<0.1	0.0
Collembola (springtail)	0.0	<0. 1	0.0	0.0	<0.1	0.0
Copepoda (copepod)	33.8	2.2	0.7	<0.1	1.6	0.5
Corixidae (water boatman)	0.0	0.0	0.0	<0.1	<0.1	0.0
Elmidae (riffle beetle)	0.0	2.3	2.6	24.9	17.7	18.3
Empididae (dagger or balloon fly)	0.0	7.3	11.7	4.4	12.2	6.7
Ephemeroptera (mayfly)	0.0	0.0	0.0	0.0	0. 7	0.0
Gammarus (scud)	6,113.8	129.1	18.1	13.5	36.7	68.9
Hydropsychidae (net-spinning caddisfly)	0.0	0.0	15.3	0.0	11.5	0. 7
Hydroptilidae (microcaddisfly)	0.0	134.0	159.2	17.9	41.9	58.6
Molophilus (crane fly)	0.0	0.0	0.0	0. 5	2.2	0.0
Nematoda (roundworm)	152.6	133.5	24.9	6.6	11.6	8.1
New Zealand mudsnail	8,637.0	74.4	8.3	27.8	32.1	27.5
Oligochaetes (earthworm/ bloodworm)	6,019.5	753.3	249.3	86.0	158.6	121.6
Ostracoda (seed shrimp)	124.9	0.3	0.0	<0.1	<0.1	0.0
Physidae (bladder snail)	690.2	4.0	0.0	8.2	0.6	4.6
Planariidae (planarian or flatworm)	571.2	45.9	1.2	0.1	4.2	0.8
Pyralidae (snout moth)	0.0	11.0	0.0	<0.1	0.5	19.0
Rhyacophilidae (free-living caddisfly)	0.0	316.4	0.0	0.0	3.8	0.0
Simuliidae (blackfly)	539.4	5,240.8	266.3	367.2	540.2	488.0
Sphaeriidae (pea or fingernail clam)	69.0	<0.1	0.0	0.0	0.0	0.0
Zygoptera (damselfly)	0.0	0.0	0.0	<0.1	0.0	0.0
Total[b]	23,714.8	7,967.1	1,312.9	1,011.2	1,520.4	1,407.1

[a] AFDM = ash-free dry mass. Site 1 is 16 mi downstream of Glen Canyon Dam (GCD, upstream of Paria River confluence), Site 2 is 45 mi downstream of GCD (downstream of Little Colorado River confluence), Site 3 is 78 mi downstream of GCD, Site 4 is 142 mi downstream of GCD, Site 5 is 180 mi downstream of GCD, and Site 6 is 240 mi downstream of GCD (upstream of Diamond Creek confluence).

[b] Productivity values <0.1 mg AFDM/m^2/yr for a taxon not included in total productivity value.

Source: Cross et al. (2011, 2013).

TABLE F-3 Average Mean Habitat-Weighted Invertebrate Abundance at Select Sites in the Colorado River, July 2006–June 2009

	Habitat-Weighted Abundance (number/m^2)[a]					
Taxon	Site 1	Site 2	Site 3	Site 4	Site 5	Site 6
Acari (water mite)	20	455	423	324	562	318
Blephariceridae (net-winged midge)	0	0	0	<1	<1	0
Ceratopogonidae (biting midge)	2	52	11	1	25	31
Chironomidae (midge or chironomid)	6,814	9,814	4,602	2,929	3,716	3,172
Cladocera (water flea)	2,497	13	2	0	3	0
Collembola (springtail)	0	<1	0	0	<1	0
Copepoda (copepod)	4,973	404	118	2	174	62
Corixidae (water boatman)	0	0	0	<1	<1	0
Elmidae (riffle beetle)	0	48	38	41	41	19
Empididae (dagger or balloon fly)	0	5	21	11	25	5
Ephemeroptera (mayfly)	0	0	0	0	1	0
Gammarus (scud)	2,930	50	11	8	20	33
Hydropsychidae (net-spinning caddisfly)	0	0	5	0	1	<1
Hydroptilidae (microcaddisfly)	0	42	81	17	22	20
Molophilus (crane fly)	0	0	0	<1	<1	0
Nematoda (roundworm)	1,199	1,846	276	67	113	62
New Zealand mudsnail	74,033	382	110	229	187	120
Oligochaetes (earthworm/bloodworm)	32,988	7,270	2,774	533	1,117	922
Ostracoda (seed shrimp)	1,023	4	0	<1	3	0
Physidae (bladder snail)	279	<1	0	4	4	2
Planariidae (planarian or flatworm)	987	81	5	<1	21	2
Pyralidae (snout moth)	0	<1	0	<1	1	5
Rhyacophilidae (free-living caddisfly)	0	15	0	0	3	0
Simuliidae (blackfly)	419	3,180	316	327	476	352
Sphaeriidae (pea or fingernail clam)	122	<1	0	0	0	0
Zygoptera (damselfly)	0	0	0	0	<1	0
Total[b]	128,286	23,661	8,793	4,493	6,515	5,125

[a] AFDM = ash-free dry mass. Site 1 is 16 mi downstream of Glen Canyon Dam (GCD, upstream of Paria River confluence), Site 2 is 45 mi downstream of GCD (downstream of Little Colorado River confluence), Site 3 is 78 mi downstream of GCD, Site 4 is 142 mi downstream of GCD, Site 5 is 180 mi downstream of GCD, and Site 6 is 240 mi downstream of GCD (upstream of Diamond Creek confluence).

[b] Abundance values <0.1/m^2 for a taxon not included in total productivity value.

Source: Cross et al. (2011, 2013).

TABLE F-4 Distribution, Ecological Importance, and Favorable Temperature Range for Select Primary Producers

Taxa	Distribution	Ecological Importance	Favorable Temperature Range
Upright epiphytic diatoms	Throughout the river, but most abundant in the Glen Canyon Dam–Paria River reach	High, easily consumed by grazers	10–15°C (50–59°F)
Adnate epiphytic diatoms	Throughout the river, but most abundant in the Glen Canyon Dam–Paria River reach	Medium, not as easily consumed by grazers	15–20°C (59–68°F)
Cladophora glomerata	Throughout the river, but most abundant in the Glen Canyon reach	High, substrate for epiphytic diatoms	13–17°C (55–63°F)
Oscillatoria spp.	Throughout the river, but most abundant below Little Colorado River	Low, not generally consumed directly, poor substrate for diatoms	18–21°C (64–70°F)
Egeria densa	Glen Canyon reach	Medium-high, substrate for epiphytic diatoms, cover for fish	15–21°C (59–70°F)
Potamogeton spp.	Throughout the river, but most abundant in the Glen Canyon reach	Medium-high, substrate for epiphytic diatoms, cover for fish	20–22°C (68–72°F)
Fontinalis spp.	Glen Canyon reach	Medium-high, secondary substrate for epiphytic diatoms	10–15°C (50–59°F)
Chara spp.	Throughout the river, but most abundant in the Glen Canyon reach	Medium-high, substrate for epiphytic diatoms, cover for fish	18–25°C (64–77°F)

Source: Valdez and Speas (2007) and references cited therein.

lower reaches of the Colorado River (Blinn et al. 1999). Macroinvertebrates are not generally associated with *Oscillatoria* because it is very compact, has little surface area for colonization, and largely lacks epiphytic diatoms (Blinn et al. 1995). For example, *Gammarus*, a major food source for trout and other fishes, prefers *Cladophora* due to its epiphytic diatoms. This relationship is strong from Glen Canyon Dam to Lees Ferry and weak from Lees Ferry to Diamond Creek (Patten 1998). This relationship does not exist between *Gammarus* and *Oscillatoria*. If *Cladophora* declines, then the contribution of *Gammarus* to the aquatic food base also declines, except perhaps in the drift (Patten 1998).

Drifting macroinvertebrates provide an important food resource for rainbow trout and other native and nonnative fish species. Flow regime, discharge, and distance from the dam influence drift of macroinvertebrates in the Colorado River (Shannon et al. 1996; Stevens et al. 1998; Sublette et al. 1998; McKinney et al. 1999). In general, a positive correlation exists between stream drift and discharge; however, reduced flows can increase stream drift through behavioral factors such as crowding, reduced oxygen concentrations, and avoidance of desiccation (Blinn et al. 1995). Kennedy et al. (2014) concluded that benthic density is the primary control on drift concentrations over long timescales (e.g., weeks to months), because increased benthic production will also increase drift. In contrast, changes in flow such as those occurring from hydropeaking have an important control on drift concentrations, but primarily on a shorter timescale (e.g., days) (Kennedy et al. 2014).

Tributary and terrestrial insects comprise a small portion of the stream drift in the Colorado River (Shannon et al. 1996). It is possible that terrestrial invertebrate drift increases during and immediately after rainstorms and is therefore an uncommon but locally important resource for river fishes through Grand Canyon (Shannon et al. 1996). Terrestrial and tributary insects contribute <0.001 and <0.1% of the total invertebrate biomass in the mainstem drift, respectively (Blinn et al. 1995). Fish production throughout Glen and Grand Canyon appears to be limited by the availability of midges and blackflies, and fish may exert top-down control over them (Carlisle et al. 2012). While blackflies and midges support between 43 and 50% of trout production, they only comprise a small percentage of total invertebrate secondary production and abundance in the Glen Canyon reach (Tables F-3 and F-4) (Cross et al. 2011).

Generally, physical, chemical, and biological attributes in the lower reaches of the Colorado River peak at or immediately downstream of tributaries. The connection between tributaries and the mainstem is important for the flow of nutrients, sediment, and wood that contribute to habitat heterogeneity and biodiversity in the mainstem (Kiffney et al. 2006). However, tributary sediment inputs can limit light availability and reduce algal production, thereby reducing food for aquatic invertebrates. High sediment loads may also limit the ability of fish to see their prey (Coggins and Yard 2011).

The Colorado River between Glen Canyon Dam and Lake Mead contains more than 400 ephemeral and 40 perennial tributaries; however, as many of the Colorado River tributaries are dry except during heavy summer rains, they contribute little to the mainstem aquatic food base (Haury 1986). All of the tributary streams have a natural seasonal range of temperatures and discharges unaffected by Glen Canyon Dam (NPS 2005). Oberlin et al. (1999) indicated that primary productivity and detritus, the major food resource for macroinvertebrates, are higher overall in clear-water tributaries and highest in those originating inside the Grand Canyon. Phytoplankton species richness also increases in clear-water tributaries (Crayton and Sommerfield 1979; Oberlin et al. 1999), increasing primary productivity and food quality in those environments (Henery 2005).

Common macroinvertebrates in the tributary streams include caddisflies, mayflies, stoneflies, midges, and blackflies. Drift of tributary macroinvertebrates into the mainstem contributes, at least locally, to the aquatic food base in the Colorado River. Macroinvertebrate productivity and diversity in the tributaries are lowest in the spring and summer, as flash floods in these seasons disrupt the benthic invertebrate communities (Oberlin et al. 1999). Tributaries provide ≤25% of the total organic stream drift in the Colorado River through Grand Canyon National Park (Blinn et al. 1995).

Tributary streams with travertine deposits (e.g., Havasu Creek) or those dominated by bedrock (e.g., Matkatamiba Creek) have little inhabitable substrate for macroinvertebrates. Steep channel gradients and erosional habitat also limit benthos in some tributaries. Overall, standing biomass of the Little Colorado River macroinvertebrate community was an order of magnitude lower (0.056 g/m^2 ash-free dry mass [AFDM]) than at the confluence with the Colorado River

(0.25 g/m^2 AFDM). A high discharge with increased suspended sediment loads negatively affects macroinvertebrate biomass (Haden et al. 1999). Even extended periods of base flow (which tends to increase macroinvertebrates and algae) do not increase productivity for the areas of the Little Colorado River that contain most of the river's humpback chub population (Haden et al. 1999).

Some of the tributaries also contain New Zealand mudsnails, probably spread by recreationists (NPS 2005). Shannon et al. (2003a) reported the New Zealand mudsnail in cobble bars of 5 of 18 tributaries they sampled, but the snail did not extend more than 32 m upstream in those streams. Since New Zealand mudsnails prefer habitats with constant temperatures and flows and high primary productivity, flash floods may diminish their long-term establishment in tributaries (Shannon et al. 2003a). The risk of quagga mussel (*Dreissena rostriformis bugensis*) introduction to tributaries also appears low. Reservoirs on the upper reaches of the Little Colorado River may eventually support the quagga mussel. Establishment of the quagga mussel in many of the tributaries is unlikely due to high summer water temperatures above the mussel's upper lethal limit (Kennedy 2007).

F.2.1.3 Influence of New Zealand Mudsnail on the Aquatic Food Base

In addition to changes brought about by Glen Canyon Dam, the loss of native species and the addition of numerous nonnative species modified the aquatic ecosystem of the Colorado River within Grand Canyon (Johnson and Carothers 1987). This applies to the aquatic food base and fish species. Nonnative species, including those intentionally introduced, are often better competitors in the homogeneous habitats of regulated rivers (Stanford et al. 1996). To date, nonnative periphyton and rooted aquatic macrophytes have not caused adverse impacts on the aquatic food base in the Colorado River below Glen Canyon Dam. Section 4.17.3.4 discusses potential impacts that could occur if the diatom *Didymosphenia geminata* ("didymo") becomes established in the Colorado River.

The New Zealand mudsnail can tolerate a wide range of water temperatures (except freezing), salinity, and turbidity. It can also withstand short periods of desiccation. Densities of New Zealand mudsnail are usually highest in systems with high primary productivity, constant temperatures, and constant flow. It occurs in all types of aquatic habitats, from eutrophic mud bottom ponds to clear rocky streams (USGS 2002; Sorensen 2010). Fitness of the New Zealand mudsnail peaks at 18°C (64°F), declining at cooler and warmer temperatures (NZMMCPWG 2007).

Numerous adaptations of the New Zealand mudsnail aid its spread within watersheds. For instance, adults can pass through the digestive systems of some fish species alive, adults can float on masses of algae, and juveniles can float freely on the water surface (Kerans et al. 2005; NZMMCPWG 2007). While the New Zealand mudsnail is not common in streams prone to periods of sediment-moving flood flows, its tough shell, small size, and hydrodynamic shape make it likely to survive scouring flows (Holomuzuki and Biggs 2006; NZMMCPWG 2007). Most New Zealand mudsnails in North America are asexually reproducing females that are born with developing embryos already present in their reproductive system (Sorensen 2010). Clonal reproduction increases the probability of success of introduction as only a single female can establish a new population (NZMMCPWG 2007). One female can carry up to 20 embryos and under proper conditions may account for over one million snails within one year (Shannon et al. 2003a).

Vinson et al. (2007) suggest that the New Zealand mudsnail introduction into the Green River below Flaming Gorge Dam may have led to a decline of total invertebrate abundance by over 25%. They concluded that decreases in mayflies due to competition from New Zealand mudsnails may jeopardize mayfly recolonization of the Green River following implementation of a more natural springtime flood flow regime. Where the New Zealand mudsnail dominates invertebrate production, it could become the dominant forage base for fishes that prey on macroinvertebrates (Vinson and Baker 2008). Field survey data in the Green River below Flaming Gorge Dam showed a sharp annual increase in the number of brown trout (*Salmo trutta*) and rainbow trout that consumed New Zealand mudsnail between 2001 and 2005 (Vinson and Baker 2008). Bioenergetic simulations suggest that fish diets high in New Zealand mudsnail would not meet energy requirements of fish, resulting in reduced growth and weight loss (Vinson et al. 2007; Vinson and Baker 2008) as discussed previously. For example, when the New Zealand mudsnail comprised between 71 and 81% of brown trout diet, the trout did not gain weight, and when the diet consisted of more than 81% New Zealand mudsnails, brown trout lost weight. Rainbow trout fed a diet of 42% New Zealand mudsnail began losing weight over the course of the experimental study (Harju 2007).

The New Zealand mudsnail represents a trophic dead end in Glen Canyon because it has a high production and consumption of primary producers, but it does not support a substantial amount of production for higher trophic levels (i.e., fish). Minnows and suckers that possess pharyngeal teeth may be capable of consuming and crushing the shells of New Zealand mudsnails (NZMMCPWG 2007). However, the New Zealand mudsnail offers little or no energy compared to other common food items in those fish successful in crushing its shell (Ryan 1982).

F.2.2 Impacts of LTEMP Alternatives on the Aquatic Food Base

The desired future conditions (DFCs) for the Colorado River ecosystem domain (Appendix A of the EIS) include these two DFCs for the aquatic food base goals:

- The aquatic food base will sustainably support viable populations of desired species at all trophic levels.
- Assure that an adequate, diverse, productive aquatic food base exists for fish and other aquatic and terrestrial species that depend on those food resources.

Attaining these DFCs while meeting existing water delivery requirements is complex. Biological resources in regulated rivers are subject to a number of spatial and temporal changes in conditions downstream from a dam: reductions in seasonal flow variability, alterations in the timing of extreme flow events, pulses in flow during periods of peak power demands, reduced turbidity and increased water clarity, diel and seasonal constancy of water temperatures, armoring of substrates in the tailwaters, modified nutrient regimes, and the appearance of lentic plankton below the reservoir (Blinn and Cole 1991; Blinn et al. 1995; McKinney and Persons 1999). Flow and temperature are the two major factors that influence the condition and availability of the aquatic food base in the Colorado River between Glen Canyon Dam and Lake Mead. The following discussion supplements the analyses presented in Sections 4.5.2.1 and 4.5.3 of the EIS.

F.2.2.1 Flow Effects on the Aquatic Food Base

Hydropeaking is the mode of hydroelectric generation that most alters the quantity and quality of habitats available to aquatic organisms. Effects can be direct (e.g., stranding, mortality, and habitat loss) or indirect (e.g., downstream displacement, depleted food production, increased stress) (Clarke et al. 2008). Impacts of flow fluctuations are typically greatest within the tailwaters of a dam and decline with distance downstream due to flow attenuation and the increasing influence of tributaries (Clarke et al. 2008; Patterson and Smokorowski 2011). Flow attenuation occurs downstream of Glen Canyon Dam; however, because of the constrained nature of the channel through most of Marble Canyon and the Grand Canyon, flow fluctuations from dam releases are still apparent in the lower Grand Canyon near Lake Mead (see Section 3.2.1.2 of the EIS). The following sections discuss flow effects on the aquatic food base with respect to the elements of base operations, adjustments of base operations, and trout management actions for the LTEMP alternatives.

Effects of Base Operations

Potential alternative-specific effects of base operations (i.e., operations in those years when no condition-dependent or experimental actions are triggered) on the aquatic food base depend on the differences in the monthly pattern in release volumes, minimum and maximum flows, daily flow ranges, and ramp rates. Monthly increases in release volumes may increase the permanently wetted zone, which could increase benthic production, if the increased monthly flows last long enough for benthic development to occur (e.g., weeks to months). Months of higher release volumes would also improve hydraulic connectivity with and maintenance of backwater habitats. Backwaters with more permanency potentially support increased planktonic and benthic communities. A decrease in the permanently wetted zone would occur when decreases in monthly release volumes occur (Reclamation 1995; Hoffnagle 2001; Melis et al. 2006; Behn et al. 2010). Pools of water left after high spring or summer flow months potentially provide habitat for mosquitoes (Blinn et al. 1995). While mosquitoes may contribute to the aquatic and terrestrial food base, they pose a potential health concern to humans.

Daily minimum flow is an important determinant of benthic standing crop because of the strong negative effects of desiccation on algae and invertebrates (Melis et al. 2006). Periods of low steady summer/fall releases (e.g., 5,000 to 8,000 cfs) are expected to result in warmer and more stable nearshore and backwater habitats and longitudinal river warming, while similar flows in winter are likely to produce greater overwinter algal and macroinvertebrate production (Blinn et al. 1995; Valdez et al. 2000). Wet channel area in low-angle habitats within the Glen

Canyon reach is reduced by about 10% at 5,000 cfs compared to 8,000 cfs. This area reduction consists of about 16 ha; however, the effects on the aquatic food base from this habitat reduction may not be detectable (Melis et al. 2014).

Restricted minimum and maximum flows and reduced ramping rates of the Modified Low Fluctuating Flow regime adopted in the Glen Canyon Dam Record of Decision (Reclamation 1996) were intended to stabilize the area available for colonization by benthic algae, thereby decreasing losses through desiccation or freezing while increasing primary and secondary production (Blinn et al. 1995). Midges, blackflies, and *Gammarus* were not observed in the varial zone above the 10,600-cfs stage (Blinn et al. 1995).

The interactions between cycles of inundation and dewatering in varial zones play a major role in structuring algal communities in regulated desert rivers (Blinn et al. 1998). Periodic exposure of nearshore and backwater habitats can result in loss of invertebrates and primary producers through desiccation, while inundation can impact the aquatic food base through sediment deposition (Valdez et al. 1998). Atmospheric exposure of benthos can be more severe than flooding because organisms are directly killed rather than displaced or buried (Blinn et al. 1995). Fluctuating flows (>10,000 cfs/day) can fragment *Cladophora* from its basal attachment and increase its occurrence in the drift. Consuming drifting *Cladophora* (with its attached epiphytes and any invertebrates) allows rainbow trout to expend less energy in searching for food (Leibfried and Blinn 1987). Daily range in flows >10,000 cfs only occur during December and January (12,000 cfs) for Alternative B.

A stabilized discharge regime could increase algae production downstream of Glen Canyon Dam. In turn, this may have positive effects on invertebrate and fish production (Kennedy et al. 2013). Basal holdfasts of *Cladophora* can dry following periods of exposure as short as 4 hours in summer (Pinney 1991). Exposure to subzero winter air temperatures for only one night resulted in ≥50% loss of chlorophyll a and mass of *Cladophora* (Blinn et al. 1995). Recovery time may take several months (Blinn et al. 1992). Since algal communities provide the dominant food resource below dams, restricting the extent of the varial zone and maintaining wetted perimeter can be important to maintaining the overall food base (Blinn et al. 1998). Potential differences among alternatives based on daily range in flows are provided in Section 4.5.3 of the EIS.

Warm or subfreezing air temperatures could cause mortality of invertebrates stranded in the varial zone (Gislason 1985). The varial zone provides poor habitat for species with multiple life history stages (Jones 2013) by dewatering of emergence and oviposition sites (Vinson 2001; Kennedy et al. 2016). High rates of egg mortality due to exposure may partially explain the rarity of mayflies in the Colorado River. For example, adult female *Baetis* species land on rocks protruding from the water surface and then crawl underwater to lay their eggs on the underside of the rocks. These rocks may become dry for possibly 12 hours during the hydropeaking cycle, causing egg mortality (Kennedy et al. 2016). The fact that midges and blackflies have more generalized egg laying behaviors that are not strictly dependent on river edges may explain why they are the predominant insects in the mainstem. Nevertheless, hydropeaking still appears to limit egg survival and thus recruitment for midges in the mainstem (Kennedy et al. 2016). In the Glen Canyon Dam tailwaters, *Gammarus* standing stock and fecundity are lower, seasonal recruitment of young is briefer, and fewer young are recruited into the population in the varial zone compared to the permanently wetted zone. In addition, *Gammarus* mortality increases in the varial zone (Angradi and Kubly 1993; Ayers and McKinney 1996; Ayers et al. 1998).

Invertebrates are continually moving and drifting to different positions in the river, thus stranding of a significant number of invertebrates in the varial zone would reduce the overall abundance in the river including that in the permanently wetted zone (Smokorowski 2010). However, there may be little colonization of shoreline areas during daily high flows, and as discharge decreases, large numbers of insects may not be present to enter the drift from areas being dewatered (Perry and Perry 1986). Nevertheless, reduction in the amplitude and duration of power peaking flow fluctuations may be an effective management strategy for enhancing aquatic insect biomass with the potential for increasing the survival and growth of fishes dependent on them (Gislason 1985).

Flow fluctuations may increase the amount of organisms available to drift-feeding fish, although this may only occur for a short period (e.g., a few days or less), depending on the density and replacement capacity of benthic invertebrates. For example, a twofold daily variation in discharge resulted in a greater than tenfold increase in drift concentrations of *Gammarus* and New Zealand mudsnails while blackfly drift concentrations decreased by more than 80% as discharge doubled. Midge drift concentrations increased proportional to discharge (Kennedy et al. 2014).

As the daily range in flows increases, there is greater divergence in habitat conditions between low and high flows, and there will likely be fewer taxa that can withstand such variability. Consequently, the ratio of the regulated high and low flows may become as important as the base flow as an influencing factor determining biotic composition (Jones 2013). Ramping rate restrictions may allow sufficient time for aquatic macroinvertebrates to respond to daily flow fluctuations (Patterson and Smokorowski 2011). Rapid up-ramping can result in rapid increases in shear stress, potentially causing catastrophic drift or the large-scale displacement of invertebrates from the sediment (Gibbins et al. 2007). Perry and Perry (1986) observed a greater number of aquatic invertebrates stranded when the down-ramping rate was rapid; indicating that unlimited down-ramping is a potential cause of increased invertebrate mortality (Smokorowski 2010). In addition, high ramping rates potentially favor adnate diatom species over the more upright species, the latter of which macroinvertebrates and fishes more readily consumed (Hardwick et al. 1992; Pinney 1991; Biggs 1996).

Miller and Judson (2014) observed that, during a daily hydropeaking schedule, macroinvertebrate drift biomass below Flaming Gorge Dam in the Green River increased during the rising limb of the daily hydrograph and declined prior to the cessation of the peak. Macroinvertebrate drift increases were correlated with the biomass of drifting vegetation. As the study by Miller and Judson (2014) occurred over winter, the rate of vegetation export declined over time due to senescence caused by decreased light levels and cooler temperatures. This at least partly accounted for the observed declines in macroinvertebrate drift after 30 to 60 days (Miller and Judson 2014).

During base operations, up-ramping rates are the same at 4,000 cfs for all alternatives except for Alternatives F and G that would not have a daily range in flows. Down-ramping rates would be highest for Alternative B (4,000 cfs for November through March and 3,000 cfs in the other months), followed by the Alternatives C, D, and E (2,500 cfs) and Alternative A (1,500 cfs). Alternatives F and G both feature steady flows in all months.

Experimental Treatments

High-Flow Experiments. Most experimental adjustments of base operations relate to high-flow experiments (HFEs). The existing HFE protocol calls for spring HFEs to occur in March–April and fall HFEs to occur in October–November, with magnitudes ranging from 31,500 to 45,000 cfs (Reclamation 2011a). Most HFEs would last from less than 1 hour to 96 hours, although HFEs longer than 96 hours could occur under Alternatives C, D, and G. There is a potential for more than one HFE to occur within the same year or between years, with a potential for up to 40 HFEs during the LTEMP period (Alternatives C, F, and G). Food webs close to Glen Canyon Dam are more energy inefficient and are expected to exhibit lower resistance to experimental flood perturbations compared to food webs downstream of major tributaries (Cross et al. 2013).

HFEs conducted in the spring and fall represent contrasting conditions, particularly with regard to light, temperature, and invertebrate biomass. Plant and macroinvertebrate recovery times may be shorter for spring HFEs than for fall HFEs as a result of longer day lengths and warmer river temperatures in spring and summer. Spring HFEs can cause ponding of tributary flows that enter the Colorado River, creating temperature refuge areas within the mainstem (Valdez et al. 2000). Spring HFEs can also re-suspend organic material stored along the shoreline and redistribute it into the mainstem (Valdez et al. 2000). The majority of the aquatic food base taxa would recover within 1 to 4 months after a spring HFE as observed for the spring 1996 and 2008 HFEs (Blinn et al. 1999; Rosi-Marshall et al. 2010), although some taxa may recover more slowly (Cross et al. 2011). Shannon et al. (2001) reported high rates of invertebrate drift for 2 months after the spring 1996 HFE. A post-flood increase in production and drift of midges and blackflies is expected after a spring HFE (Cross et al. 2011), likely due to the flushing of fines in the interstitial spaces between gravel and around macroalgae holdfasts used by these invertebrates for cover. *Gammarus* is expected to be slower to recover because of its greater susceptibility to being exported by river currents than most invertebrate species (Reclamation 2011a). In addition, slow-growing taxa such as *Gammarus* take longer to recover to pre-flood levels relative to faster growing taxa with aerial life stages such as blackflies and midges (Robinson and Uehlinger 2008).

Fall HFEs precede winter months of minimal insolation, low temperatures, and reduced gross primary productivity. Thus, recovery times for aquatic food base organisms take longer than for spring HFEs (Melis et al. 2006). Following the fall 2004 HFE, *Gammarus* was extremely scarce for many months in Lees Ferry (Melis et al. 2006). Even longer recovery times could occur if a fall HFE is followed by a spring HFE. The 4 to 5 months between a fall and spring HFE could preclude full recovery of most benthic invertebrate assemblages. The following spring HFE could scour the remaining primary producers and susceptible invertebrates and further delay recovery. A spring HFE followed by a fall HFE may not have as great an impact because of the rapid recovery of the food base expected over the summer (Reclamation 2011a).

The 2008 spring HFE reduced annual invertebrate production in the Lees Ferry tailwater by >50%, driven primarily by significant reductions in production of New Zealand mudsnails and *Gammarus*. Large numbers of *Gammarus* dislodged during high flows are transported considerable distances downstream, making them available to fishes. Windrows of stranded *Gammarus* carcasses along some shorelines following the spring 1996 controlled flood (Valdez 1999) became available to terrestrial consumers such as shorebirds, lizards, and spiders. Reductions in mudsnails and *Gammarus* persisted at least 15 months after the HFE (when the study concluded) and coincided with a significant decline in the annual production of these taxa (e.g., New Zealand mudsnail production declined from 11 to 13 g AFDM/m^2/yr to 2 g AFDM/m^2/yr and *Gammarus* production from 7 to 8 g AFDM/m^2/yr to 3 g AFDM/m^2/yr). Reductions in aquatic worms recovered in about 4 to 6 months (Rosi-Marshall et al. 2010). However, midges and blackflies increased by 30 and 200%, respectively, in the year following the HFE, and they supported a 200% increase in rainbow trout production (Cross et al. 2011). During the flood, the concentrations of invertebrate prey available in the drift increased from an average of 0.093 mg/m^3 AFDM before the flood to an average 0.163 mg/m^3 after the flood (Rosi-Marshall et al. 2010; Cross et al. 2011).

The concentrations of midges and blackflies in the drift increased 400 and 800%, respectively, after the 2008 HFE, and this effect persisted for at least 15 months. Biomass and production of both groups also increased after the HFE (Rosi-Marshall et al. 2010; Cross et al. 2011). The March 2008 HFE resulted in an increase in the area of backwater habitat that persisted for at least two months, but returned to conditions similar to those before the HFE by about 6 months after the HFE (Melis 2011). In addition to scouring benthic algae and invertebrates, high flows can capture large quantities of terrestrial organic matter that may temporarily increase the amount of food base available for drift-feeding fish (Valdez and Hoffnagle 1999; Gloss et al. 2005).

Generally, more frequent HFEs may cause a shift to more resistant taxa or to new taxa that would colonize the river. However, if such taxa are not present, more frequent HFEs may reduce macroinvertebrate diversity and possibly abundance, resulting in a reduction in the aquatic food base (Reclamation 2011a). Any benefits from HFEs along downstream segments of the Colorado River (particularly the lower portion of the Grand Canyon reach) will likely be smaller in magnitude than in the Lees Ferry reach (Melis 2011; Kennedy et al. 2013). The average number of HFEs during the LTEMP period would be 39.3 under Alternative F; range from 17.1 to 24.5 under Alternatives C, D, E, and G; and be only 7.2 under Alternative B and 5.5 under Alternative A.

The most notable differences among the alternatives is for Alternative A, which would not have HFEs after 2020; Alternative B, which would not exceed one spring or fall HFE every other year; and Alternative E, which would not have spring HFEs during the first 10 years (Table 2-2 of the EIS).

A comprehensive study on the ecological effects of repeated HFEs in the River Spöl in Switzerland indicated that one or two high-flow events per year can enhance and sustain long-term ecological integrity (Scheurer and Molinari 2003) and that such releases must be repeated on a regular basis (annually) to maintain their benefits (Robinson and Uehlinger 2008). The first experimental flood in the River Spöl reduced macroinvertebrate abundance by about 50%. However, subsequent experimental floods had less effect, indicating that a new assemblage had established that was more resilient to flood disturbance. The response of macroinvertebrates to experimental floods occurs over a period of years, rather than months, as species composition adjusts to the new flow conditions (Robinson and Uehlinger 2008). Robinson et al. (2003) observed that the abundance of amphipods and planarians decreased while the abundance of baetid mayflies, blackflies and midges increased. Some mayfly, stonefly, and caddisfly taxa initially decreased in abundance but subsequently increased. The results of experimental floods in the River Spöl imply that the experimental flood regime needs maintaining to sustain the development of a more natural macroinvertebrate assemblage (Robinson et al. 2003). While three to five consecutive HFEs from Glen Canyon Dam may alter the aquatic food base composition, the absence of an HFE for one or more seasons might reset the current aquatic food base community (Reclamation 2011a). It is anticipated that, regardless of the HFE regimen, midges and blackflies would remain important components of the aquatic food base downstream of Glen Canyon Dam.

A large portion of the aquatic food base in the Lees Ferry reach would likely be scoured by an HFE of 41,000 to 45,000 cfs regardless of the time of year. The initial hydrostatic wave produces the scouring effect, and the duration of the flow is an important factor in transporting the material downstream (Rosi-Marshall et al. 2010). The HFE conducted in March 1996 (7-day discharge of 45,000 cfs) resulted in benthic scour and entrainment of both primary and secondary producers at all study sites along the 239-mi river corridor. Over 90% of the benthos was removed by the hydrostatic wave or within 24 hr from the start of the test flood. In addition, drift mass reached highest levels during the first 2 days of the HFE (an order of magnitude higher than under normal dam operations) and subsided after that period (Shannon et al. 2001). Recovery rates to pre-flood levels were fast for benthic algae (1 month) and invertebrates (2 months) (Blinn et al. 1999). Recovery of the macrophytes *Chara*, *Potamogeton*, and *Elodea* to pre-flood conditions took 1 to 7 months (Shannon et al. 2001).

It is hypothesized that mucilaginous algae found in miscellaneous algae, macrophytes, and bryophytes (MAMB) can outcompete *Cladophora* under the combination of reduce nutrient conditions and elevated discharge regimes of about 25,000 cfs. However, if discharge increases to 45,000 cfs or more, MAMB will scour, allowing *Cladophora* to recolonize regardless of nutrient conditions because of strong holdfast attachment and lack of competition (Benenati et al. 2000). HFEs up to 45,000 cfs may occur under Alternatives C, D, and G.

HFEs longer than 96 hours may also occur under Alternatives C, D, and G. These longer-duration HFEs could scour much of the aquatic food base, especially within the Glen Canyon reach, and reduce the standing crop of benthic invertebrates. The extended-duration HFEs may increase the aquatic food base available for drift-feeding fishes, particularly during the initial hours of the flood. An extended-duration HFE may also help to control the abundance of

New Zealand mudsnails in the Glen Canyon reach, but it may possibly contribute to their downstream abundance. Potential effects of sustained spring flows include:

- High, turbid main-channel flow and a surge of increased macroinvertebrate drift, increased feeding opportunities for non-sight feeders, and increased density of terrestrial invertebrates washed from shoreline.

- Rebuilt backwater habitats and increased primary and secondary production in backwaters following redistribution of organics (Valdez et al. 1998).

Steady Flows. Steady flows (or nearly steady flows with some instantaneous fluctuations associated with ancillary services) would occur prior to or following spring and/or fall HFEs (prior to spring and fall HFEs under Alternative C and before fall HFEs only for Alternatives D and E; Alternatives F and G already feature steady flows). Potential effects of steady flows include (1) warmer shoreline and backwaters and an increase in backwater production; (2) warmer main channel and an increase in primary and secondary production and potential for parasite maturation and proliferation; and (3) stable main channel and less turbidity and an increase in shoreline primary and secondary production and reduced macroinvertebrate drift as food for fish (Valdez et al. 1998). However, mainstem warming, particularly in the Glen Canyon and Marble Canyon reaches, would be limited. Ralston et al. (2007) observed that biological and physical parameters were unaffected by daily fluctuations in flow of 2,700 cfs and steady-flow releases. Reduced flow fluctuations prior to an HFE could increase production of primary producers and consumers. The HFE could increase drift biomass. Reduced flow fluctuations following an HFE could hasten benthic recolonization.

Low Summer Flows. Low summer flows may be tested under Alternatives C, D, and E; and are an annual component of Alternative F. Low summer flow tests would involve flows between 5,000 and 8,000 cfs to warm the Colorado River at the confluence with the Little Colorado River to at least 13°C (55°F), 14°C (57°F), or 16°C (61°F) under Alternatives C, D, and E, respectively. Dropping to low flows in the summer would necessitate increasing mean daily flows in other months relative to base operations. Low summer flow tests may increase primary and secondary benthic production but reduce macroinvertebrate drift. In particular, the density of New Zealand mudsnails may increase under low summer flows. However, the opposite conditions, compared to base operations without low summer flow tests, may occur in non-summer months. Potential impacts on the aquatic food base from low summer flows under Alternative F are described in Section 4.5.3.6 of the EIS.

Hydropower Improvement Flows. Hydropower improvement flows (increased fluctuation levels proposed as an experiment under Alternative B) would entail a daily change from a minimum flow of 5,000 cfs to a maximum flow of 15,000 to 25,000 cfs (depending on season). This could decrease primary and secondary production, although macroinvertebrate drift may increase. Down-ramp rates of 5,000 cfs/hr may increase stranding of organisms in the varial zone compared to base operations that range from 1,500 cfs/hr (Alternatives A, F, and G) to 4,000 cfs/hr (Alternative B from November through March). Conversely, higher up-ramp rates of 5,000 cfs/hr coupled with sustained high flows may flush increased amounts of terrestrial invertebrates (and other items such as leaf litter) from shoreline areas into the drift compared to base operations for all alternatives (up-ramp rates of 4,000 cfs/hr).

Sustained Low Flows for Benthic Invertebrate Production. An aquatic resource-related experiment unique to Alternative D would be to test the effects of macroinvertebrate production flows in May through August on benthic invertebrate production and diversity. It has been demonstrated that the large varial zone created by fluctuating flows limits recruitment of mayflies (order Ephemeroptera), stoneflies (order Plecoptera), and caddisflies (order Trichoptera), collectively referred to as EPT, due to high egg mortality. Because EPT taxa cement eggs principally along the river edge habitats, eggs laid during stable low flows over the weekend would not be subjected to drying prior to hatching. Depending on the findings from the first test, this experiment may be conducted two to three times during the LTEMP period, but not during the first 2 years. In addition to potentially increasing EPT, macroinvertebrate production flows may benefit other aquatic food base organisms that have terrestrial adult life stages such as dragonflies and true flies (including midges and blackflies). Some loss of benthic production is possible in the shoreline areas that remain dewatered over the weekend. If this results in an unacceptable risk (e.g., decreased benthic production), the experiment would not be repeated. There is also the possibility that this experimental procedure may result in confounding interactions with trout management flow (TMF) experiments, which are also expected to be conducted during the LTEMP period.

Trout Management Flows. The 2003 Ecological Restoration Flows that began on January 1, 2003, consisted of daily fluctuations between 5,000 and 20,000 cfs in an attempt to disadvantage nonnative fish, particularly trout, during their winter spawning period. Overall, the 2003 Ecological Restoration Flows caused a drop in benthic biomass at cobble bars in Glen Canyon during the January to March flows followed by recovery through the summer (Shannon et al. 2003b). The flows did not have a long-term adverse impact on New Zealand mudsnail biomass and densities throughout the river. For example, at –3 Mile Bar, mudsnail biomass dropped by 70% between December and January collections; however, by June, New Zealand mudsnails had recovered to 90% of the December estimate (Shannon et al. 2003b).

TMFs conducted in spring and summer months (May–July), featured in all alternatives but Alternative A, would consist of several days at relatively high sustained flows (e.g., 20,000 cfs) followed by a rapid drop to low flows (e.g., 5,000 cfs), which would be held for a brief period (e.g., <24 hr). This pattern would be repeated for a number of cycles. Conditions for primary production should decrease slightly with increased turbidity during the higher discharge portion of TMFs (Reclamation et al. 2002). Although a temporary increase in total wetted area would occur under TMFs, areas would not be inundated for sufficient time to allow for benthic colonization (Benenati et al. 1998; Blinn et al. 1995). Thus, desiccation losses due to substrate exposure from dewatering of the varial zone would be minimal. Aquatic food base drift may increase during up-ramping to 20,000 cfs/hr associated with TMFs. Drift biomass has been observed to increase during the rising limb of the hydrograph (Miller and Judson 2014).

F.2.2.2 Temperature Effects on the Aquatic Food Base

One of the primary effects of dams, particularly those with hypolimnetic releases, is a change in water temperature, which is primarily responsible for the decline in invertebrate biodiversity. Warmer winter water temperatures can impact invertebrates in a number of ways including loss of physiological signals; disruption of normal growth, fecundity, and emergence; lack of winter chill to break insect egg or larval diapause; and early emergence. Cooler summer water temperatures can also impact invertebrates. For example, water temperatures high enough to complete development may not occur. Other impacts may include decreased fecundity, temporal separation of male and female emergence, delayed emergence, and prolonged emergence. The greater thermal constancy in annual and diel stream water temperatures downstream from dams also tends to decrease food base biodiversity (Vinson 2001).

Seasonal variation in water temperatures decreased gradually from Glen Canyon Dam closure in 1963 until about 1971, when water began to be drawn from the hypolimnion of Lake Powell. Main channel temperatures are now relatively isothermal at 7.2 to 10°C (45 to 50°F), but warm somewhat downstream in summer. There is an estimated maximum warming of the Colorado River mainstem of about 1°C (1.8°F) for every 35 mi, and water released at 10°C (50°F) from Glen Canyon Dam is expected to warm to about 17°C (62.6°F) near Diamond Creek (RM 225) in May or June (Benenati et al. 2002; Valdez 1994). Backwater habitats near the channel margins are one of the few aquatic habitats that warm above these levels. Backwater temperatures tend to warm with distance downstream from the dam (Valdez et al. 1998).

In winter, the mainstem water temperature near Diamond Creek is only about 1°C (1.8°F) higher than at Glen Canyon Dam (Cross et al. 2013). From 1988 to 2005, the average temperature of water released from Glen Canyon Dam was 9°C (48.2°F), and annual high temperatures at Diamond Creek between 1990 and 2002 were about 18°C (64.4°F). A drought that began in 2003 reduced water levels in Lake Powell and resulted in water temperatures that reached an annual high of 21°C (69.8°F) at Diamond Creek in 2005 (Hamill 2009).

In a two-week laboratory study, epiphytic diatom communities from the cold tailwaters of Glen Canyon Dam (12°C [54°F]) were incubated at 18 and 21°C. No change occurred in diatom composition between 18 and 21°C (64 and 70°F), but a significant change occurred between 12 and 18°C (54 and 64°F). At the higher water temperatures, smaller and closely adnate taxa became more important numerically than larger, upright diatoms (Blinn et al. 1989). This shift in diatom species composition may affect macroinvertebrates that feed on diatoms (Lechleitner 1992). Table F-4 provides the distribution, ecological importance, and favorable temperature range for select primary producers in the Colorado River, while Table F-5 provides temperature requirements for common zooplankton taxa.

TABLE F-5 Temperature Requirements for Common Zooplankton

	Temperature, °C (°F)		
Species	Minimum	Maximum	Optimum
Daphnia pulex (cladoceran)	10 (50)	28 (82)	20 (68)
Daphnia galeata (cladoceran)	10 (50)	25 (77)	20 (68)
Daphnia lumholtzi (cladoceran)	10 (50)	30 (86)	25 (77)
Leptodora sp. (cladoceran)	15 (59)	30 (86)	20 (68)
Bosmina sp. (cladoceran)	6 (43)	28 (82)	20 (68)
Diaphanosoma sp. (cladoceran)	10 (50)	30 (86)	25 (77)
Rotifers	15 (59)	30 (86)	25 (77)
Calanoid copepods	10 (50)	30 (86)	25 (77)
Cyclopoid copepods	10 (50)	30 (86)	25 (77)

Source: Valdez and Speas (2007).

If stream temperatures are raised by only a few degrees in winter, many aquatic insects that normally emerge in May or June may emerge in February or March and face death by freezing or will be prevented from mating because they are inactivated by low air temperatures. In addition, increases in stream temperatures may exaggerate the separation between the emergence of males and females (e.g., males may emerge and die before females emerge) (Nebeker 1971). Overall, temperatures above or below the optimum can lead to the production of small adults and lower fecundity (Vannote and Sweeney 1980). Slower warming of streams throughout the summer can reduce fecundity of emerging adults, exaggerate the separation of male and female emergence, prolong the emergence period of individual generations (which reduces the number of insects emerging at any given time, which may increase the individual risk of predation by trout or other fish), and reduce the growth rate such that emergence might occur later in the year when air temperatures are suboptimal for mating (Rader et al. 2008; Vinson 2001).

The lack of temperature variability in the Colorado River downstream of Glen Canyon Dam has selected for macroinvertebrates that do not require temperature cues to complete their development. This may at least partially account for the low levels of mayflies and caddisflies and the absence of stoneflies in the mainstem of the Colorado River (Oberlin et al. 1999). Fecundity of *Gammarus* in the tailwaters of Glen Canyon Dam is lower than that reported in other locations, probably due to water temperatures being well below the optimum of 18°C (64°F) for reproduction (Ayers et al. 1998). Table F-6 provides the distribution, importance to higher trophic levels, and temperature range for common benthic macroinvertebrates that occur in the Colorado River.

There is the possibility of an increase in the distribution and prevalence of fish diseases and parasites from river warming (Hoffnagle 2001; Valdez et al. 2000). Warmer, more stable backwaters could provide additional habitats for the Asian tapeworm (*Bothriocephalus acheilognathi)* and anchor worm (*Lernaea cyprinacea*) to substantially increase in abundance,

TABLE F-6 Distribution, Importance to Higher Trophic Levels, and Temperature Range for Common Benthic Macroinvertebrates Downstream of Glen Canyon Dam

Taxa	Distribution in Project Area	Importance to Higher Trophic Levels	Temperature Range in Project Area[a]	Favorable Temperature Range[a]
Gammarus lacustris (amphipod)	Glen Canyon Dam–Paria River (also important component of drift below this reach)	High	7–10 (45–50)	7–29 (45–84)
Simulium (blackfly)	RM 1.0 to Lake Mead and various tributaries	Medium-high	5–31 (41–88)	10–26 (50–79)
Cricotopus (midge)	Glen Canyon Dam–Paria River (also important component of drift below this reach)	Medium	7–10 (45–50)	15–21 (59–70)
Eukiefferiella (midge)	Glen Canyon Dam–Paria River (also important component of drift below this reach)	Medium	7–10 (45–50)	12–18 (54–64)
Orthocladius (midge)	Glen Canyon Dam–Paria River (also important component of drift below this reach)	Medium	7–10 (45–50)	8–18 (46–64)
Chironomus (midge)	RM 1.0 to Lake Mead	Medium	4–23 (39–73)	9–25 (48–77)
Aquatic worms	Glen Canyon Dam–Paria River (also important component of drift below this reach)	Low	4–23 (39–73)	8–25 (46–77)
Aquatic snails	Glen Canyon Dam–Paria River	Low	7–10 (45–50)	7–39 (45–102)
Potamopyrgus antipodarum (New Zealand mudsnail)	Glen Canyon Dam–Paria River	Low	4–23 (39–73)	7–28 (45–82)
Pisidium (pill clam)	Glen Canyon Dam–Paria River	Low	7–10 (45–50)	2–20 (36–68)
Dugesia (planarian)	Glen Canyon Dam–Paria River	Low	7–10 (45–50)	5–16 (41–61)
Baetis (mayfly)	Various tributaries	High	3–31 (37–88)	4–18 (39–86)
Hydropsyche (caddisfly)	Various tributaries	High	3–3 (37–88)	7–30 (45–86)
Megaloptera: Corydalidae (dobsonflies)	Various tributaries	High	5–28 (41–82)	5–30 (41–86)

[a] Temperature in °C (°F).

Source: Valdez and Speas (2007) and references cited therein.

resulting in their spread along the mainstem and into additional warmwater tributaries. Reported maximum temperature warming above those in the main channel for nearshore habitats ranges from 2.2°C (4.0°F) in eddies to 13°C (23.4°F) in backwaters (Vernieu and Anderson 2013). Warming of nearshore areas is somewhat ephemeral (e.g., decreases at night and during day under windy or cloudy conditions) (Vernieu and Anderson 2013). Temperatures greater than 20°C (68°F) would allow maturation of the Asian tapeworm, while temperatures greater than 15°C (59°F) would allow maturation of the anchor worm (Valdez et al. 1998). Section 4.5.2.4 of the EIS discusses fish parasite and disease incidence for mainstem locations. Table F-7 presents temperature requirements of the Asian tapeworm, anchor worm, and trout nematode. Whirling disease infection prevalence and disease severity reaches its highest levels at 10–15°C (Steinbach Elwell et al. 2009).

As analyzed in Section 4.5.3 of this EIS, temperature differences among alternatives would be minimal. Therefore, no significant changes in the aquatic food base due to elements of the base or condition-dependent operations are expected.

F.2.3 Conclusion

Table 4.5-1 of this EIS summarizes the impacts from the alternatives on the aquatic food base, while Section 4.5.3 presents the impacts of each alternative in more detail. Under Alternative A, existing conditions and trends in the composition, abundance, and distribution of the aquatic food base are expected to persist over the LTEMP period. The cessation of HFEs after 2020 may result in a shift to a food base community not dominated by midges and blackflies (important contributors to the diet of trout). Water temperatures, and their resultant influences on species composition, diversity, and production of the aquatic food base, would be similar to current temperatures in the Colorado River downstream of Glen Canyon Dam.

TABLE F-7 Temperature Requirements for the Asian Tapeworm, Anchor Worm, and Trout Nematode

	Host Activity Temperature Requirements[a]			Infestation Temperature Requirements[a]		
Species	Minimum	Maximum	Optimum	Minimum	Maximum	Optimum
Asian tapeworm	18 (64)	20 (68)	19 (66)	20 (68)	30 (86)	25 (77)
Anchor worm	20 (68)	30 (86)	25 (77)	18 (64)	30 (86)	25 (77)
Trout nematode	16 (61)	20 (68)	18 (64)	16 (61)	20 (68)	18 (64)

[a] Temperature in °C (°F).

Source: Valdez and Speas (2007).

Under Alternative B, benthic food base production would be similar to that under Alternative A. HFEs conducted less often than annually may lower the potential to establish a food base adaptable to flood conditions (i.e., one dominated by midges and blackflies). Hydropower improvement flows could decrease benthic food base production, which over the long term may also decrease drift (Kennedy et al. 2014). Over the short term, TMFs could also cause short-term increases in drift rates and slightly decreased primary production compared to Alternative A. Temperature impacts on the aquatic food base under Alternative B would be similar to those under Alternative A.

Under Alternative C, benthic food base productivity may be higher in December through June compared to Alternative A due to higher volumes and larger wetted area, but lower from August through November compared to Alternative A due to lower volumes and smaller wetted area. The more frequent HFEs compared to Alternative A would favor midge and blackfly production. Low summer flows are expected to lower food base production compared to higher flow conditions. Over the short term, TMFs could increase drift rates and slightly decrease primary production compared to Alternative A. Slightly warmer water temperatures for August and September at RM 225 under Alternative C may slightly increase food base production compared to Alternative A, although this could be offset by changes in diatoms from stalked to adnate forms and favoring *Oscillatoria* over *Cladophora*.

The relatively consistent monthly release volumes under Alternative D compared to Alternative A would produce a more consistent and stable aquatic food base. The more frequent HFEs under Alternative D are expected to favor midge and blackfly production compared to Alternative A. Over the short term, TMFs could increase drift rates and slightly decrease primary production compared to Alternative A. Macroinvertebrate production flows in May through August under Alternative D would be tested to determine whether they increase benthic food base production and diversity including the recruitment of mayflies, stoneflies, and caddisflies (important food base organisms currently rare to absent throughout much of the mainstem below Glen Canyon Dam). Low summer flows would provide ideal egg laying conditions for aquatic insects throughout the summer growing season and would therefore be expected to increase food base production; however, the lower magnitude of discharge might lead to lower drift concentrations of invertebrates. Temperature impacts on the aquatic food base under Alternative D would be similar to those under Alternative C.

Under Alternative E, relatively consistent monthly release volumes would favor aquatic food base productivity, but this effect would be offset by larger daily fluctuations. The frequent HFEs under Alternative E will favor midge and blackfly production, although the number of HFEs would be less than under Alternatives C, D, F, and G. Temperature impacts on the aquatic food base for Alternative E would be similar to those under Alternatives C and D.

Because of the comparatively lower flow volumes under Alternative F, food base biomass from July through the following March would be low compared to all other alternatives. Flow stabilization may allow for high benthic densities of New Zealand mudsnails. Over the long term, increased benthic production from flow stabilization may increase drift rates of food base organisms (Kennedy et al. 2014). Higher flow volumes in April through June may increase benthic food base biomass compared to Alternative A. The frequent HFEs will favor blackfly and midge production. The warmer water temperatures for August and September at RM 225 under Alternative F may slightly increase food base production even more than Alternative D, although this could similarly be offset by changes in diatoms from stalked to adnate forms and favoring *Oscillatoria* over *Cladophora*.

Under Alternative G, consistent and stable aquatic food base conditions would persist throughout the year. Benthic food base biomass would probably be greater under Alternative G, compared to Alternative F, because flows from July through the following February would be higher. However, stable flows may favor dominance by the New Zealand mudsnail. Potentially higher drift rates from spring flows under Alternative F would not occur under Alternative G. However, increased benthic production may increase drift rates over the long term (Kennedy et al. 2014). The frequent HFEs are expected to favor blackfly and midge production. Temperature impacts on the aquatic food base for Alternative G would be similar to those under Alternatives C, D, and E.

F.3 MODELING EFFECTS OF LTEMP ALTERNATIVES ON RAINBOW TROUT AND HUMPBACK CHUB

This section describes the methodology, results, and conclusions from a model developed in support of the LTEMP EIS by Yackulic (Grand Canyon Monitoring and Research Center [GCMRC]), Coggins (U.S. Fish and Wildlife Service [FWS]), and Korman (EcoMetrics) to evaluate effects of alternatives on rainbow trout and humpback chub populations. Although other models were considered for use, the combined rainbow trout-humpback chub model described here was developed to incorporate recent information from the Natal Origins, Juvenile Humpback Chub Monitoring, and Near-Shore Ecology projects being conducted by GCMRC and to utilize newer approaches to modeling humpback chub population demographics. In addition, there was a need to develop the model in a software environment in which batch processing of model runs for multiple hydrologic and sediment input scenarios for each alternative would be feasible and computationally efficient. The model used existing data to inform parameter estimates whenever possible.

F.3.1 Model Overview

The trout-chub model consists of three combined submodels: (1) a model of rainbow trout population dynamics in the Lees Ferry reach, (2) a model of rainbow trout movement and survival downriver from Lees Ferry (trout routing model), and (3) a model of the response of humpback chub population dynamics in the Little Colorado River and Colorado River to monthly mainstem temperatures and monthly trout abundances. The model of the Lees Ferry rainbow trout population dynamics is similar to previous models used for the Glen Canyon reach. The trout movement and humpback chub models, on the other hand, were developed for this application to reflect recent advances, with an emphasis on deriving parameter values from data. The following paragraphs give a brief overview of the three submodels, with more detailed descriptions of each submodel available in subsequent sections (Sections F.3.1.1, F.3.1.2, and F.3.1.3).

Output from a flow model drives the Lees Ferry rainbow trout submodel. This model can be run independently from the trout routing and humpback chub submodels, as it does not include any feedbacks. Simulations include interannual variability in recruitment, outmigration, and growth based both on regression-derived predictions and variation around these predictions. Simulations also include parameter uncertainty, which includes a critical uncertainty related to the effectiveness of TMFs. The parameter associated with the critical uncertainty was fixed at two values (0.10 and 0.50) encompassing a hypothesized range of effectiveness, while all other parameters were drawn from the multivariate distribution estimated from data. Outputs from this submodel include four performance metrics, including simulated outmigration, which was used as input to the trout routing submodel.

The trout routing submodel includes a single biological parameter describing movement, as well as multiple inputs related to implementation of mechanical removal, all of which are fixed. The trout routing submodel model is run a year at a time, after which it passes monthly rainbow trout abundances to the humpback chub submodel.

The humpback chub submodel simulates the impacts of rainbow trout and temperature (forecasted by a temperature model) on humpback chub population dynamics at a monthly time step. It returns the adult population abundance at the end of the year, which is used as one of the performance metrics and is also used by the trout routing model to determine whether mechanical removal would be triggered in the next year. The humpback chub submodel includes parametric uncertainty including levels for a critical uncertainty related to the effect of rainbow trout on humpback chub survival and growth, as well as variation in other parameters. The humpback chub submodel also includes interannual variability in recruitment and outmigration.

More detailed information about each of the submodels is provided in the following sections.

F.3.1.1 Glen Canyon Trout Submodel

An age-structured population dynamics model was used to predict the abundance and growth of rainbow trout in Glen Canyon and the number of those fish that migrate into Marble Canyon. The model makes predictions on an annual time step for ages 1–6 yr. Annual recruitment, which was defined as the number of age-0 fish (i.e., fish hatched in the current year) that enter the population in a given year, is predicted based on flow statistics, and growth is predicted as a decreasing function of overall rainbow trout abundance. Abundance, in combination with age-specific angling vulnerabilities, is used to make predictions of angler catch per hour of effort. Predicted abundance and size distributions are used to compute the number of high-quality fish (trout ≥16 in. total length) potentially available for capture. The number of fish migrating from the Glen Canyon reach into Marble Canyon each year (out-migrants) is predicted as a proportion of the previous year's recruitment, and is used to determine the potential number of fish that eventually migrate down to the confluence of the Little Colorado River (RM 61), where their effects on humpback chub are simulated. Basic simulation parameters and those for key functional relationships were derived or fitted to values from the Korman et al. (2012) stock synthesis model. This model used 21 years of electrofishing-based catch-per-effort data for Glen and Marble Canyons, in conjunction with length frequencies and other information, to estimate annual recruitment, survival rate, von Bertalanffy growth parameters, and outmigration patterns (numbers, size, and timing). Specifics of the Glen Canyon trout simulation model are provided below.

Recruitment

The annual recruitment of age-0 trout in Glen Canyon was predicted based on a multiple linear regression driven by flow-derived independent variables. The model predicted log annual recruitment as a function of annual Glen Canyon Dam release volume, the range in mean daily flows during the critical early life history rearing period (May–August), and the presence of a spring HFE in each year or in the previous year (Korman et al. 2011c; Avery et al. 2015). The model explained 55% of the annual variation in the recruitment estimates from the Korman et al. (2012) stock synthesis model between 1990 and 2010 (Figure F-1). The flow-dependent regression model predicted that recruitment would be higher in years with greater annual volumes, reduced daily variation in flow between May and August, and when spring HFEs occur. In the simulation model, log recruitment each year is predicted from a random normal distribution, with the mean determined by linear regression parameters and hydrologic statistics, and the extent of error determined by the residual error in the regression model.

Recruitment for a given year was predicted to be higher if a spring HFE occurred in that year or in the previous year, based upon empirical relationships reported by Korman et al. (2011c). However, there is insufficient information to draw a conclusion about whether HFEs that occur in the fall would have a similar effect on recruitment of trout. The model considered this uncertainty about the effect of fall HFEs on recruitment of rainbow trout in the Glen Canyon reach by examining two hypotheses: (1) fall HFEs would have no effect on recruitment and (2) recruitment would increase at the same rate as seen with spring HFEs, but for only one year instead of two.

As described in Section 2.3.3.2 of this EIS, TMFs are a special type of fluctuating flow designed to reduce the recruitment of trout by disadvantaging young-of-the-year (YOY) trout. TMFs have been proposed and developed on the basis of research described in Korman et al. (2005). TMFs are included as elements of some alternatives evaluated in the LTEMP EIS, and the Glen Canyon trout submodel incorporated the ability to consider the effects that occurrence of TMFs could have on trout resources. For alternatives and associated long-term strategies that included TMFs, these flows were triggered in the model during years in which the initial production of YOY rainbow trout (based on hydrologic characteristics) in the Glen Canyon reach was anticipated to be greater than 200,000 individuals. Because there is uncertainty regarding how effective TMFs would be at disadvantaging YOY trout, the model was used to evaluate two different levels of effectiveness by reducing the number of YOY trout surviving to age-1 by either 10% or 50% for each 20-year simulation period.

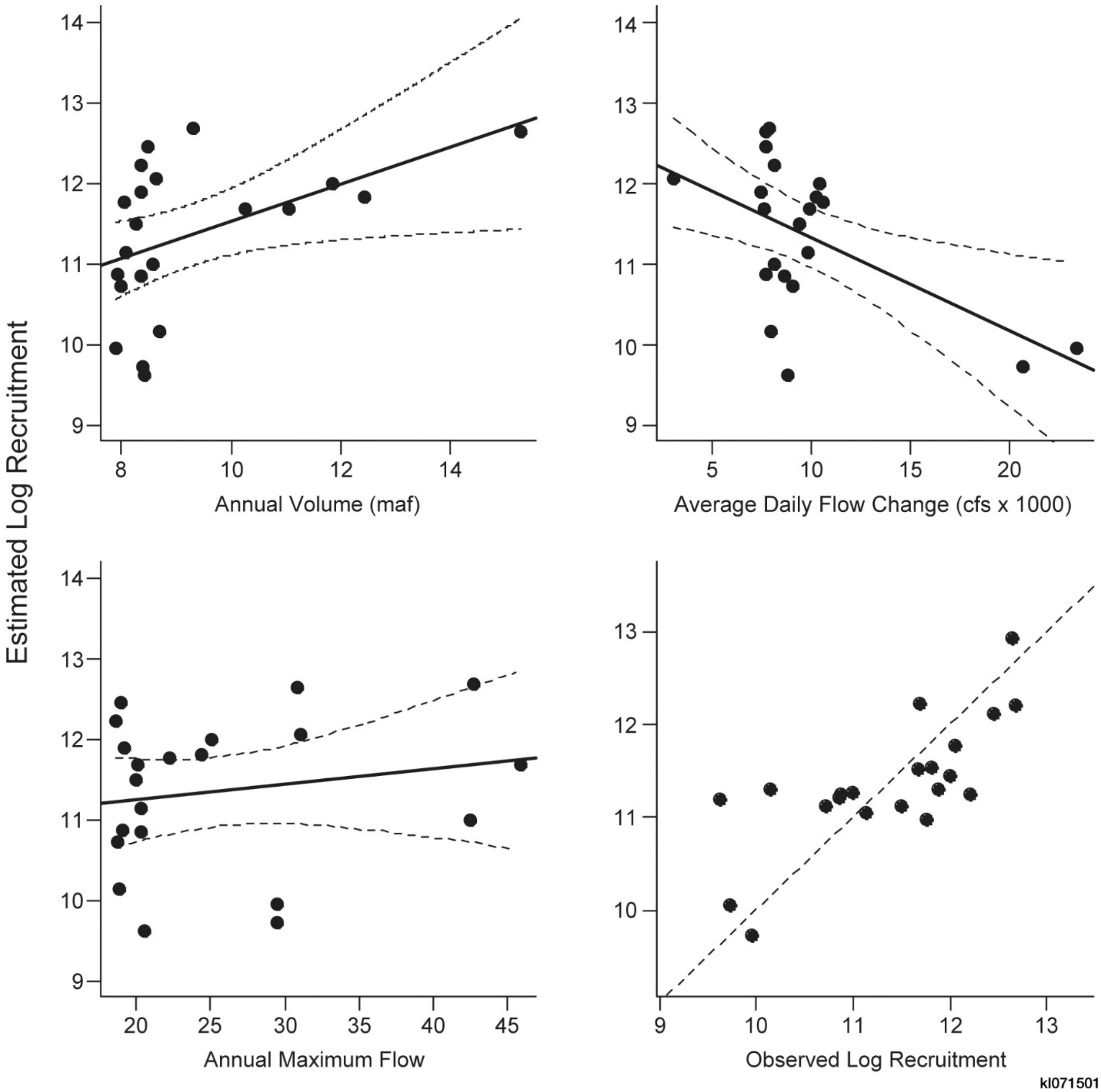

FIGURE F-1 Fit of Regressions Predicting the Log of Recruitment of Rainbow Trout in the Glen Canyon Reach Estimated by the Korman et al. (2012) Stock Synthesis Model as a Function of the Annual Release Volume from Glen Canyon Dam (million acre-feet), the Range of Mean Daily Flows during May–August (thousand cfs), and the Maximum Flow (cfs) Each Year (The bottom-right plot compares the overall fit of a multiple regression model with annual volume and range of mean daily flows during May–August as independent variables, and with the maximum annual flow independent variable replaced with a dummy variable with values of 1 for years prior to or with spring HFEs. The dashed line in the bottom-right graph indicates the 1:1 relationship and the 95% confidence interval in other graphs. The multiple regression model explained 55% of the variation in log recruitment and was statistically significant [p = 0.002].)

Growth

Length-at-age was calculated assuming a von Bertalanffy relationship that depends on the Brody growth coefficient (*vbK* = 0.55), the asymptotic length (L_{inf}, size at the terminal age), the coefficient of variation in length-at-age (*cvLen* = 0.1), and the mean size at age 1 (L_0 = 130 mm). Parameter estimates were derived from the stock synthesis model, which was fit to length-frequency and supplemental growth data (Korman et al. 2012, 2011a,b). Annual variation in asymptotic length was predicted as a linear function of the abundance of trout >150 mm. This model predicts only 18% of the annual variation in the annual asymptotic length estimates from the stock synthesis model (Figure F-2). To simulate interannual variation in L_{inf} in the model, annual deviates of L_{inf} in log space were added to a base value (5.89). In the simulation, predicted deviates from the $L_{inf} - N > 150$ regression model were added to the base value, and these formed the mean of a random normal distribution with a standard deviation equal to the residual error of the regression model.

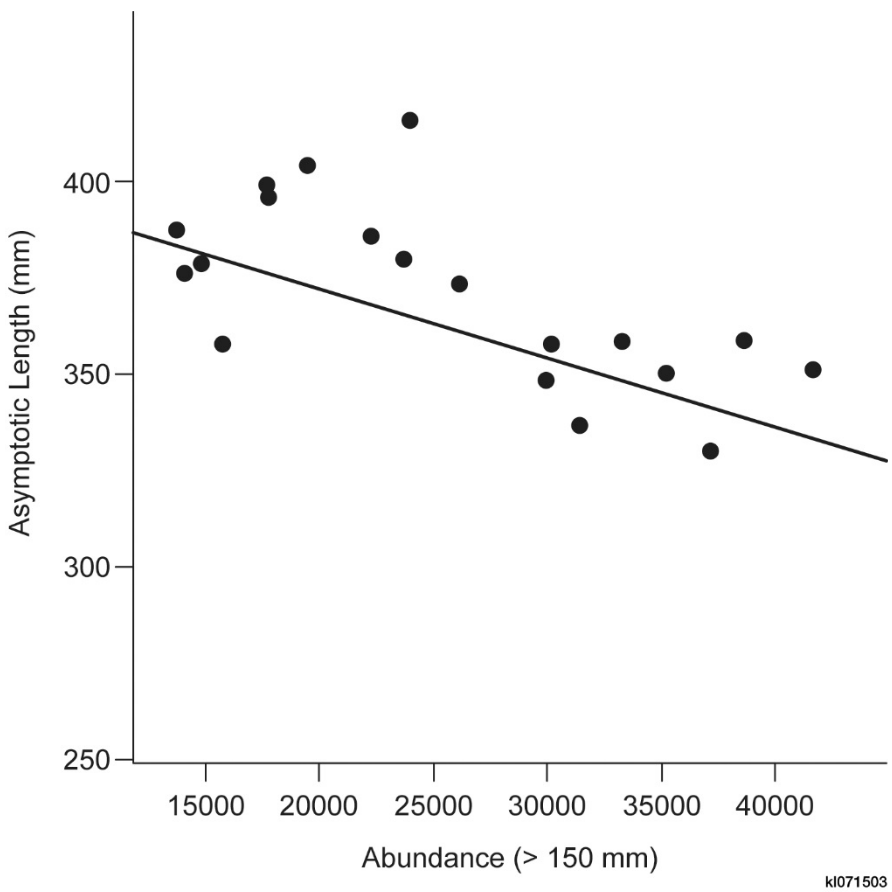

FIGURE F-2 Relationship between Annual Estimates of the Asymptotic Length of Rainbow Trout in Lees Ferry Predicted by the Stock Synthesis Model as a Function of the Estimated Abundance for Fish >150 mm (approximately age-1+) Each Year (The solid line is the best-fit relationship. This relationship explained 18% of the annual variation in asymptotic length and was not statistically significant [p = 0.056].)

Performance Metrics from Glen Canyon Trout Submodel

Four performance metrics were derived from the trout submodel in order to evaluate the relative degree to which the various alternatives would achieve a healthy high-quality recreational trout fishery in Glen Canyon National Recreation Area and reduce or eliminate downstream trout migration consistent with National Park Service fish management and Endangered Species Act (ESA) compliance needs. These four trout performance measures were:

1. Glen Canyon trout abundance index (for age-1+ fish)

2. Catch rate index (number/hr) for age-2+ fish

3. Number of trout >16 in. total length

4. Trout emigration estimate (number of age-0 trout moving into Marble Canyon from Glen Canyon)

The Glen Canyon trout abundance index was calculated as the average of modeled annual abundance of trout that were 1 year of age or older during each 20-year simulation period. The model used an age-structured population dynamics model to calculate the annual abundance for age classes 1 through 6 based upon annual recruitment rates and density-dependent survival rates.

The catch rate index was calculated as the average annual angling catch per unit of effort (number of fish per hour) during the 20-year simulation period. Only fish 2 years of age or older were considered vulnerable to angling. The annual angling catch per effort in the fishery (CPE_{yr}) was predicted as the sum of products of an overall catchability coefficient ($q = 4.25e^{-05}$), age-specific vulnerabilities ($V_1 = 0$, $V_2 = 0.5$, and V_3 *to* $V_6 = 1$), and the predicted age-specific abundance for the year ($N_{yr,a}$):

$$CPE_{yr} = \sum_{a=0}^{6} \left(q \times V_a \times N_{yr,a} \right)$$

To estimate q, the simulation model was run using the recruitment estimates from the stock synthesis model to predict age-specific abundance between 1990 and 2010. The value for q was then calculated from the back-transformed average of the log of the ratio of the observed CPEs to the estimates of the vulnerable population each year. Thus, q represents the average scalar required to convert predicted vulnerable abundance to the observed CPE.

In order to evaluate the potential for large trout to be present in the population under a given alternative, a performance metric was calculated as the average of the annual modeled number of fish equal to or greater than 16 in. that would be present in the Glen Canyon reach. The number of trout in the population with total lengths equal to or greater than 16 in. during a given year was predicted as the sum of the products of the abundance-at-age and the proportion of the age with lengths greater than or equal to 16 in. That proportion meeting the length criterion is predicted based on a normal distribution (*pnorm*) with a mean predicted by expected length-at-age ($Len_{yr,a}$) determined using the von Bertalanffy relationship and a standard deviation determined by the coefficient of variation in length-at-age (*cvLen*):

$$N_{qual,yr} = \sum_{a=0}^{6} \left[N_{yr,a} \times pnorm\left(16, Len_{yr,a}, cvLen \times Len_{yr,a}\right)\right]$$

The trout emigration performance metric was calculated as the average of the annual modeled number of trout migrating from Glen to Marble Canyon during a 20-year simulation period. The Glen Canyon Trout submodel computes the number of trout migrating to Marble Canyon as a fraction of the recruitment estimate from the previous year (Figure F-3). A linear model with a zero intercept explained about 70% of the estimated outmigration from Korman et al. (2012). The model predicts that on average, the number of out-migrants is 42% of the recruitment value from the previous year; however, there is considerable interannual variation in this percentage (95% of values are between 0 and 91%). A normal distribution (*rnorm*) with a mean equal to the mean of the logit-transformed annual proportions and a standard deviation equal to the standard deviation of the transformed proportions was used to simulate the proportion of fish out-migrating in each year of the simulation. The back-transformed proportions were then multiplied by the previous year's recruitment ($Rec_{\text{yr-1}}$) to calculate the out-migration each year:

$$Out_{yr} = Rec_{yr\text{-}1} \times \text{logit}[rnorm(\text{mean} = –0.35, \text{sd} = 1.65)]$$

Parameter estimates for the key linear models (recruitment-flow, out-migration-recruitment, asymptotic length-abundance) were estimated by linear regression. The variance-covariance matrices for these models, which represent the extent of uncertainty in parameter estimates and their covariation, were used to generate 1,000 different parameter values for each relationship. The simulation model integrated over these values to incorporate uncertainty in key functional relationships when making predictions for any long-term strategy and hydrologic trace.

In addition to the performance metrics that were used to evaluate the potential effects of alternatives and long-term strategies on the trout fishery and downstream migration of trout, the trout submodel also kept track of the number of TMFs expected to be triggered during each 20-year simulation period. The number of TMFs during a 20-year period was used as one indicator of how American Indian Tribes—some of which consider lethal actions to fish an adverse effect if there is no beneficial use—could be affected by alternatives and long-term strategies (see Appendix I), rather than as a measure of effects on the trout fishery itself.

Evaluation of Trout Submodel

Annual flow statistics for Glen Canyon Dam were computed from the historical record between 1990 and 2010 and used as input to the Glen Canyon trout model to compare predictions with observations and best estimates of key state variables such as recruitment,

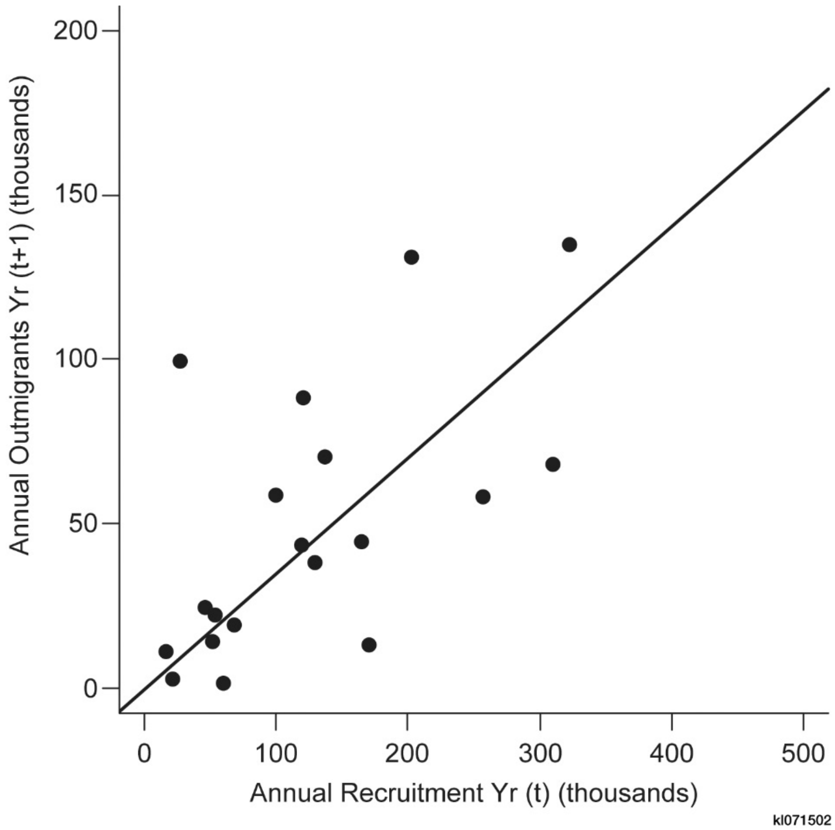

FIGURE F-3 The Relationship between Annual Recruitment of Rainbow Trout in Lees Ferry Estimated by the Korman et al. (2012) Stock Synthesis Model and the Number of Trout That Emigrate from Lees Ferry into Marble Canyon the Following Year (The solid line represents the best-fit relationship, which assumes no out-migration when there is no recruitment in the previous year [i.e., the line is forced through the origin]. This relationship predicted 72% of the annual variation in estimated out-migration and was statistically significant [$p < 0.001$].)

out-migration, and size at the terminal age. Predictions of angler CPE were compared to estimates of annual CPE from the Arizona Game and Fish Department (AZGFD) creel survey (Makinster et al. 2011). Simulations were based on most likely parameter estimates from the key regression models (recruitment-flow, outmigration-recruitment, asymptotic length-abundance) and did not include interannual variation in predictions to facilitate comparisons of predictions and data. Predictions of abundance were compared to the interannual trend in AZGFD electrofishing surveys. Other predictions (recruitment, asymptotic length, and out-migration) were compared to best-fit estimates from the Korman et al. (2012) stock synthesis model.

The historical flow-driven predictions of recruitment made by the simulation model produced an interannual trend quite similar to the estimates produced by Korman et al. (2012; Figure F-4, top-left panel). However, the model substantially over-predicted recruitment in 1996 and under-predicted recruitment in 2007–2009. The effects of high annual volumes and spring floods on recruitment may be confounded with other variables in the multiple regression model due to the low frequency of these events in the period of record.

The trend in predicted abundance from the model generally matched the trend in electrofishing-based CPE (Figure F-4, middle-left panel). The model over-predicted abundance in 2005–2007, perhaps because it did not account for a number of unusual events in earlier years that likely affected recruitment and adult mortality (e.g., a sudden change in minimum flow due to an emergency shutdown of Glen Canyon Dam generators, very few spawners in 2006, mortality of adults due warm water and low dissolved oxygen in releases during the fall of 2004). The trend in asymptotic length predicted by the model did not provide a good fit to the trend from the Korman et al. (2012) stock synthesis model. This is not surprising, as trout abundance was a relatively poor predictor of asymptotic length (Figure F-3), especially in years when other factors (e.g., low food availability, high mud snail abundance) appeared to have strong effects. Factors such as food availability and quality and long-term trends in reservoir productivity are likely more important drivers of growth than abundance.

The model was only partially able to reproduce the observed trend in angling CPE (Figure F-4, top right). It correctly predicted an increase with abundance between 1992 and 1997. However, observed CPE for the following 3 years was relatively stable, while model predictions indicated that CPE increased by about threefold. As the model provided a relatively good fit to the observed trend in electrofishing CPE over the majority of the historical period, this likely indicates that catchability (*q*) declined beginning in 1999. Possible mechanisms include a reduction in *q* at higher trout densities, as a greater fraction of fish use less vulnerable habitats, or reduced *q* at lower flows (which began in 1999). The predicted number of quality-sized fish in the population (dashed line, top right) has been low over the entire historical period (<1000) and declined from maximum values at the start of the period due to increasing abundance (top left), which reduced asymptotic length (bottom left).

The trend in simulated out-migration estimates was reasonably close to the historic trend estimated by the stock synthesis model (Figure F-4, bottom right). The proportion of recruitment that out-migrates each year is not constant (Figure F-2), and this simplification in the application of the simulation model to historical data leads to some of the error in out-migration estimates. Departures between the best recruitment estimates (Korman et al. 2012) and those derived from the flow regression (Figure F-4, top left) increases the extent of error in out-migration estimates.

F.3.1.2 Trout Movement Submodel

One component of the LTEMP trout/humpback chub simulation model is the movement of rainbow trout from Glen Canyon to near the confluence of the Little Colorado River. The trout movement model predicts the monthly abundance of trout within each mile segment of the Colorado River from RM 0 to RM 150 and reports monthly abundance over broader river reaches as required for the humpback chub population dynamics model. While the LTEMP rainbow trout-humpback chub model is not focused on locations below approximately RM 66, the trout movement model extends below this location to avoid problems with modeling boundary conditions near the Little Colorado River. Key inputs to the trout movement model

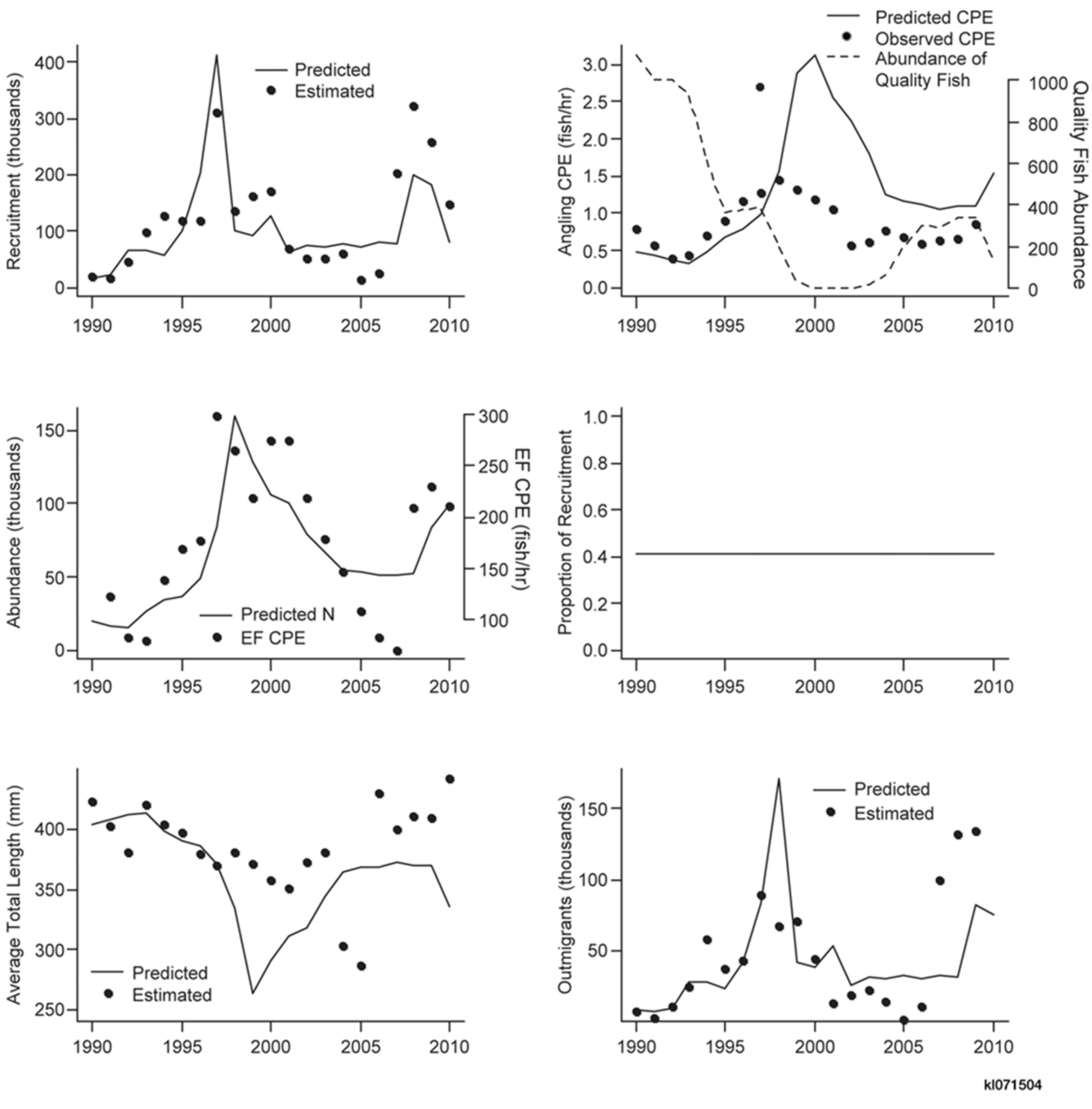

FIGURE F-4 Fit of the Glen Canyon Rainbow Trout Simulation Model to Predictions of Recruitment (top left), Asymptotic Length (bottom left), and the Number of Out-migrants (bottom right) Predicted by the Korman et al. (2012) Stock Synthesis Model (Also shown is the predicted abundance relative to catch per effort [CPE] from AZGFD electrofishing surveys [middle left], the predicted angling CPE compared to AZGFD creel survey estimates [top right], and the average proportion of recruitment that migrates from Glen to Marble Canyon each year [middle right].)

include the monthly number of age-0 trout out-migrating from the Glen Canyon reach, parameters that control the movement and dispersion of rainbow trout in Marble/Grand Canyons, the natural mortality rate of rainbow trout, and the number, intensity, and timing of nonnative mechanical removal trips conducted each year. The rules governing the implementation of mechanical removal were specified based on the Biological Opinion on nonnative fish control (Reclamation 2011b). Monthly movement and dispersion do not depend on trout density and are modeled as a random process following a Cauchy distribution of movement distances. To allow parameter estimation and to evaluate the ability of the model to reconstruct historic trout abundance patterns, model-predicted catch of rainbow trout from sampling efforts in 2000–2009 was compared to the observed catch.

The trout movement model accounts for the abundance of rainbow trout at a monthly time step and in 1-mi-long river segments (RM segments) from RM 0 to RM 150. The age and size structure of the population was not modeled, although all immigrants from Glen Canyon are assumed to be YOY. At the end of each month, trout within each RM segment are diminished by some survival rate and then distributed to other RM segments according to a RM segment-specific movement distribution. This calculation is accomplished via matrix operations as:

$$\mathrm{n}(t+1) = \mathrm{MSn}(t),$$

where *n* is a vector containing the abundance of rainbow trout within each RM segment, *M* is the movement matrix specifying how the abundance at a particular RM segment is distributed to other segments, *S* is the survival matrix where the diagonal contains the survival of fish within each RM segment and all other elements are zero, and t is the month of the year.

Number of Trout from the Glen Canyon Reach

The number of fish entering the upstream-most RM segment in the model (RM 1) each month equals the number of annual emigrants calculated by the Glen Canyon trout submodel. The monthly number of trout entering the reach was assumed to be 1/12 of the annual total emigrants, as migration timing was assumed to be uniform across months.

Survival

Instantaneous natural mortality rate (M = 0.49/year) was assumed to be temporally and spatially constant and corresponded to a monthly survival rate of 0.96 based on mark recapture-based methods from the Natal Origins project (Korman et al. 2015). In the RM segments RM 56 to RM 66, monthly survival is also potentially influenced by mechanical removal operations. Survival rate associated with mechanical removal was modeled as:

$$MR_{surv} = (1-p),$$

where MR_{surv} is survival from mechanical removal, p is the electrofishing capture probability, and D is the number of times fish are removed from each RM segment (number of passes). Thus monthly survival rate in RM segments where mechanical removal is not conducted was 0.96, and in RM segments where mechanical removal was conducted, it was 0.96 × MR_{surv}. The diagonal elements of the survival matrix *S* contained these RM segment survival rates and non-diagonal elements were zero. Capture probability (p) was assumed to be 0.10, based on recent work from the Natal Origins project (Korman et al. 2015).

Mechanical Removal

Mechanical removal in RM 56–66 was triggered in a particular year when three conditions were simultaneously met: (1) mechanical removal was authorized under the alternative being modeled, (2) the estimated abundance of rainbow trout in the trigger reach (RM 63–64.5) during September of the previous year was greater than 760 individuals, and (3) the estimated number of adult humpback chub (from humpback chub submodel, see Section F.3.1.3) was less than 7,000 individuals. When the triggering conditions were met, mechanical removal was implemented as six removal trips that occurred from February through July. Occurrence of removal trips reduced the number of trout in the vicinity of the Little Colorado River during the month, based upon the abundance of trout and electrofishing capture probability estimated from past removal efforts.

Trout Movement

The movement of fish between RM segments was assumed to be a diffusion process in which the probability of a fish moving from each RM segment to every other RM segment followed a truncated Cauchy distribution. The distribution is said to be truncated as movement upstream of the RM 0 segment or downstream of the RM 150 segment was disallowed. The probability distribution for each RM segment was assumed to represent the proportions of fish that would move to every other segment and formed a row vector in a movement matrix.

Performance Metrics from Trout Movement Submodel

The principal purpose of the trout movement submodel was to provide inputs to the humpback chub population submodel pertaining to monthly estimates of the number of rainbow trout that would be present in the vicinity of the Little Colorado River, and to calculate the number of trout that would be removed by mechanical removal efforts. Although no aquatic ecology performance metrics were generated, the trout movement submodel was used to calculate the numbers of years in which mechanical removal trips were triggered for each 20-year simulation, and that calculation was used as an indicator of how Tribal resources could be affected by alternatives and long-term strategies (see Appendix I).

Two factors must coincide to trigger mechanical removal trips in the submodel: (1) there must be more than 760 adult rainbow trout projected for the test reach in the vicinity of the Little Colorado River confluence (RM 63–RM 64.5) and (2) the projected adult humpback chub population must be less than 7,000 individuals. The number of adult humpback chub is calculated by the humpback chub population submodel and provided as input to the trout movement submodel. Once triggered, the model assumes that six mechanical trip passes would occur during the year.

Estimating Model Parameters and Evaluating Model Predictions

Rainbow trout electrofishing catch data from 2000 through 2009 and between RM 0 and RM 65.7 were used to estimate the Cauchy scale distribution parameter and the catchability coefficient (q). These data were composed of the annual electrofishing catch and effort by the 10-mi reaches between RM 0–50 and by the reaches RM 50–61.5 and RM 61.5–65.7 (Makinster et al. 2011). The annual predicted catches (C_i) for each reach (i) were computed as:

$$C_i = n_i \times E_i \times q,$$

where n_i is the model-predicted abundance of rainbow trout within reach i during the month of June and E_i is AZGFD electrofishing effort in June.

The observed catch was assumed to be distributed as a Poisson random variable with mean equal to the model-predicted catch. Estimation of the Cauchy scale ($\gamma = 3.38$) parameter and the catchability coefficient ($q = 3.4e^{-06}$) was accomplished via the method of maximum likelihood and the function "optim" within R (R Core Team 2013). These estimates provided a reasonably good fit to the data (Figure F-5), providing confidence that the simulation model would accurately portray movement dynamics of rainbow trout. A more complex parameterization of the Cauchy distribution was tested, where the location parameter (which specifies the most probable movement distance) was estimated as a free parameter. The maximum likelihood estimate of the location parameter was approximately $5.0\ e^{-03}$, confirming that most fish do not change location on a monthly basis. A normal distribution also was considered to describe movement distance, but there was a better fit to the data using the Cauchy distribution. In addition, the Cauchy distribution of movement implies a smaller probability of fish moving long distances within a month than the normal distribution (Figure F-6) and is more biologically reasonable, considering the observed movement of tagged trout.

F.3.1.3 Humpback Chub Population Submodel

A size- and location-structured population dynamics model was used to predict the size of the adult population of humpback chub over time. The model assumes five size classes of humpback chub (40–99 mm, 100–149 mm, 150–199 mm, 200–249 mm, and >250 mm, size classes 1–5, respectively) and two locations (Little Colorado River and Colorado River) for a total of 10 "states" (where a state is a unique combination of size and location; for example, a fish in the Little Colorado River that is 40–99 mm is in state 1; Figure F-7). The structure of this model is based on recent modeling work (Yackulic et al. 2014) as well as a new set of candidate models developed specifically to address the effects of temperature and rainbow trout on humpback chub survival (see "Model Selection and Development" below). The model uses a monthly time step and assumes constant survival for all states except for state 6, corresponding to juveniles in the Colorado River. Survival for this state depends on rainbow trout abundance. Growth of size class 1 (40–99 mm) humpback chub depends on both water temperature and rainbow trout abundance. Growth for all other size classes in the Colorado River is temperature-dependent, while Little Colorado River growth is assumed to be constant. Movement between

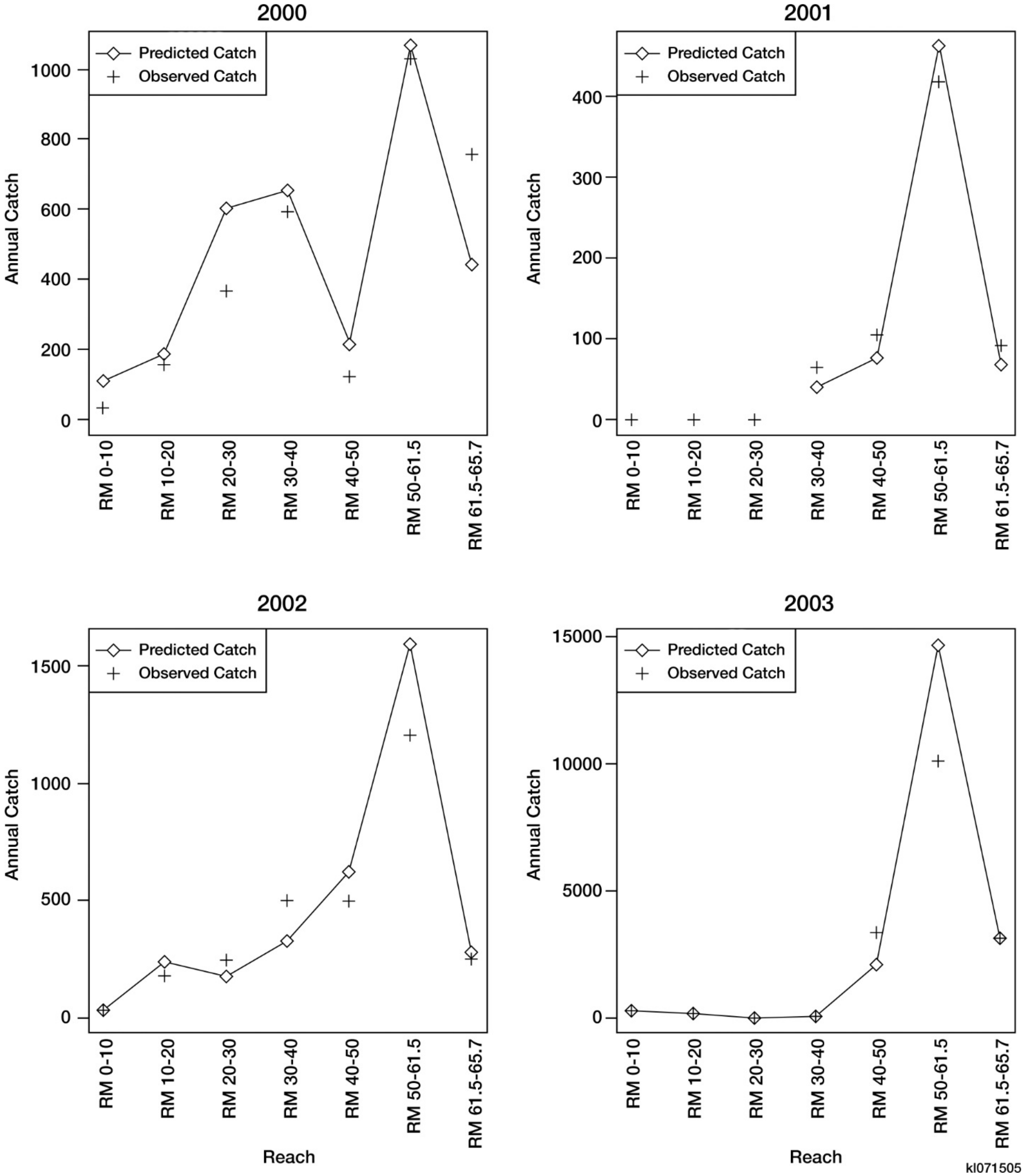

FIGURE F-5 Predicted and Observed Annual Catch of Rainbow Trout by Year and River Reach

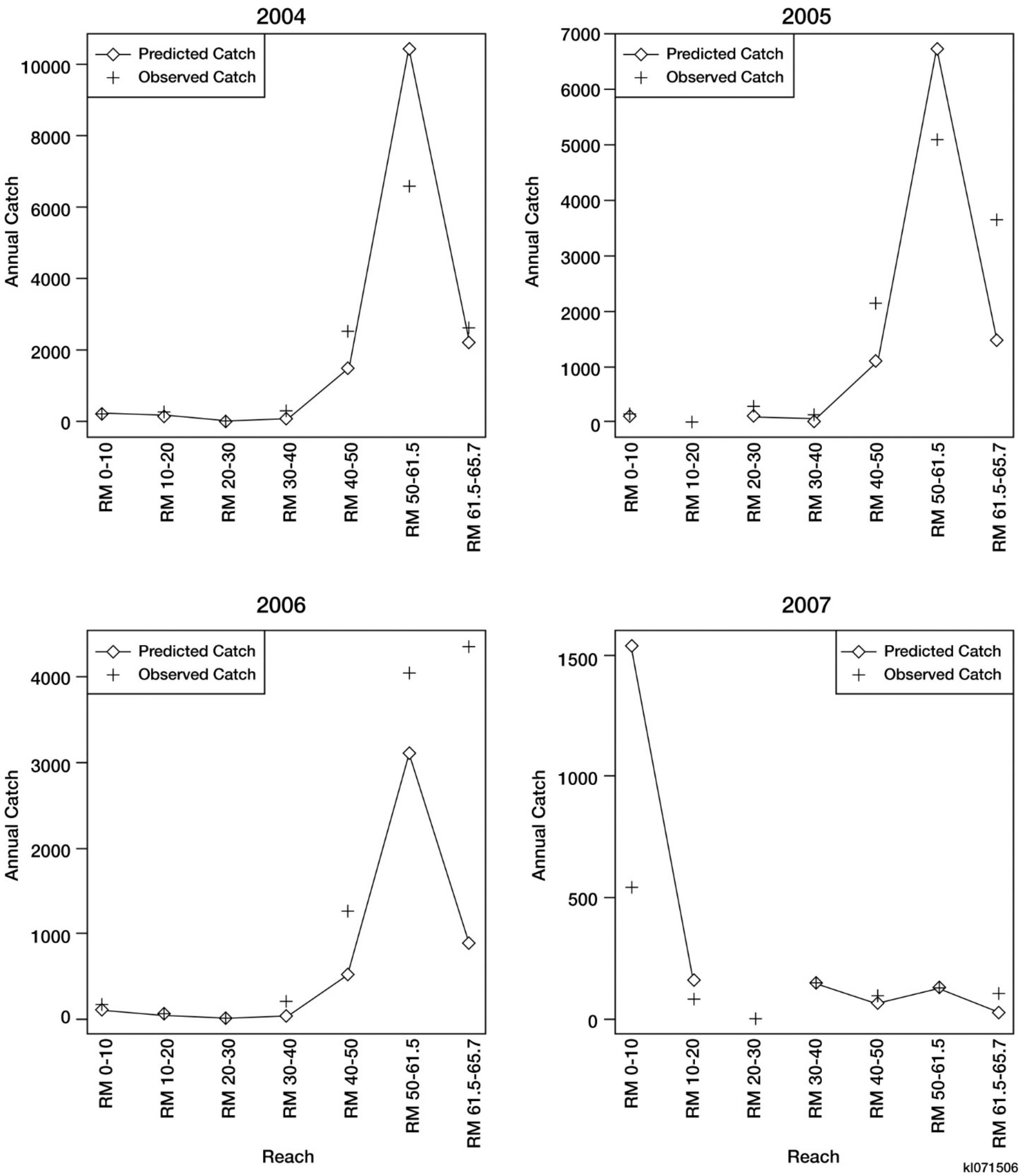

FIGURE F-5 (Cont.)

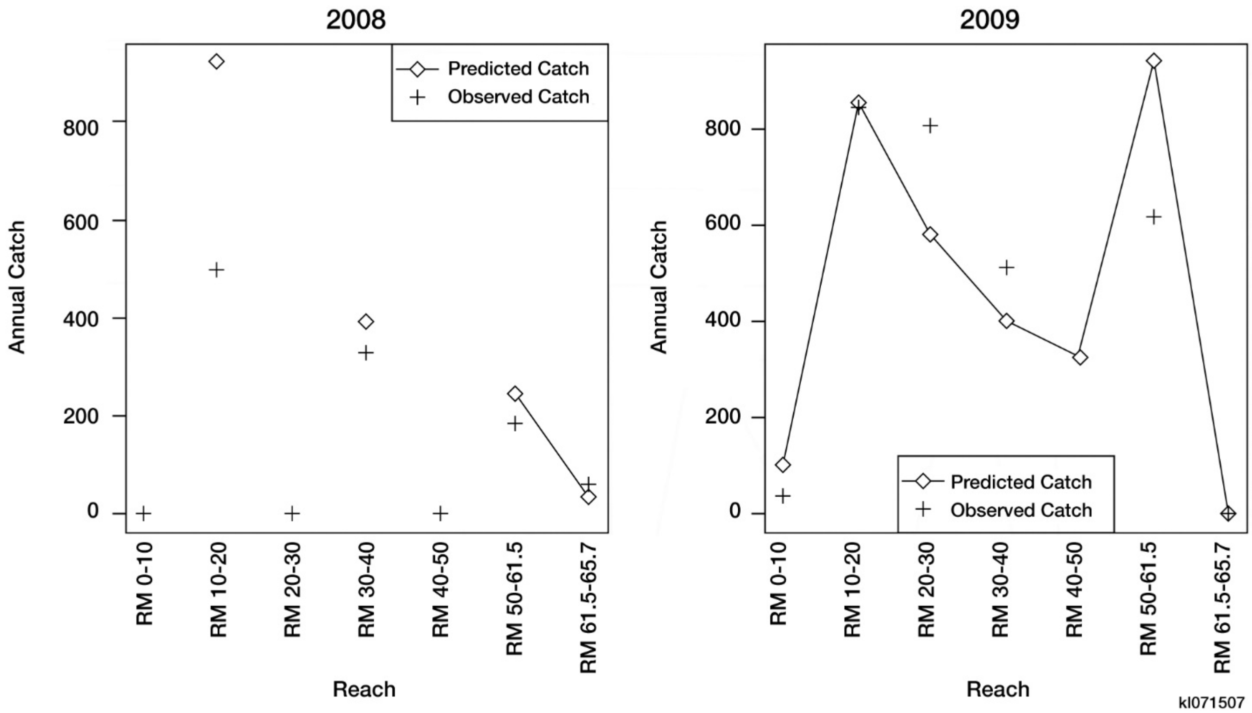

FIGURE F-5 (Cont.)

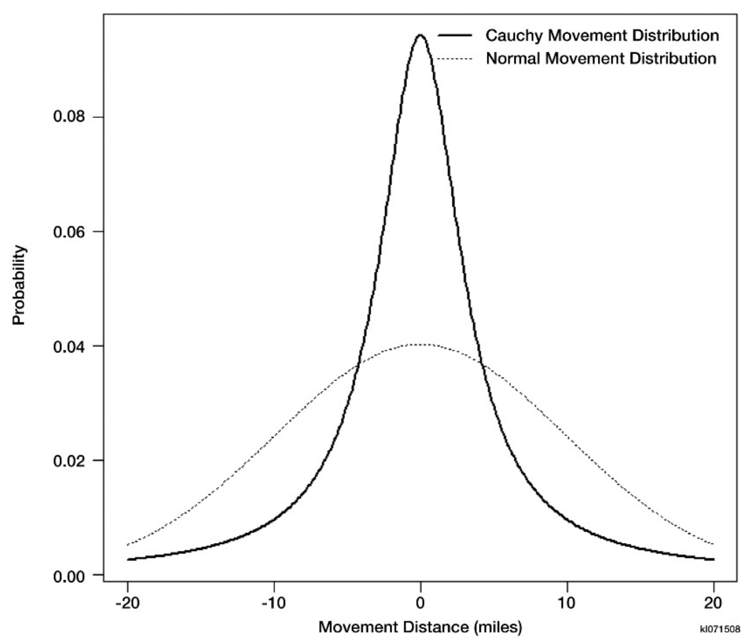

FIGURE F-6 Best-Fitting Distributions Describing Monthly Movement of Rainbow Trout in Marble Canyon Assuming Either a Normal or Cauchy Distribution (The Cauchy distribution implies a lower probability of large monthly movements and agrees better with movement observations from tagging data.)

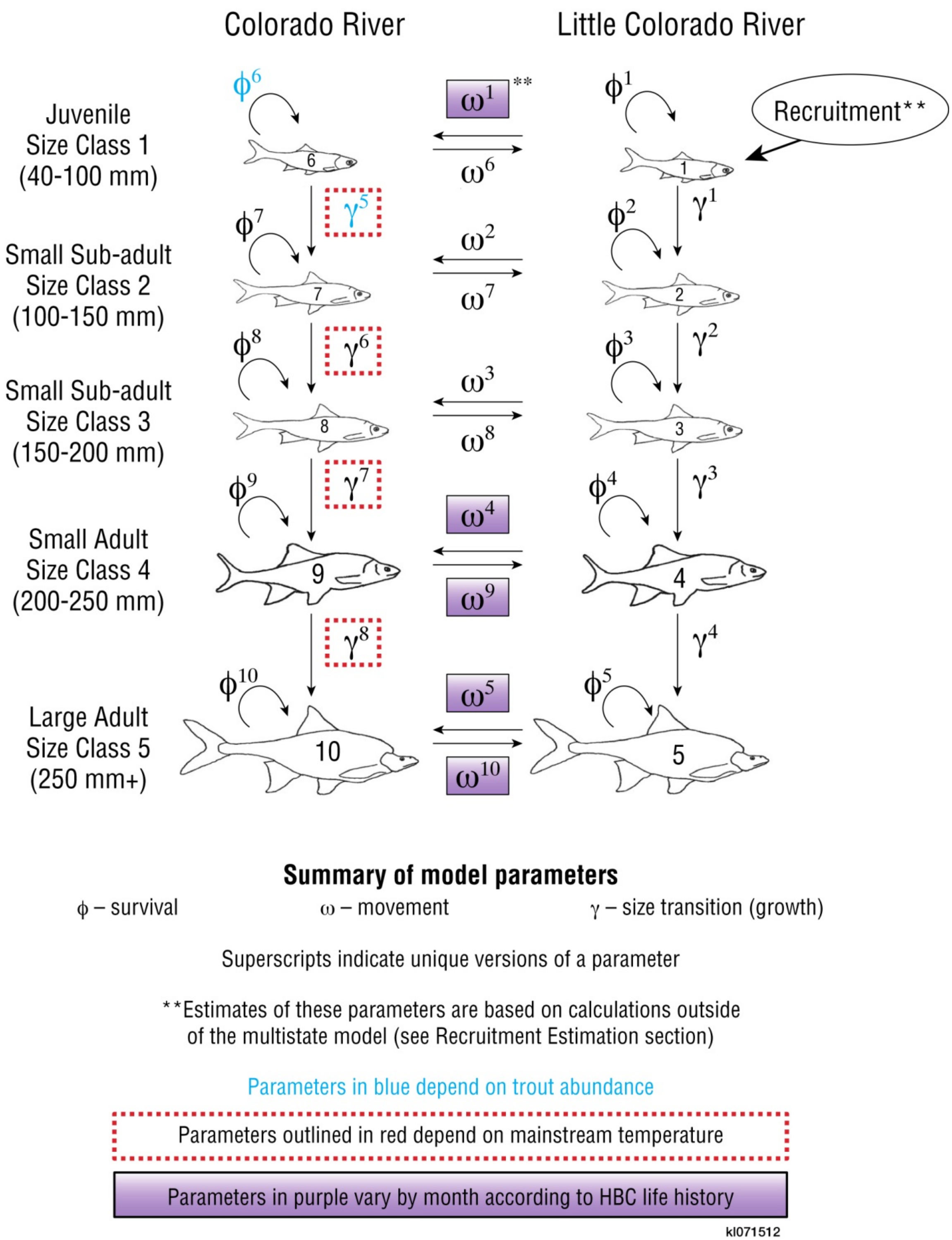

FIGURE F-7 Visual Summary of Humpback Chub Population Model Structure (The number on each fish represents its state number. Modified from Yackulic et al. [2014].)

the Little Colorado River and Colorado River is modeled via movement parameters that vary depending on the month and size class (see Yackulic et al. 2014, for more details regarding movement parameters). In addition to parameters describing survival, movement, and growth, the simulation model also relies on estimates of the starting abundance in each of the 10 states, as well as assumed annual recruit abundance.

All YOY fish recruit to state 1 (i.e., size class 1 fish in the Little Colorado River) in July. Most parameters were estimated directly from data collected during 2009–2013 in the Colorado River and 2009–2012 in the Little Colorado River. Recruitment was approximated by comparing estimated juvenile abundances in the Colorado River and Little Colorado River between 2009 and 2012. This analysis also led to modification of the value for the movement parameter associated with juvenile out-migration from the Little Colorado River (see "Recruitment Estimation" below). Yackulic et al. (2014) had previously speculated that this parameter might be biased, since July marking of juveniles has, until recently, been limited to a small section of the Little Colorado River proximate to the Colorado River that is likely to experience higher overall out-migration than the Little Colorado River as a whole. Recruitment estimates were also influenced by recent research suggesting severely diminished recruitment in years with little winter runoff in the Little Colorado River (Van Haverbeke et al. 2013).

Having estimated the maximum likelihood ("best") values of parameters based mainly on data collected from 2009 to 2013, the simulation model was run using a 20-year sequence of observed temperatures near the Little Colorado River between 1990 and 2009, as well as predictions of rainbow trout abundance for this period from the Glen Canyon and trout movement submodels. These outputs were compared to trends reported in Coggins and Walters (2009), as discussed in "Evaluating Model Predictions," below. Potential uncertainties in model predictions are discussed in "Model Uncertainties" below.

Model Selection and Development

The primary objective for the humpback chub population model was to estimate the effects of mainstem temperature and trout abundance on humpback chub population dynamics (i.e., growth and survival). Six candidate models that represent different a priori hypotheses concerning potential effects were evaluated:

- Model A: Rainbow trout and temperature have no effect on growth and survival.

- Model B: Survival of size class 1 humpback chub in the Colorado River is a logit linear function of rainbow trout abundance. Growth of all size classes in the Colorado River are logit linear functions of temperature with independent intercepts for each size class and a shared slope (a model with different slopes for each size class was considered, a posteriori, but this did not improve the fit considerably). Model B was based on the hypotheses that temperature is a primary control on growth rates and that rainbow trout mainly affect humpback chub by lowering the survival of juvenile humpback chub. This does not mean that rainbow trout effects are solely predatory, as competition with trout could lead to lowered survival if humpback chub were forced to forage longer or in suboptimal habitat, leading to increased predation risk by species other than rainbow trout.

- Model C: As in Model B, but growth of size class 1 fish is a function of rainbow trout abundance in addition to temperature. Rainbow trout are hypothesized to affect humpback chub growth by forcing them into suboptimal habitats and directly consuming food resources that might otherwise be consumed by young humpback chub. This effect is likely to be greatest in young fish because they are frequently found in the nearshore environments that rainbow trout also prefer.

- Model D: As in Model C, but growth of size class 2 humpback chub is also a logit linear function of rainbow trout abundance in addition to temperature.

- Model E: As in Model B, but survival of size class 1 fish is a function of temperature in addition to rainbow trout abundance. Increased temperature is expected to increase the swimming ability of juvenile humpback chub, which should in turn aid them in avoiding predation by a variety of fish species in the system.

- Model F: A combination of Models C and E.

A general model structure modified from Yackulic et al. (2014) was used to fit a series of mark-recapture multistate models using maximum likelihood. For more technical details, see Yackulic et al. (2014). Yackulic et al. (2014) suggested three important features of humpback chub movement between the Little Colorado River and Colorado River:

1. Juveniles out-migrate from the Little Colorado River at a different and higher rate during July through September compared to the rest of the year,

2. Smaller and larger adults spawn at different rates, and

3. There is evidence for a resident Little Colorado River population.

The models that were considered include the first two of these elements, but ignore the third element. The third element is ignored because it would make simulations more difficult and is likely to only apply to a relatively small portion of the adult population (about 15%). Moreover, since the model only considers those fish that move into the mainstem, the movement dynamics of the system can be well represented without this detail.

Monthly temperatures were calculated using data from U.S. Geological Survey (USGS) gage 09383100 located on the Colorado River above the confluence with the Little Colorado River. Rainbow trout abundance in 2012 and 2013 was calculated by averaging trip estimates from the Natal Origins project within each year. Rainbow trout estimates for 2009–2011 were back-calculated based on the relationship between the catch between RM 63.4 and RM 64.8 and the estimated abundance in the same area.

Models were fit using general-purpose optimization algorithms provided by "optim" in R (version 3.0.2) using the BFGS (Broyden, Fletcher, Goldfarb and Shanno) method[1] and were run until convergence of all models was obtained. The variance inflation factor (c-hat) was calculated based on model F, and models were compared using the quasi-Akaike's Information Criterion (qAIC) calculation.[2] Model selection based on qAIC favored model C (summarized in Figure F-7), and estimates from this model were used for further steps. Figure F-8 illustrates the estimated relationships between temperature and trout and various survival and growth parameters. The maximum likelihood estimates from these relationships were used for backcasting, while combinations of the draws from the multivariate normal and critical uncertainties were used to characterize these relationships in simulations conducted to compare LTEMP alternatives.

Recruitment Estimation

The one value needed for simulation that was not estimated in the model selection section is the mean annual recruitment, along with the variability around this mean. Annual recruitment is defined here as the number of YOY humpback chub present in the Little Colorado River in July. By July, most YOY are typically above 40 mm in total length. While some YOY will have left the Little Colorado River before this, several lines of evidence suggest that fish that leave the Little Colorado River before July do not contribute appreciably to population growth, given the temperature typically found in the Colorado River during May and June (Robinson and Childs 2001). Unfortunately, direct estimates of July YOY abundance in the Little Colorado River are not available. However, estimates from the Little Colorado River during September–October are available for 2001 through 2012 in Van Haverbeke et al. (2013), and were used here. The parameters estimated in the model should allow back-calculation of July YOY abundance in the Little Colorado River from the September abundance using the following formula:

$$N_J^1 = \frac{N_S^1}{\left[\emptyset^1\left(1 - \omega_{JS}^1\right)\right]^2}$$

where N_J^1 is recruitment to state 1 in July, N_S^1 is recruitment to state 1 in September, ϕ^1 is the probability of survival during state 1 in the Little Colorado River, and ω_{JS}^1 is the probability of moving from the Little Colorado River to the mainstem Colorado River during the July-to-September period.

1 For details regarding the "optim" function in the "stats" package for Program R, see http://www.inside-r.org/r-doc/stats/optim.

2 For additional information, refer to http://www.inside-r.org/packages/cran/MuMIn/docs/QAIC.

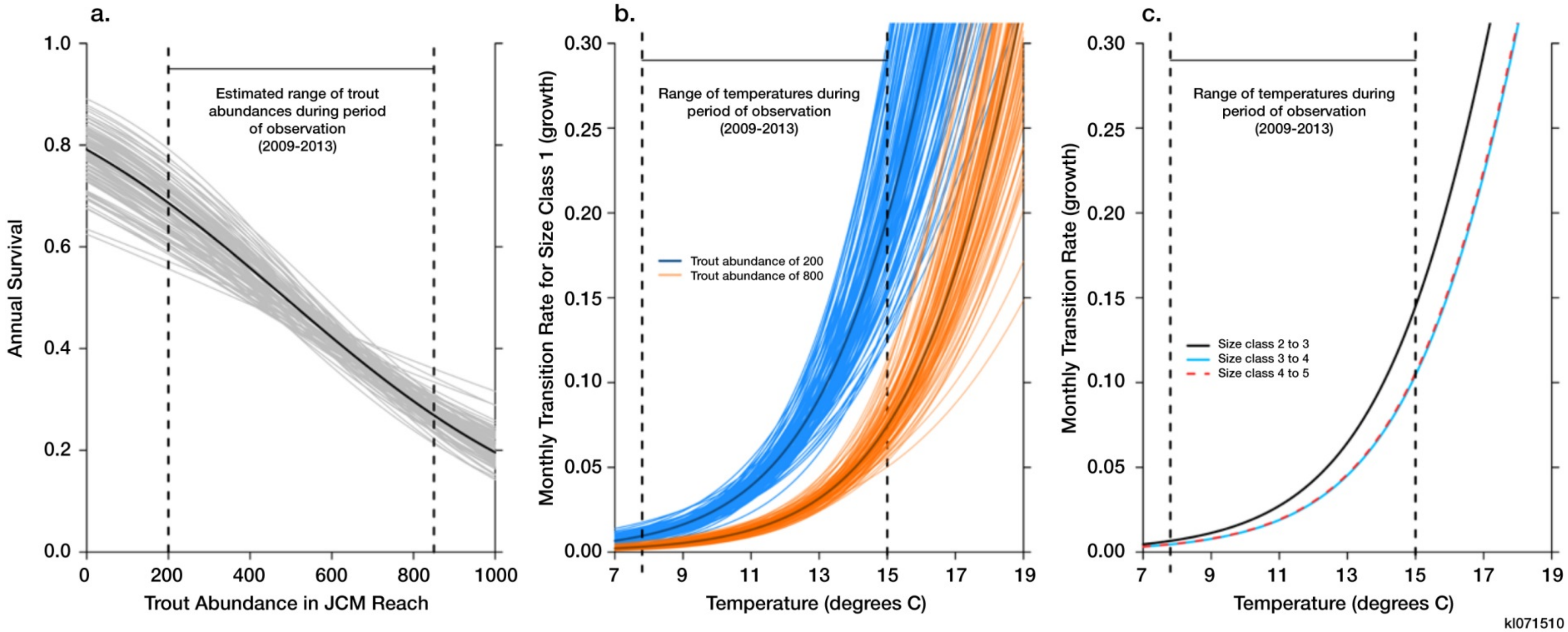

FIGURE F-8 Modeled Effects of Trout Abundance and Temperature on Humpback Chub Survival and Growth ([a] Annual survival of juvenile humpback chub [40–99 mm] declines relative to estimated trout abundance in the Colorado River; gray lines are based on 100 draws from a multivariate normal distribution based on maximum likelihood estimates and associated covariance matrix and give an indication of uncertainty around the maximum likelihood estimates in black. [b] Monthly size transition rates [proportional to growth] as a function of temperature with trout abundance set at either 200 or 800; as in panel [a], dark lines indicate best estimates and lighter lines are draws from a multivariate normal distribution giving an indication of uncertainty. [c] Dependence of size transition rates [growth] for larger fish on temperature. Note that relationships have a common slope but different intercepts. Uncertainty in these rates is comparable to the uncertainty around either of the curves in panel [b], with slightly more uncertainty around the intercept associated with the transition from size class 4 to size class 5.)

However, this approach is based on estimates of the monthly probability of size class fish moving from the Little Colorado River during the July–August and August–September intervals, ω_{JS}^1. Unfortunately, this is the parameter in the multistate model that is most likely to be biased because of details of sampling. In short, until 2013, all July and August marking of humpback chub at the Little Colorado River was limited in its spatial extent to an area near the confluence, which is likely to have a higher rate of export than the Little Colorado River as a whole. Moreover, over two-thirds of the marked fish were marked in 2011 and 2012, years that exhibited large increases in the abundance of size class 1 humpback chub in the juvenile chub monitoring (JCM) reach, thereby suggesting higher export (see Yackulic et al. [2014] for a full discussion). Last, both July recruitment, N_J^1, and movement out of the Little Colorado River, ω_{JS}^1, may exhibit substantial interannual variability (in comparison to, say, adult survival), even though the limited number of marked fish released in July and August into the Little Colorado River in 2009 through 2012 does not allow us to estimate interannual variability in our models.

Therefore, a different approach was taken that is based on the estimated increase in size class 1 abundance in the JCM reach between July and September, as well as the estimated proportion of humpback chub in the JCM reach, τ, and survival rates in both the Little Colorado River, ϕ^1, and Colorado River, ϕ^6. (Note that humpback chub in size class 1 in the Colorado River are frequently 2 or more years old, whereas almost all size class 1 fish caught in the Little Colorado River in the fall are YOY fish). This approach involved solving the following equations for N_J^1 and ω_{JS}^1 for each year:

$$N_J^1 = \frac{N_S^1}{\left[\emptyset^1\left(1-\omega_{JS}^1\right)\right]^2}$$

and

$$N_J^1 = \frac{N_S^6 - N_J^6 \times \emptyset^6 \times \emptyset^6}{\left[\emptyset^1 \times \omega_{JS}^1 \times \emptyset^6 + \emptyset^1 \times \left(1-\omega_{JS}^1\right) \times \emptyset^1 \times \omega_{JS}^1\right] \times \tau}$$

This approach was applied to abundance estimates from 2009 to 2012 and resulted in estimated values of ω_{JS}^1 of 0.15, 0.28, 0.45, and 0.52 (mean, 0.35), with associated values of N_J^1 of 5,000, 17,000, 45,000, and 35,000 (mean, 25,000). Another aspect of recruitment highlighted in Van Haverbeke et al. (2013) is that in years with low runoff between January 1 and May 31, there appears to be weak recruitment, at least in terms of the number of YOY remaining in the Little Colorado River in the fall. Six years between 1990 and 2013 meet this criterion (1990, 1996, 1999, 2000, 2002, and 2006).

When backcasting historical trends, the mean values across years for both ω_{JS}^1 and N_J^1 were used, with the exception of "weak recruitment years" in which recruitment was assumed to be 2,500, based on an examination of estimates in Van Haverbeke et al. (2013). For forecasting, "weak" versus "strong" recruitment years were modeled as a Bernoulli process in which weak years occur with a probability of 0.25 (based on the observed frequency of these hydrologic conditions in the Little Colorado River from 1990 to 2013). For "strong" years, annual recruitment values were drawn from a uniform distribution between 0 and 50,000. For both "strong" and "weak" years, out-migration was chosen randomly from a uniform distribution between 0.15 and 0.55.

Performance Metrics from Humpback Chub Population Submodel

The resource goal identified for humpback chub is to "meet humpback chub recovery goals including maintaining a self-sustaining population, spawning habitat, and aggregations in the humpback chub's natural range in the Colorado River and its tributaries below the Glen Canyon Dam" (EIS Section 1.4). The humpback chub population submodel was used to calculate an estimate of the number of adult (i.e., >200 mm total length) humpback chub that would be present in the aggregation associated with the Little Colorado River for each year of a 20-year simulation period. In order to evaluate and compare the potential for alternatives and long-term strategies to lead to extinction or improvement of the humpback chub population in the Grand Canyon, the modeled minimum number of adult humpback chub that would occur during each 20-year simulation period was used as the performance metric.

Evaluating Model Predictions

Humpback chub population dynamics were backcasted using maximum likelihood estimates of parameters (see "Model Selection and Development" section), with the exception of the parameters related to recruitment and juvenile out-migration from the Little Colorado River (see "Recruitment Estimation" section) and an initial vector of abundances by state. Initially dynamics were simulated using a monthly time step; however, the time scale was coarsened to 6-month intervals so as to minimize potential issues related to numerical diffusion. This was accomplished by calculating a 6-month transition matrix and then removing any transitions of more than one size class and adding these to the cells corresponding to a one size class transition. The initial structure of the population was based roughly on estimates from 2009 to 2012. For the Little Colorado River, abundance by size classes 1–5 was 4,000, 2,500, 1,800, 1,200, and 800, respectively, while corresponding Colorado River abundances by size were 20,000, 7,000, 5,500, 4,000, and 5,000. The simulation model was also provided a 20-year sequence of observed temperature near the Little Colorado River between 1990 and 2009, as well as predictions of trout abundance for this period that resulted from the Glen Canyon trout and trout movement submodels (Sections F.3.1.1 and F.3.1.1, respectively). While the backcasted simulation (Figure F-9) suggests a later decline than the Age-Structured Mark Recapture (ASMR) estimates (Coggins and Walters 2009), followed by a quicker recovery, the patterns are remarkably similar, given that parameters from the simulation model were derived primarily from a more recent period (2009–2013). Moreover, the ASMR estimation method is known to have some biases (Coggins and Walters 2009), so minor discrepancies are expected.

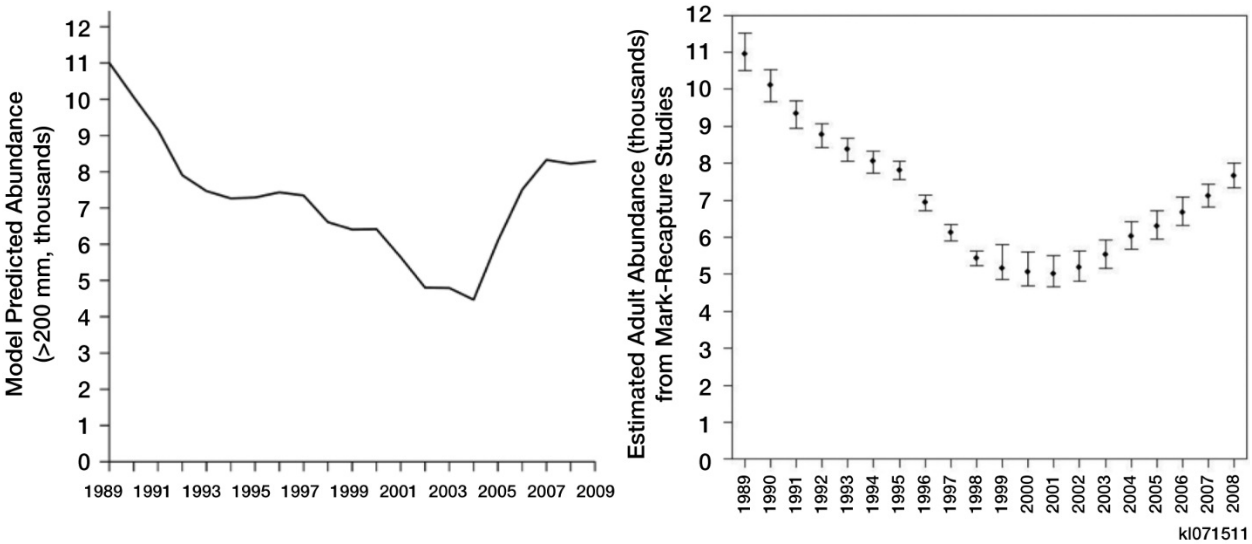

FIGURE F-9 Simulated Adult Abundances from Backcasted Model (left) Compared to Patterns Reported in Coggins and Walters (2009) (right)

Model Uncertainties

The model described here is based on the best available information and does a good job of backcasting the dynamics of humpback chub populations for a period of time (1990–2008) that is separate from the period of time (2009–2012) over which most parameters were estimated. However, like all models, it is only a representation of the actual system it seeks to describe. There are a number of conditions that could lead to dynamics in humpback chub populations that are different from those predicted with the model. Some of these conditions are listed here, in no particular order:

- No portion of this model explicitly models short- or long-term impacts of flows or temperature on the aquatic food base. Flow, particularly an increased frequency of flooding, has the potential to permanently change the composition of the invertebrate assemblage, as has been observed in other regulated rivers (Robinson 2012; also see Section F.2). This shift could be beneficial for both rainbow trout and humpback chub resources, positive for one and negative for the other, or detrimental to both, and initial impacts may differ from long-term consequences. Similar, unpredictable shifts in the invertebrate assemblage could also occur because of long-term changes in release temperatures associated with climate change and lower Lake Powell reservoir elevations.

- Temperature–growth relationships estimated here are based on a relatively short period of record and do not consider seasonal patterns in food availability, light, and turbidity. As such, the humpback chub submodel assumes a temperature of 11°C (52°F) observed in clear water in June during midge emergence will lead to the same growth as a temperature of 11°C (52°F) in August in turbid water. Moreover, monthly mean temperatures at the Little Colorado River confluence from 2009 to 2013 peaked at roughly 15°C (59°F), suggesting that modeling the effects of substantially warmer temperatures on humpback chub populations represents an extrapolation. On the other hand, the model did a reasonable job of backcasting dynamics during the 1990–2009 era, even though monthly temperatures reached 16.7°C (62°F) in one year (2005).

- The humpback chub model does not consider the potential effects of other fish species besides rainbow trout that are already relatively common in the system and known to eat humpback chub (e.g., brown trout and various catfish species), nor does it attempt to account for the negative effects of other warmwater nonnative fishes that could become prevalent if temperatures above 16°C (61°F) become common. Potential effects of cannibalism by humpback chub are also not directly considered by the model.

- Climate change could lead to increases in the proportion of "weak" recruitment years in the Little Colorado River, particularly if winter precipitation in the Little Colorado River watershed becomes less frequent.

F.3.2 Results for LTEMP Alternatives

The results for the rainbow trout-humpback chub model for each of the alternatives (including associated long-term strategies) are summarized in the following sections. Values for the means of the six metrics resulting from the model are summarized in Table F-8. The magnitude of effects on rainbow trout and humpback chub populations are estimated using the performance metrics identified in Sections F.3.1.1, F.3.1.2, and F.3.1.3.

F.3.2.1 Rainbow Trout Performance Measures

This section summarizes the results for the performance measures for rainbow trout that were derived from the rainbow trout-humpback chub model.

Rainbow Trout Population Estimates

The rainbow trout population estimates for the 19 LTEMP alternatives and associated long-term strategies are summarized in Figure F-10. Among all of the long-term strategies evaluated, the modeled average abundance of age-1 (i.e., individuals that are 1 year old) and older rainbow trout during the simulations of 20-year LTEMP periods ranged from about 48,000 to 242,000 individuals in the Glen Canyon reach. Overall means (i.e., mean abundance for all simulations) for the various long-term strategies ranged from approximately 61,000 individuals under long-term strategy E6 to approximately 160,000 individuals under Alternative F

(Table F-8; Figure F-10). The differences among the modeled population levels for rainbow trout reflect the estimated levels of annual recruitment based on the empirically derived flow-dependent regressions in the model that predict that annual recruitment of rainbow trout will increase as a function of greater annual volumes, reduced daily variation in flow between May and August, the occurrence of spring HFEs, and implementation of management actions (i.e., TMFs) that would decrease annual survival of YOY trout (see "Recruitment" in Section F.3.1.1) in high-recruitment years. Table 4.1-1 identifies the experimental elements included in the various long-term strategies, and Appendix E of this EIS describes the number and duration of HFEs that would be expected under the various long-term strategies.

Although there is a considerable amount of overlap in the ranges of the estimates for some long-term strategies, the overall modeled average rainbow trout abundance in the Glen Canyon reach was greatest under long-term strategies C2, C4, and D3, and Alternatives F and G. With the exception of Alternative G, all of these long-term strategies implement spring HFEs and would have steadier flows (at least for the May–August portion of the year) than Alternative A and would not include implementation of TMFs. Although Alternative G would include implementation of TMFs, the annual production of trout would be expected to be very high due to a high proportion of years with HFEs and the steady pattern of flows that would be

TABLE F-8 Summary of Metrics Values from the Rainbow Trout-Humpback Chub Model[a]

Alternatives and Long-Term Strategies	Trout Abundance	Number of Trout ≥16 in. Total Length	Catch Rates (fish/hr)	Number of Out-migrants (fish/year)	Number of Years with Trout Management Flows	Number of Years with Mechanical Removal	Minimum Humpback Chub Population
A	94,667	769	2.11	36,699	0.0	0.07	4,991
B1	74,078	867	1.67	29,586	3.0	0.44	5,392
B2	62,822	920	1.46	24,172	3.1	0.30	5,541
C1	102,342	748	2.23	43,683	6.5	0.00	5,016
C2	150,285	640	3.18	66,890	0.0	0.00	4,527
C3	85,181	830	1.90	33,559	0.0	0.74	5,335
C4	127,129	707	2.72	55,076	0.0	2.80	4,874
D1	92,854	811	2.02	40,784	3.9	1.67	5,247
D2	99,452	796	2.15	43,981	6.9	2.02	5,181
D3	123,448	711	2.63	55,811	0.0	2.95	4,876
D4	93,312	810	2.03	40,936	3.8	1.69	5,241
E1	87,812	826	1.93	37,614	2.6	0.00	5,269
E2	108,046	761	2.33	47,450	0.0	0.00	5,015
E3	73,727	891	1.68	28,499	0.0	0.47	5,477
E4	100,330	781	2.19	42,806	0.0	1.73	5,103
E5	73,848	890	1.68	28,561	0.0	0.00	5,470
E6	60,600	956	1.42	22,415	2.4	0.00	5,708
F	160,297	592	3.37	71,869	0.0	0.00	4,450
G	131,816	702	2.81	58,533	11.0	3.05	4,741

[a] Mean values for 63 modeled hydrology–sediment conditions.

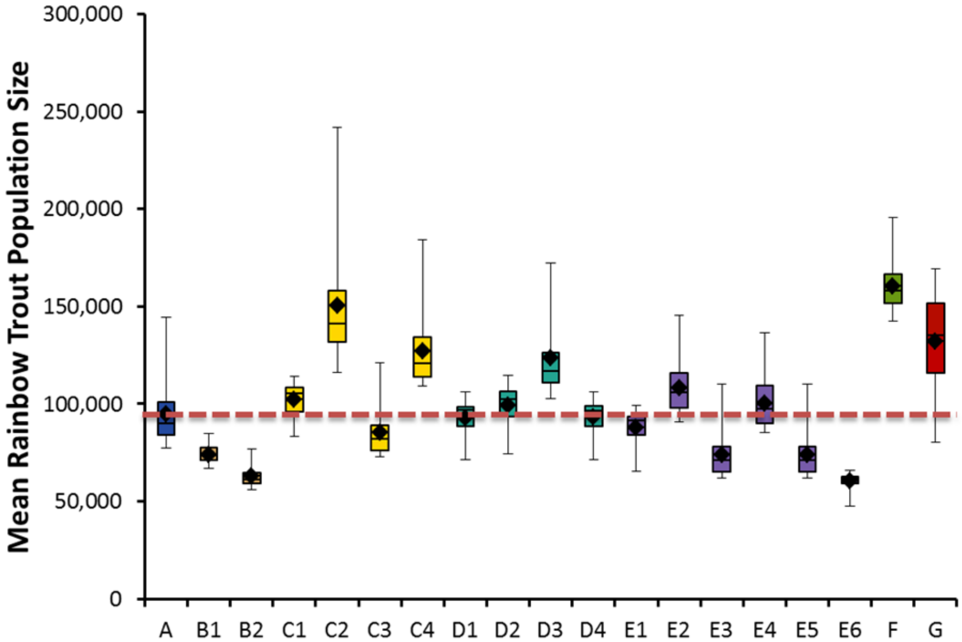

FIGURE F-10 Modeled Average Population Size of Age-1 and Older Rainbow Trout in the Glen Canyon Reach during the 20-year LTEMP Period under LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

maintained throughout the year without monthly differences in flow (other than those needed to adjust operations in response to changes in forecast and other operating requirements such as equalization); even at the highest evaluated levels of effectiveness for TMFs (50% reduction in age-0 trout), average annual recruitment would be expected to be quite high under Alternative G.

The overall modeled average rainbow trout abundance in the Glen Canyon reach was lowest under long-term strategies B1, B2, E3, E5, and E6. These long-term strategies generally would not allow spring HFEs (e.g., long-term strategies E3, E5, and E6) or would be expected to have considerably fewer HFEs during the LTEMP period (e.g., long-term strategies B1and B2) than other long-term strategies, would maintain levels of fluctuations in flow similar to or greater than Alternative A, and (with the exception of long-term strategies E3 and E5) would implement TMFs. Thus, average annual recruitment levels would be expected to be lowest under these alternatives.

Modeled levels of trout abundance were intermediate and similar to Alternative A under long-term strategies C1, C3, D1, D2, D4, E1, E2, and E4. These long-term strategies generally included implementation of combinations of flow actions that would be expected to result in intermediate levels of trout recruitment (e.g., no spring HFEs in all or a portion of the LTEMP period together with higher levels of fluctuation) or included TMFs that would function to control recruitment in years with high levels of trout production (e.g., years with HFEs).

Abundance of Rainbow Trout >16 in. Total Length

The modeled abundance of large rainbow trout in the Glen Canyon reach (i.e., trout that would be larger than 16 in. total length) under LTEMP alternatives are summarized in Figure F-11. Among all the long-term strategies evaluated, the modeled abundance of these larger trout during the simulations of 20-year LTEMP periods ranged from 480 to 1,039 individuals (Figure F-11). Overall modeled means (i.e., mean number of large trout for all simulations) for the various long-term strategies ranged from 592 large fish under Alternative F to 956 large fish under long-term strategy E6 (Table F-8; Figure F-11). Compared to Alternative A, the model suggested that long-term strategies C2, C4, and D3, and Alternatives F and G would have fewer large trout; long-term strategies D1, D4, E1, and E4 would have similar numbers of large trout; and the remaining long-term strategies would have greater numbers of

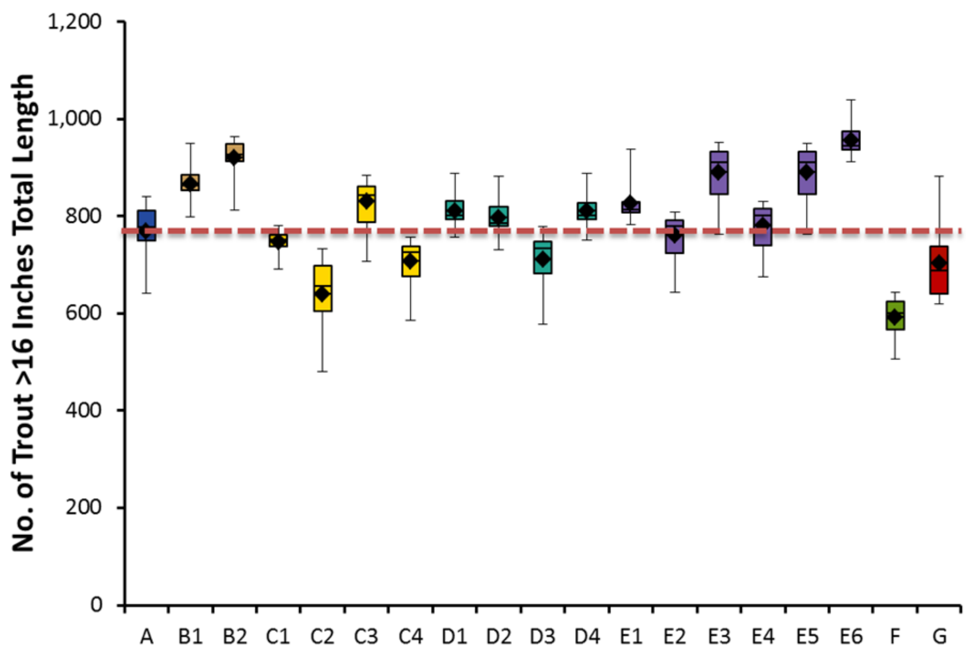

FIGURE F-11 Modeled Mean Annual Number of Rainbow Trout in the Glen Canyon Reach Exceeding 16 in. Total Length during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

large trout (Figure F-11). It is generally expected that the average size of rainbow trout in the population would be inversely proportional to the average population size because of the effects of trout density on growth rates due to competition for food and other resources; this was supported when the modeled results for average number of large trout were compared to the average number of trout in the Glen Canyon reach (Figure F-12). Because of their effect on lowering recruitment levels and population size, long-term strategies (such as long-term strategies B2 and E6) that have fewer HFEs and higher daily fluctuations, and that implement TMFs, are expected to have a greater number of large trout. Table 4.1-1 identifies the experimental elements included in the various long-term strategies, and Appendix E of this EIS describes the number and duration of HFEs that would be expected under the various long-term strategies.

Trout Catch Rates

The modeled angler catch rates for rainbow trout in the Glen Canyon reach under the LTEMP alternatives and long-term strategies are shown in Figure F-13. Modeled average catch rates during the simulations of 20-year LTEMP periods ranged from approximately 1.1 fish/hr to

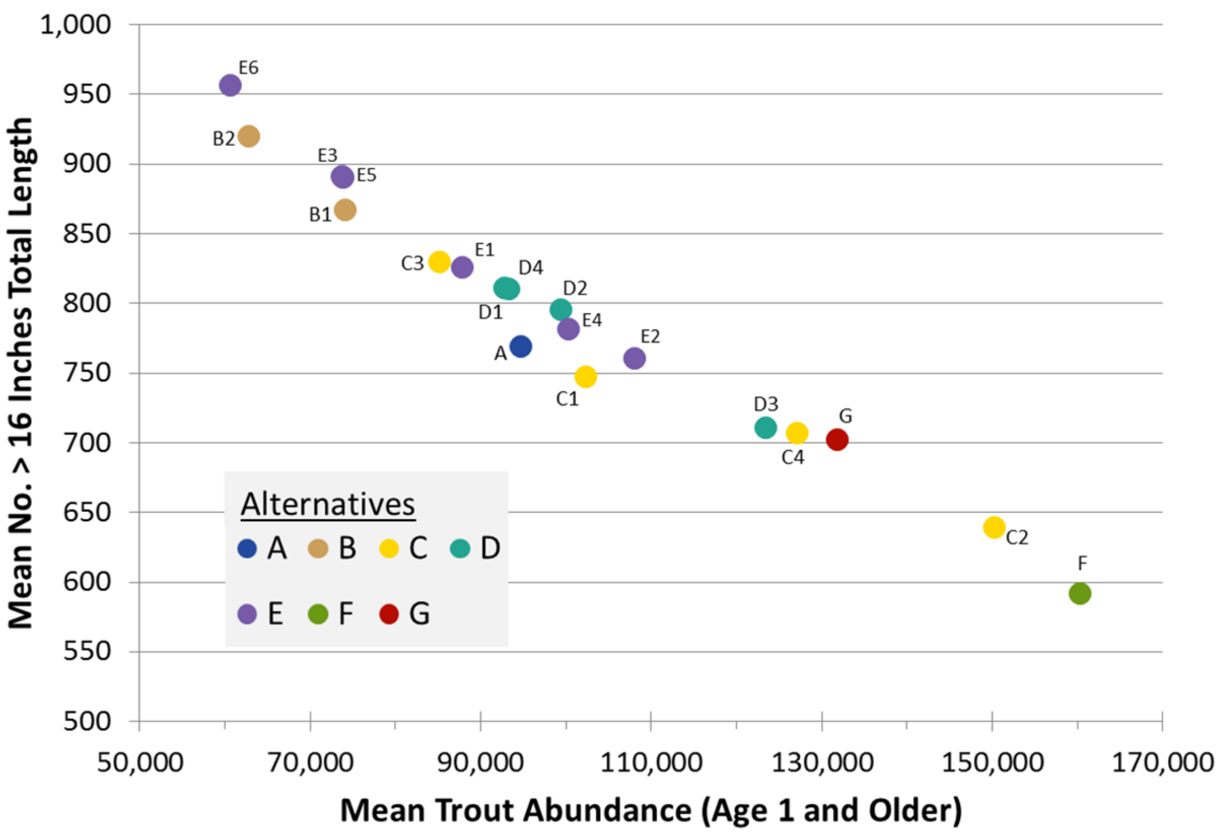

FIGURE F-12 Relationship between Modeled Mean Rainbow Trout Abundance in the Glen Canyon Reach and the Mean Number of Rainbow Trout Exceeding 16 in. Total Length during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies

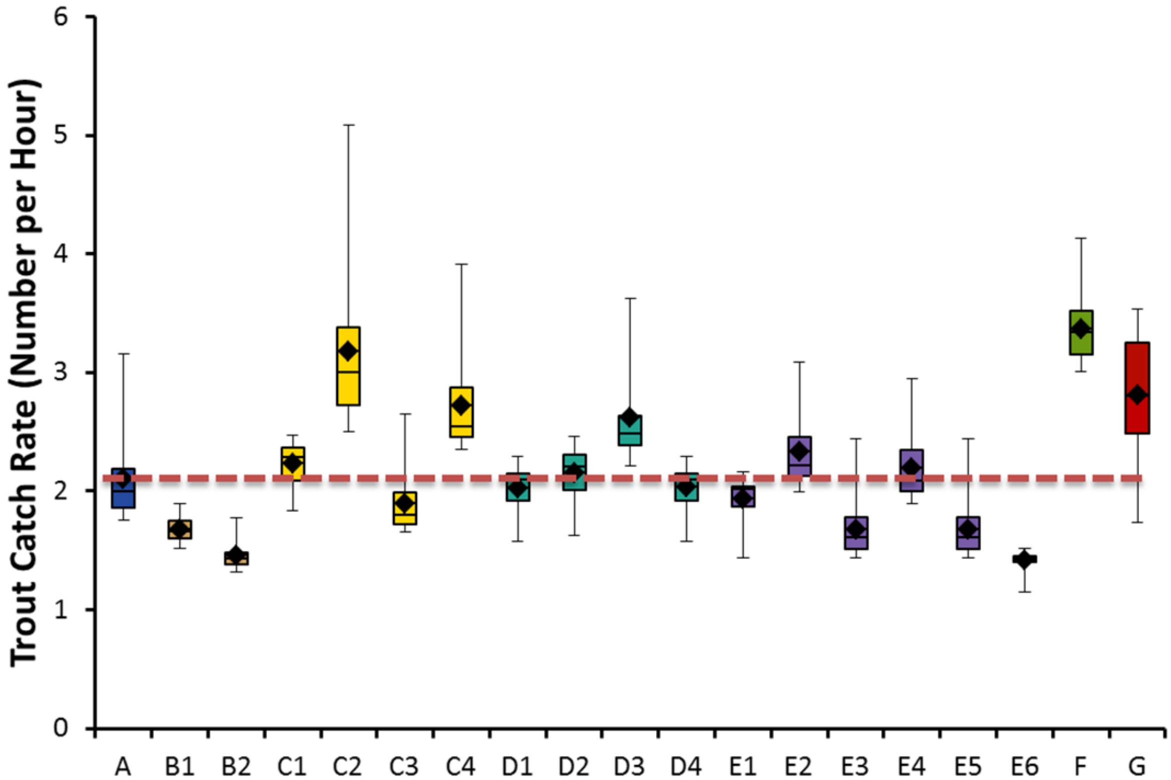

FIGURE F-13 Modeled Mean Annual Angler Catch Rate for Rainbow Trout in the Glen Canyon Reach during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

5.1 fish/hr (Figure F-13). Modeled mean catch rates (i.e., mean catch rates for all simulations) ranged from 1.4 fish/hr under long-term strategy E6 to 3.4 fish/hr under Alternative F (Table F-8; Figure F-13). Compared to Alternative A, the model indicated that long-term strategies B1, B2, C3, E1, E3, E5, and E6 would have lower catch rates; long-term strategies C1, D1, D2, D4, and E4 would have similar catch rates; and long-term strategies C2, C4, D3, and E2, and Alternatives F and G would have higher catch rates (Figure F-13). Although the modeled vulnerability of individual trout to angling varies depending on the age of the trout, modeled average angler catch rates are highly correlated with average population levels of the long-term strategies, as shown in Figure F-14.

For this reason, the same combinations of experimental elements that drive recruitment levels and affect rainbow trout abundance would be expected to drive angler catch rates (see "Recruitment" in Section F.3.1.1 and "Rainbow Trout Population Estimates" in Section F.3.2.1). Thus, long-term strategies that result in more frequent HFEs (especially spring HFEs) have steadier flows and do not include TMFs (e.g., Alternatives F and G and long-term strategies C2

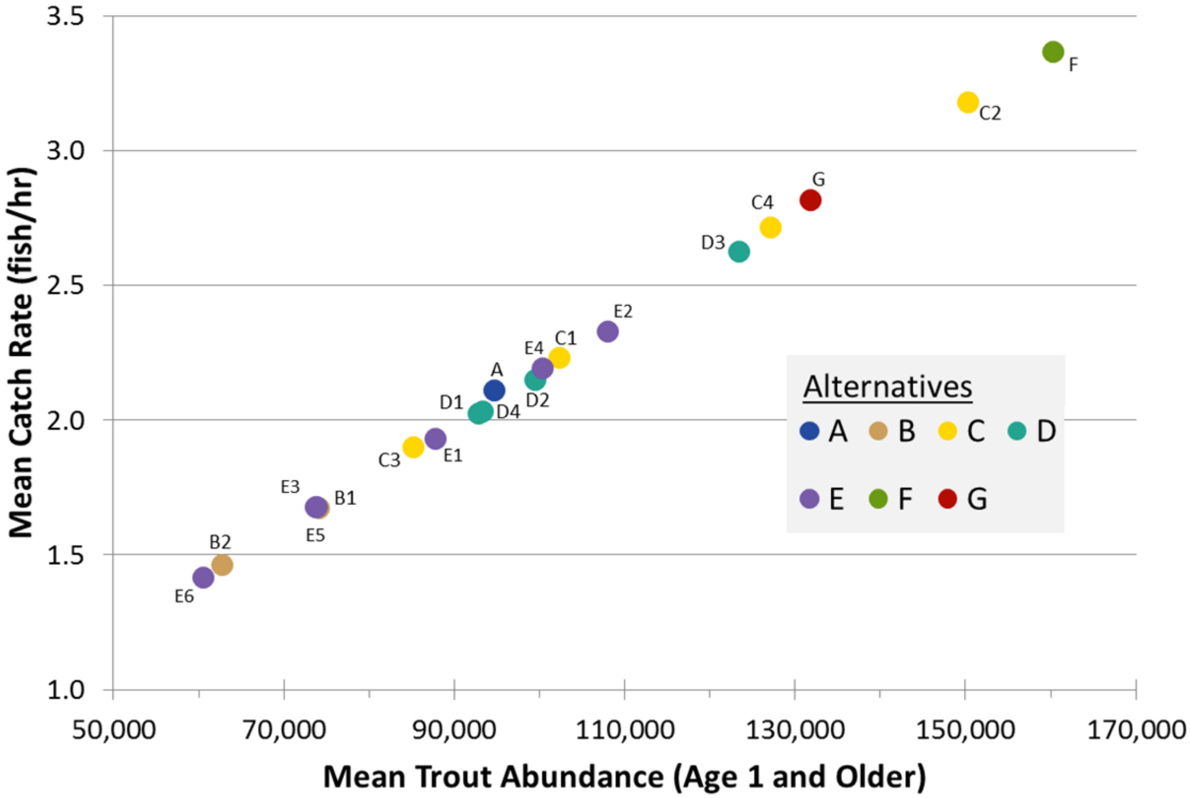

FIGURE F-14 Relationship between Modeled Mean Rainbow Trout Abundance in the Glen Canyon Reach and Mean Angler Catch Rates during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies

and D3) would be expected to have higher trout numbers and would lead to greater angler catch rates for rainbow trout, while long-term strategies that have fewer HFEs, more variable flows, and include TMFs (e.g., long-term strategies B1, B2, and E6) would be expected to have lower trout abundance and lower mean angler catch rates. Table 4.1-1 identifies the experimental elements included in the various long-term strategies, and Appendix E of this EIS describes the number and duration of HFEs for each.

Trout Emigration

The modeled number of trout emigrating (i.e., number of out-migrants) from the Glen Canyon reach into the Marble Canyon reach of the Colorado River under the LTEMP alternatives and long-term strategies are summarized in Figure F-15. Modeled annual number of out-migrants ranged from approximately 18,200 fish/year to 114,900 fish/year (Figure F-15). The modeled mean annual number of out-migrants (i.e., mean number of out-migrants for all simulations) ranged from 22,415 fish/year under long-term strategy E6 to 71,869 fish/year under Alternative F (Table F-8; Figure F-15). Compared to Alternative A, the model indicated that long-term strategies B1, B2, E3, E5, and E6 would have lower numbers of out-migrants; long-term strategies C3 and E1 would have similar numbers of out-migrants; and long-term strategies

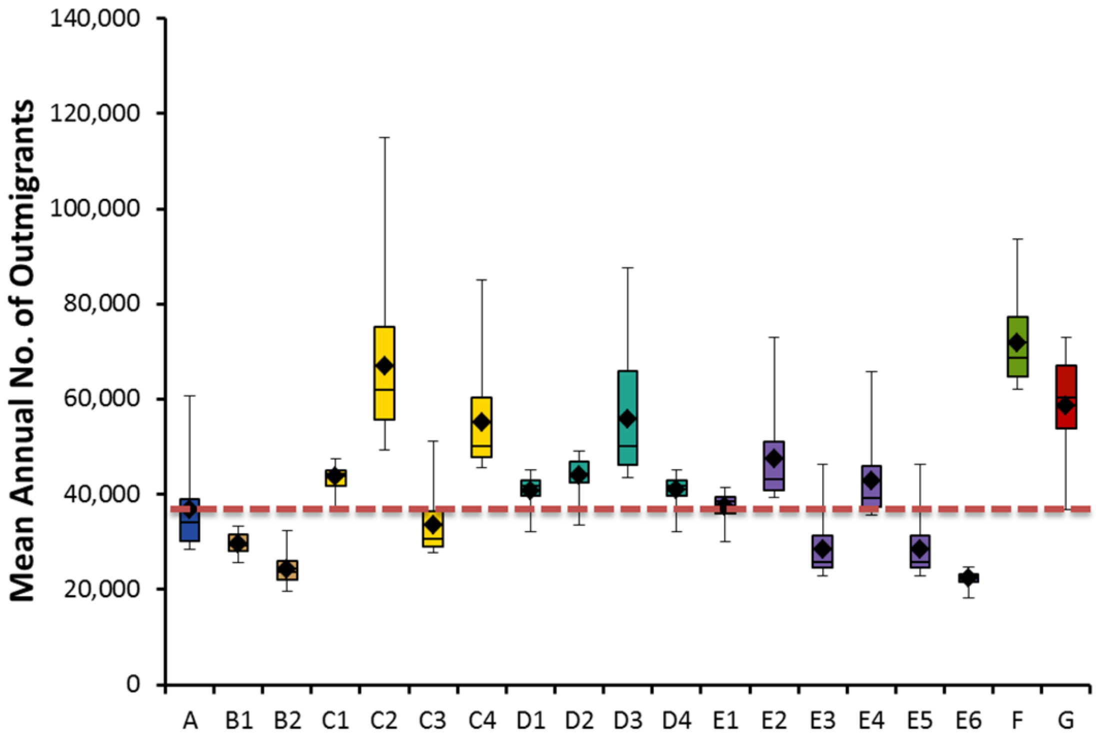

FIGURE F-15 Modeled Annual Average Number of Rainbow Trout Emigrating into the Marble Canyon Reach from the Glen Canyon Reach during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

C1, C2, C4, D1, D2, D3, D4, E2, E4, and Alternatives F and G would have higher numbers of out-migrants (Figure F-13).

As described in Section F.3.1.1, the annual number of trout emigrating from Glen Canyon into Marble Canyon was calculated as a function of the level of trout recruitment during the previous year. Thus, long-term strategies that result in more HFEs (especially spring HFEs), less variability in flows, and do not include TMFs (e.g., Alternatives F and G and long-term strategies C2, C4, D3, and E2) had higher modeled levels of trout emigration than long-term strategies with fewer HFEs, more variable flow regimes, and included TMFs (e.g., long-term strategies B1, B2, and E6). Table 4.1-1 identifies the experimental elements included in the various long-term strategies, and Appendix E of this EIS describes the number and duration of HFEs that would be expected under each.

Mechanical Removal of Trout in the Little Colorado River Reach

The modeled frequency of years in which mechanical removal of trout would be triggered in the Little Colorado River reach under the LTEMP alternatives and long-term strategies is summarized in Figure F-16. Mechanical removal is not included under long-term strategies C1, C2, E1, E2, E5, and E6, and Alternative F. Among the remaining long-term strategies, the average number of years in which mechanical removal was triggered ranged from approximately 0.1 under Alternative A to approximately 3.1 under Alternative G (Table F-8; Figure F-16). The average maximum number of years in which mechanical removal would be triggered is 6.3 out of 20 years under long-term strategy D3. In general, long-term strategies that result in more frequent HFEs (especially spring HFEs), have steadier flows, and do not include TMFs (e.g., Alternatives F and G and long-term strategies C2, C4, and D3) have higher levels of recruitment, increase the number of trout that move downstream to the Little Colorado River reach, and meet conditions in the model that trigger mechanical removal of trout with a greater frequency. Long-term strategies that result in fewer HFEs and more variable flow levels (e.g., long-term strategies B1, B2, and E6) have lower levels of trout recruitment on average; inclusion of TMFs acts to further decrease the potential for large recruitment events. As a consequence, these long-term strategies result in lower numbers of trout entering the Little Colorado River reach and fewer years when mechanical removal is triggered. Table 4.1-1 identifies the experimental elements included in the various long-term strategies, and Appendix E of this EIS describes the number and duration of HFEs that would be expected under each.

F.3.2.2 Humpback Chub Performance Measures

The modeled minimum population sizes for humpback chub adults under the LTEMP alternatives and long-term strategies are summarized in Figure F-17. Modeled minimum adult population sizes ranged from 1,433 to 13,478 fish (refer to upper and lower whiskers in Figure F-17). Overall modeled means (i.e., mean minimum number of adult humpback chub for all simulations) ranged from 4,450 individuals under Alternative F to 5,708 individuals under long-term strategy E6 (Table F-8; refer to diamonds in Figure F-17). The lowest modeled minimum adult population size (1,433 fish) was observed under long-term strategy C2, and the highest modeled minimum adult population size was observed under long-term strategy E6, although the lowest minimum adult population values were relatively similar among all long-term strategies (refer to lower whiskers in Figure F-17). Compared to Alternative A, the model indicated that long-term strategy C2 and Alternative F would have somewhat lower mean minimum adult population sizes; long-term strategies C1, C4, D1, D2, D3, D4, E1, E2, E4 and Alternative G would have similar mean minimum adult population sizes; and long-term strategies B1, B2, C3, E3, E5, and E6 would have higher mean minimum adult population sizes (Figure F-17). These results indicate that although there are small differences among the long-term strategies with regard to the predicted minimum number of adult humpback chub in the Little Colorado River aggregation, all long-term strategies would likely maintain the population above at least 1,000 adults throughout the 20-year LTEMP period.

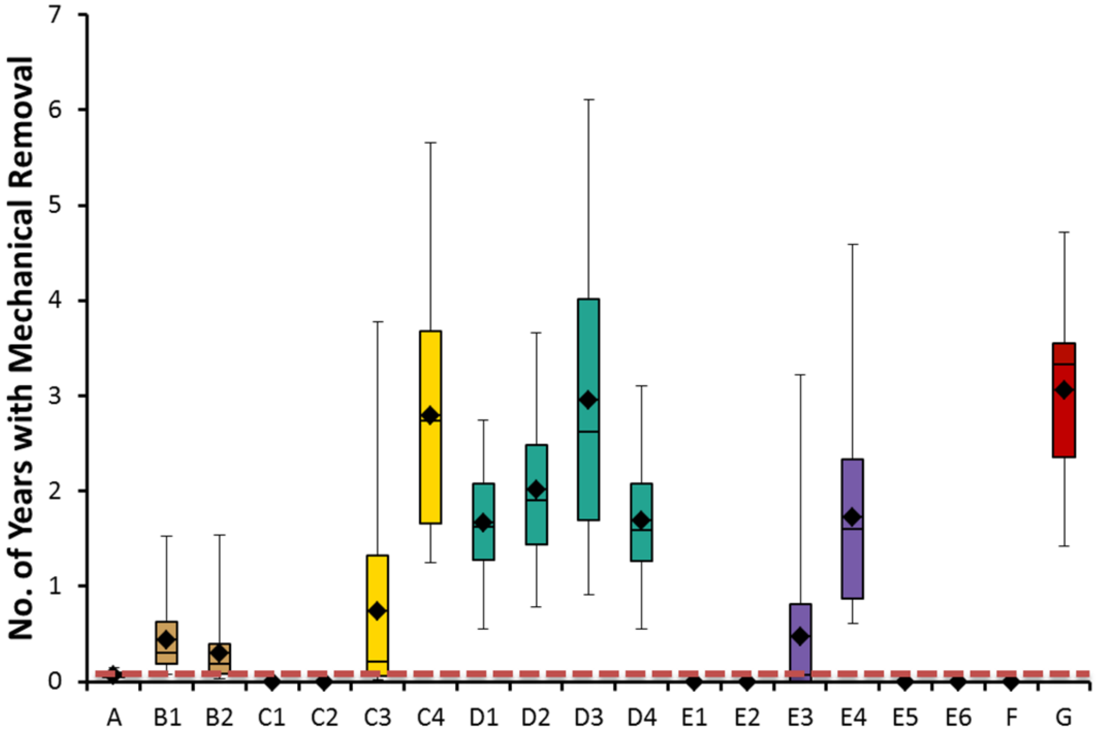

FIGURE F-16 Modeled Frequency of Triggered Mechanical Removal for Rainbow Trout in the Little Colorado River Reach during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

In the humpback chub submodel, the factors that affect annual recruitment and survival of humpback chub are mainstem water temperatures and the number of trout in the Little Colorado River reach (Section F.3.1.3). Because there is little variation among the long-term strategies in modeled mainstem water temperatures at the confluence with the Little Colorado River, the differences in modeled numbers of adult humpback chub among the long-term strategies were primarily affected by the estimated abundance of trout in the Little Colorado River reach where survival of age-0 and juvenile humpback chub and subsequent recruitment of adult humpback chub could be affected by increased competition and predation (e.g., Yard et al. 2011). Because the modeled abundance of trout in the Little Colorado River reach is driven by modeled emigration of rainbow trout from the Glen Canyon reach, there is a strong relationship between the average adult humpback chub population size and the average number of trout emigrating from the Glen Canyon reach for the various long-term strategies (Figure F-18). Refer to the section above entitled "Trout Emigration" for information about the experimental elements of long-term strategies that affect the levels of trout emigration. Although the model predicts that

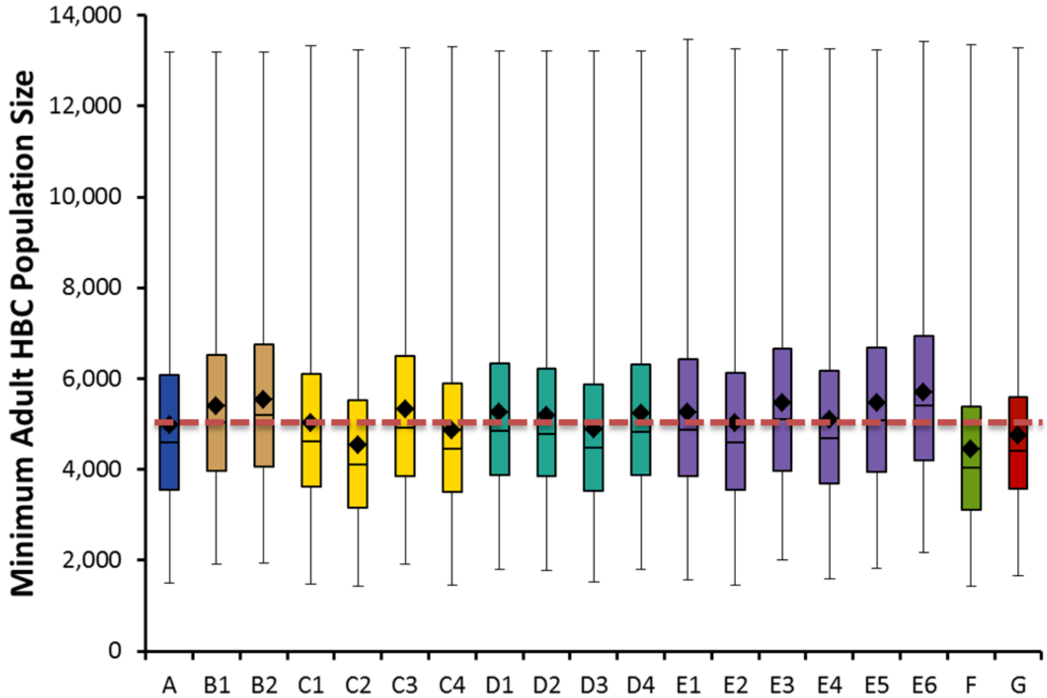

FIGURE F-17 Modeled Minimum Population Size for Humpback Chub (HBC) during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum annual value for all simulations; upper whisker = maximum annual value for all simulations; horizontal dashed line identifies mean value for Alternative A.)

the number of trout at the confluence with the Little Colorado River is related to trout recruitment in the Glen Canyon reach, the actual relationship is unclear and still under investigation.

F.4 MODELING THE EFFECTS OF LTEMP ALTERNATIVES ON TEMPERATURE SUITABILITY

This section describes the modeling approach used to evaluate the effects of LTEMP EIS alternatives on temperature suitability for fishes and invertebrate parasites in the mainstem Colorado River downstream of Glen Canyon Dam. The goal of the temperature suitability modeling was to evaluate the potential for each of the alternatives to result in temperature conditions that would promote maintenance and/or establishment of various fish and invertebrate

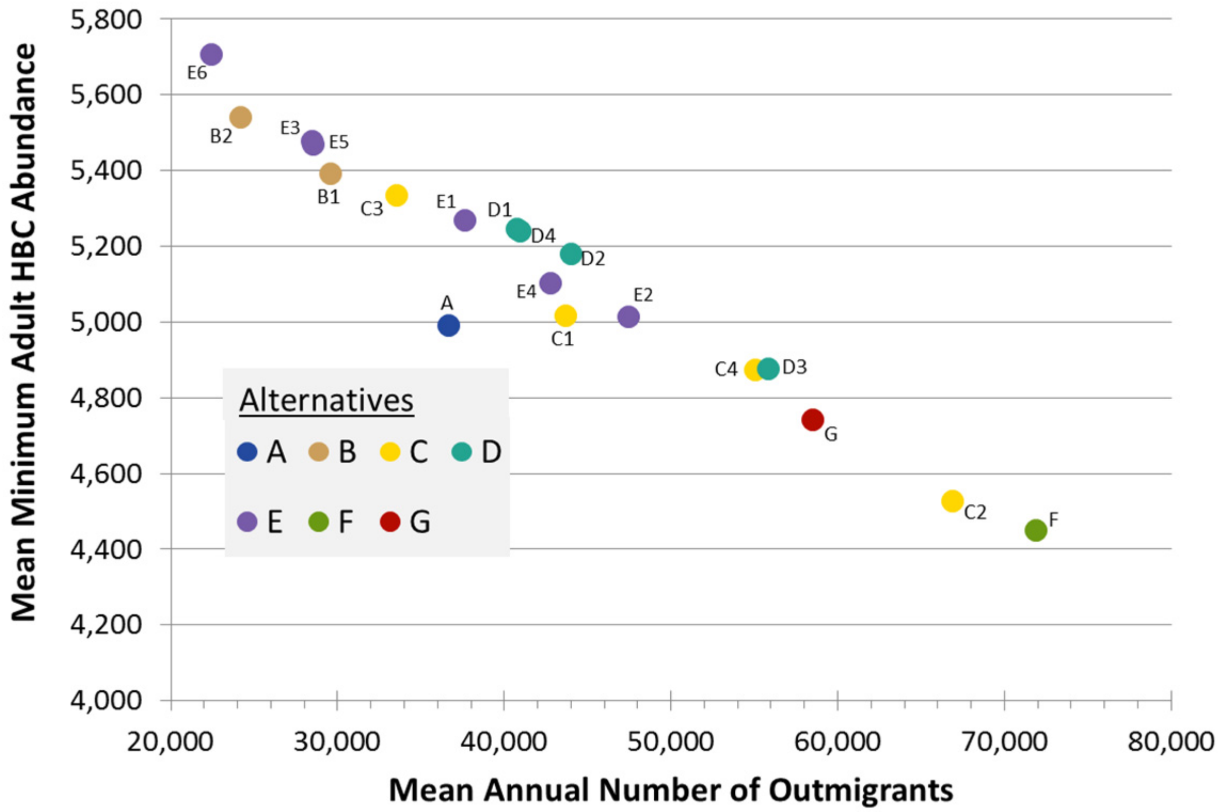

FIGURE F-18 Relationship between Modeled Mean Numbers of Rainbow Trout Out-migrants from the Glen Canyon Reach and the Modeled Mean Minimum Abundance of Adult Humpback Chub during the 20-year LTEMP Period under the LTEMP Alternatives and Long-Term Strategies

species of management concern. In particular, the temperature suitability modeling is intended to evaluate effects of alternatives on temperature suitability for four species groups:

1. Temperature suitability for establishment and maintenance of self-sustaining aggregations of humpback chub at various river locations in the Colorado River downstream of Glen Canyon Dam;

2. Temperature suitability for establishment and maintenance of self-sustaining populations of native warmwater fish other than humpback chub at various locations in the Colorado River downstream of Glen Canyon Dam;

3. Temperature suitability for establishment and maintenance of self-sustaining populations of nonnative fish species at various locations in the Colorado River downstream of Glen Canyon Dam; and

4. Temperature suitability for establishment and maintenance of self-sustaining populations of parasitic invertebrate species at various locations in the Colorado River downstream of Glen Canyon Dam.

The following sections describe the general modeling approach for evaluating temperature suitability, specific modeling considerations applied in order to implement the modeling approach for each of the species groups to be evaluated, the input data needs and sources for each of the ecological components, and the approach for statistically evaluating the output of the models in order to compare the effects of the various operational alternatives on temperature suitability for each species group.

F.4.1 Model Overview

In general, the temperature suitability modeling considers how well mainstem water temperatures at selected locations downstream of Glen Canyon Dam would meet the temperature requirements for three life history components—spawning, egg incubation, and growth—for each species group evaluated. To accomplish this, monthly water temperature values in a multiyear time series were compared to temperature suitability profiles for life history components of each species group considered. The seasonal timing or period of the year during which the temperature needs for each life history component must be met is taken into account by the model. Possible values for temperature suitability can theoretically range from 0 (completely unsuitable for one or more life history component) to 1 (magnitude and timing of temperatures would be optimal for all life history components). However, since optimal conditions for all life history components cannot be simultaneously met in many cases due to different optimal temperatures during overlapping time frames, the maximum attainable value for a given species would generally be less than one.

The temperature suitability modeling evaluates the potential for all life history components to be met in the mainstem river, even though some species are known to sometimes use tributaries to accomplish particular needs. Thus, the model can predict relatively low temperature suitability for some areas even though species populations appear to be abundant and self-sustaining. In addition, modeled water temperatures used as inputs do not consider the potential for warming near tributary mouths, backwater habitats, or in shallow nearshore areas. Thus, the results of temperature suitability modeling are used to compare relative effects of alternatives on species-specific temperature needs in the mainstem Colorado River, rather than as an exact predictor of the potential for the presence or absence of fish or parasite species at particular locations.

For fish species, the model considers the suitability of each day's water temperature for three life history components (spawning, egg incubation, and growth). The model bases the potential for self-sustaining populations of fish species being successful on the combined temperature suitability scores for spawning, incubation, and growth, and it is assumed that some level of both mainstem spawning and egg incubation would be required to support self-sustaining populations of fish species. The annual potential for successful spawning and egg incubation is assumed to be related to the suitability of the annual temperature regimes for spawning and egg incubation during the spawning and egg incubation periods. It was assumed that the potential for successful rearing and survival of fish species within the mainstem at each evaluation location was related to the suitability of temperatures throughout the year for growth. The suitability of various temperatures for spawning, egg incubation, and growth needs of fish was calculated using triangular probability functions[3] based upon reported suitable ranges and optimal temperatures for each life history aspect of each species (Valdez and Speas 2007).

For parasite species, the model bases the potential for unacceptable parasite conditions on the temperature suitability scores for host activity and infestation. It is assumed that both elevated host activity and infestation rates would be needed to result in unacceptable infestations of the parasite species and the annual potential for unacceptable infestations is assumed to be related to the suitability of the temperature regimes for host activity and infestation throughout the year. The suitability of various temperatures for host activity and infestation needs of a group of four parasite species was calculated using triangular probability functions based on the reported range of suitable temperatures and the reported optimal temperature for each species (Valdez and Speas 2007). The model calculates daily temperature suitability scores for the life history components based on the triangular suitability relationships and the seasonal time periods during which the temperature needs for each life history component must be met.

Annual temperature suitability for each life history component is calculated as the mean of the daily suitability values that fall within the specified seasonal time period during a given water year. The overall annual temperature suitability for each species is calculated as the geometric mean of the annual temperature suitability scores for the applicable species-specific life history components. Temperature suitability over a 20-year period is based on the mean of the annual temperature suitability values. Evaluations were conducted for each river location to be assessed or using the overall annual means for combinations of downstream locations. The mean of the annual suitability scores for multiple fish or parasite species was used as an indication of the overall suitability of each year's temperature regime for groups of native fish, nonnative fish, or parasite species.

The LTEMP temperature suitability model requires inputs pertaining to daily water temperatures for each of the downstream locations to be assessed, and it requires identification of temperature requirements for the life history aspects of each species to be evaluated. Species-specific temperature requirement information includes the minimum, optimal, and maximum suitable temperatures for important life history components and information describing the appropriate months of the year during which conditions for each life history component should be met. Table F-9 summarizes the input data needs and the anticipated sources of the input values. The model is formulated to consider daily water temperatures for multiyear periods. The daily water temperature input values were derived from external modeling (i.e., not calculated within the LTEMP temperature suitability model) following formulas developed by Wright et al. (2009) to predict mean monthly water temperatures at various locations downstream of Glen

[3] With the triangular functions used, the temperature suitability value rises linearly from 0 at the minimum suitable temperature to 1 at the optimum temperature, then falls linearly from 1 at the optimum to 0 at the maximum suitable temperature). Each of these functions was based on species-specific temperature requirements as reported by Valdez and Speas (2007). See Figure F-19 for example functions.

TABLE F-9 Description of Input Parameters for the LTEMP Temperature Suitability Model

Input Parameter	Description of Input Data	Comments
$TW_{x,y}$	Mean daily water temperature (°C) for a specific day (*x*) in a given year (*y*)	Provided by water temperature modeling. Although daily water temperatures are used as inputs into the model, modeled mean monthly water temperatures were used to provide the mean daily temperatures to be used within the months for each year. The model is formulated to accommodate multiyear traces of daily temperature data. A water temperature time series covering the same time period was developed for each downstream location.
$T_{Min(s,l)}$	The minimum suitable temperature (°C) to meet a given life history need (*l*) for a given species (*s*)	Values obtained from Valdez and Speas (2007).
$T_{Max(s,l)}$	The maximum suitable temperature (°C) to meet a given life history need (*l*) for a given species (*s*)	Values obtained from Valdez and Speas (2007).
$T_{Opt(s,l)}$	The optimum suitable temperature (°C) to meet a given life history need (*l*) for a given species (*s*)	Values obtained from Valdez and Speas (2007).
$MonthStart_{(s,l)}$	The beginning month of the water year during which a given life history need (*l*) for a given species (*s*) should be met	Used to identify the beginning of the appropriate time period for meeting each species–life history component combination.
$MonthEnd_{(s,l)}$	The ending month of the water year during which a given life history need (*l*) for a given species (*s*) should be met	Used to identify the end of the appropriate time period for meeting each species–life history component combination.

Canyon Dam based on assumed meteorological conditions, the expected magnitude of water releases, and the temperature of the water being released from Lake Powell for each of the LTEMP alternatives. The temperature suitability for each alternative/long-term strategy was evaluated using a total of 63, 20-year temperature input scenarios generated from conditions expected during operations for a range of hydrology–sediment trace combinations. The temperature suitability model was implemented using R (R Core Team 2013; see http://www.r-project.org/about.html).

The following sections provide specific information regarding implementations and results of the temperature suitability modeling approach to evaluate suitability for (1) self-sustaining aggregations of humpback chub; (2) self-sustaining populations of native warmwater fish species other than humpback chub; (3) self-sustaining populations of coldwater and warmwater nonnative fish species; and (4) establishment and maintenance of invasive parasitic invertebrate species.

F.4.2 Humpback Chub Aggregations

The temperature suitability model evaluates how well alternatives would provide mainstem water temperatures suitable for spawning, egg incubation, and growth of humpback chub at reported aggregation locations. The model based the potential for a self-sustaining aggregation of humpback chub becoming successfully established at each location on the combined potential for successful spawning, successful incubation, and successful growth of humpback chub. The time series of water temperatures was based upon estimated water temperatures for eight mainstem Colorado River locations (Table F-10) where humpback chub aggregations have been reported to occur. As described in Section F.4.1, the water temperatures used as inputs for these locations were modeled using a water temperature model developed by Wright et al. (2009).

It was assumed that mainstem spawning would be required to support self-sustaining aggregations at all locations except for the aggregation at the confluence of the mainstem and the Little Colorado River (RM 61), where successful tributary spawning is known to occur. Thus, except for the Little Colorado River aggregation, the annual potential for successful spawning is assumed to be related to the suitability of temperature regimes in the mainstem Colorado River for spawning. The potential for successful spawning at various temperatures was calculated using a triangular probability function based upon the reported range of suitable spawning temperatures (16–22°C) (61–72°F) and the reported optimal spawning temperature (18°C) (64°F) for humpback chub (Valdez and Speas 2007). The calculated suitability of various water temperatures for successful humpback chub spawning is shown in Figure F-19.

April, May, and June were identified as encompassing the possible spawning period for humpback chub aggregations (Figure F-20), based on observations of fish in spawning condition reported by Valdez and Ryel (1995) for aggregations and by Gorman and Stone (1999) for spawning in the Little Colorado River. The annual suitability values for spawning were set to a value of 1 for the Little Colorado River aggregation, since water temperature in the Little

TABLE F-10 Humpback Chub Aggregation Locations

Aggregation Location	River Mile (RM)[a]
30-mile	RM 30
Little Colorado River confluence	RM 61
Bright Angel Creek	RM 88
Shinumo Creek	RM 108
Stephen Aisle	RM 119
Middle Granite Gorge	RM 125
Havasu Creek	RM 157
Pumpkin Spring	RM 213

[a] River mile distances are calculated as the distance downstream from the Lee Ferry gage.

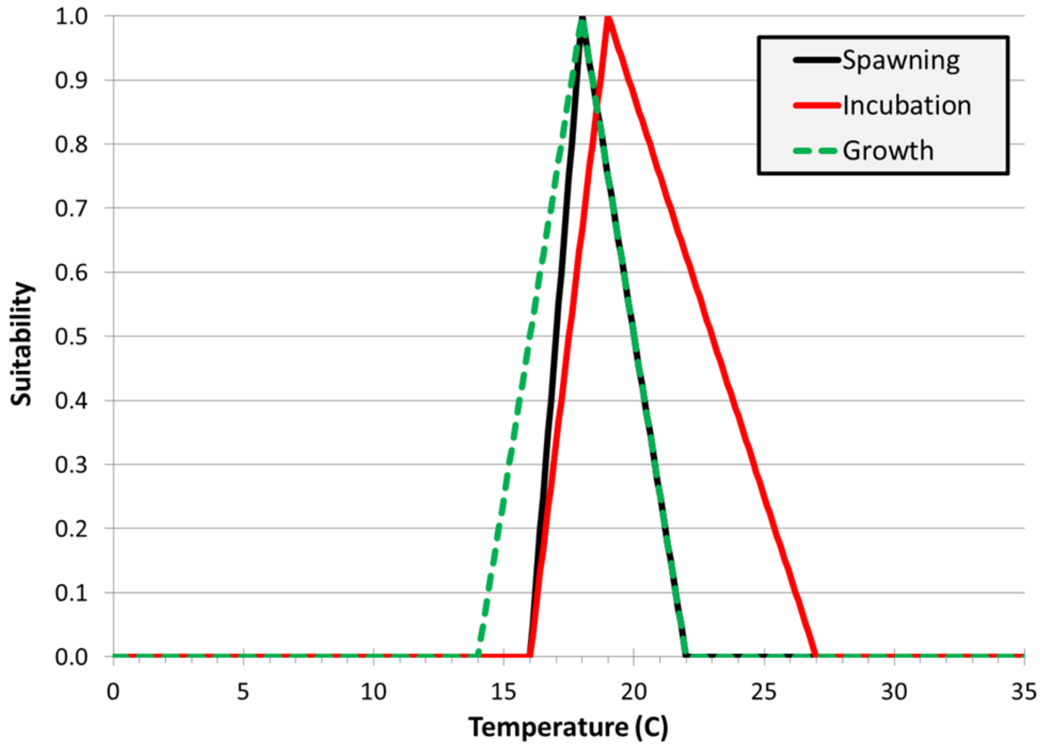

FIGURE F-19 Suitability for Spawning, Egg Incubation, and Growth of Humpback Chub as a Function of Water Temperature (based on minimum, maximum, and optimum temperature values presented in Valdez and Speas 2007)

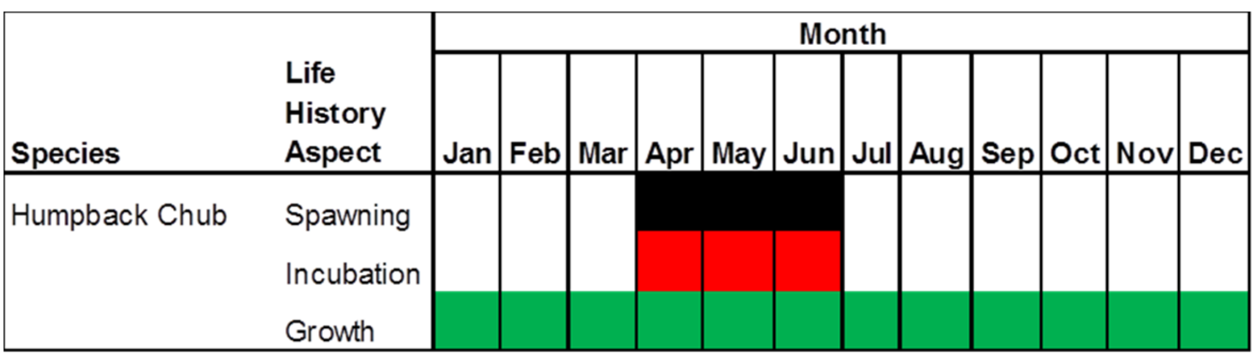

FIGURE F-20 Months for Which Annual Temperature Suitability for Specific Life History Aspects of Humpback Chub Were Calculated

Colorado River is known to support spawning needs for this aggregation. The potential for successful spawning for each aggregation during a given water year was calculated as the average of the estimated suitability scores during April through June (Figure F-20).

It was assumed that mainstem egg incubation would be required to support self-sustaining aggregations at all locations except for the aggregation at the confluence of the mainstem and the Little Colorado River (RM 61), where successful tributary spawning is known to occur. The suitability for incubation in the Little Colorado River (RM 61) aggregation was assumed to be 1. At other aggregation locations, the annual potential for successful egg incubation was assumed to be related to the suitability of mainstem temperature regimes for incubation during the spawning period, because incubation of humpback chub eggs may require as little as 3 days at optimal temperatures. Thus, it was assumed that the spawning period of April, May, and June also encompassed the egg incubation period for aggregations (Figure F-20). The suitability of various temperatures for egg incubation was calculated using a triangular probability function based upon the reported range of suitable egg incubation temperatures (16–27°C) (61–81°F) and the reported optimal egg incubation temperature (19°C) (66°F) for humpback chub (Valdez and Speas 2007; Figure F-19).

It was assumed that the potential for successful rearing of humpback chub within the mainstem at each aggregation location is related to the suitability of temperatures throughout the year for humpback chub growth. The suitability of various temperatures for growth of humpback chub was calculated using a triangular probability function based upon the reported range of suitable temperatures (16–22°C) (61–72°F) and the reported optimal temperature (18°C) (64°F) for growth (Valdez and Speas 2007; Figure F-19). The annual suitability of daily temperatures for growth was calculated as the mean of daily suitability values during the entire water year (Figure F-20).

The geometric mean of the annual temperature suitability values for spawning, egg incubation, and growth was used as an indicator of the annual potential for an aggregation to be successful (and self-sustaining) at a particular location. The arithmetic mean of the annual suitability scores for each of the eight aggregation locations was used as an indication of the overall relative suitability of each year’s temperature regime for supporting humpback chub aggregations in the mainstem Colorado River downstream of Glen Canyon Dam.

F.4.2.1 Historic Temperature Suitability for Humpback Chub

Historic temperature suitability of mainstem water temperatures for humpback chub aggregations was examined using modeled water historic temperatures at the aggregation locations for a 23-year period from October 1, 1989, through September 30, 2012 (water years 1990–2012), as the temperature inputs (Figure F-21). The annual values of the modeled historic temperature suitability for the various aggregation locations are summarized in Figure F-22.

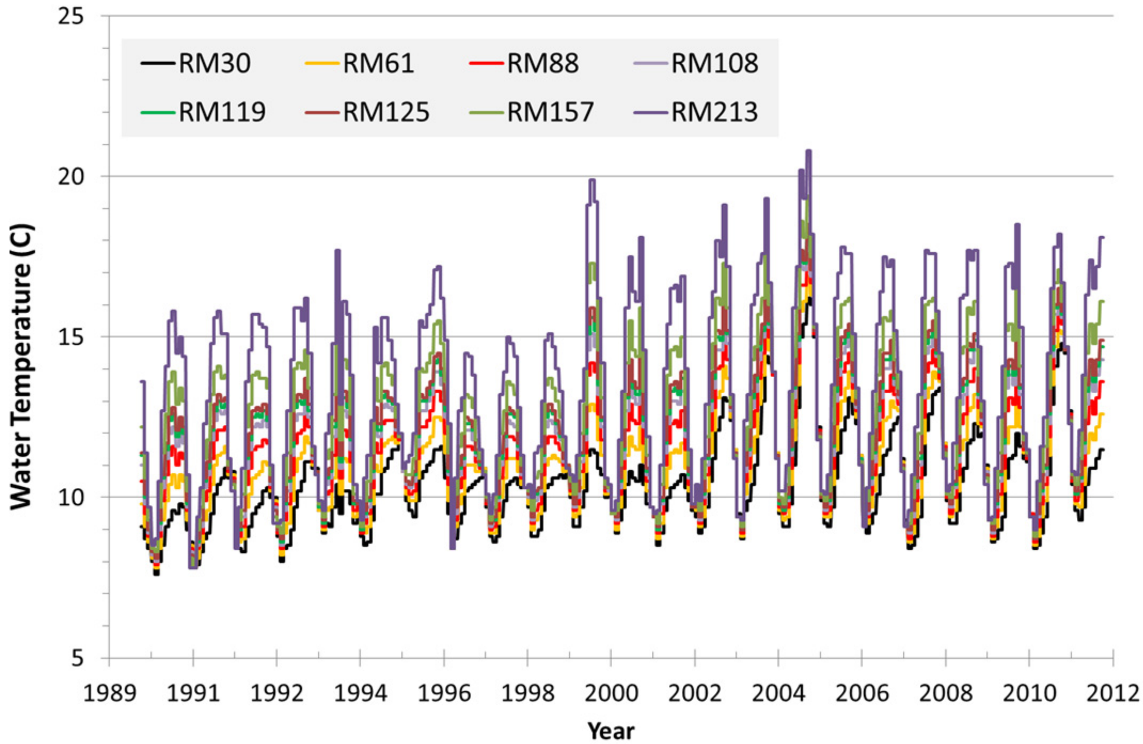

FIGURE F-21 Modeled Historic Water Temperatures in the Colorado River at Humpback Chub Aggregation Locations, Water Years 1990–2012 (Source: Williams 2013)

F.4.2.2 Results for LTEMP Alternatives

Figure F-23 summarizes the temperature suitability for humpback chub at aggregation locations under the LTEMP alternatives and long-term strategies. Modeled main channel water temperature suitability for humpback chub was relatively low and similar to Alternative A under all the long-term strategies for most aggregation locations. Modeled mean annual main channel temperature suitability for humpback chub at RM 61 (the Little Colorado River confluence) was slightly higher under Alternative F than under the other long-term strategies (Figure F-23), because the lower summer and fall flows of this alternative resulted in warmer water that would benefit growth during those seasons; note that the overall suitability score for RM 61 reflects temperature suitability for growth in the main channel, and optimal spawning and egg incubation temperatures in the Little Colorado River where the species spawns. Because the water warms as it travels downstream from the dam (for spring through fall months), temperature suitability improves with increasing distance. At RM 213, mean annual temperature suitability for humpback chub was similar to Alternative A under all long-term strategies except for C1, C2, C3, and C4, and Alternative F. Compared to Alternative A, long-term strategies C1, C2, C3, and C4 were slightly lower, although differences were small (Figure-F-23). Modeled temperature suitability at RM 213 was lowest under Alternative F (Figure F-23), reflecting the higher, colder flows expected to occur under this alternative during spawning and egg incubation periods (April through June). Based on these results, the combined suitability of mainstem temperatures for spawning, egg incubation, and growth by humpback

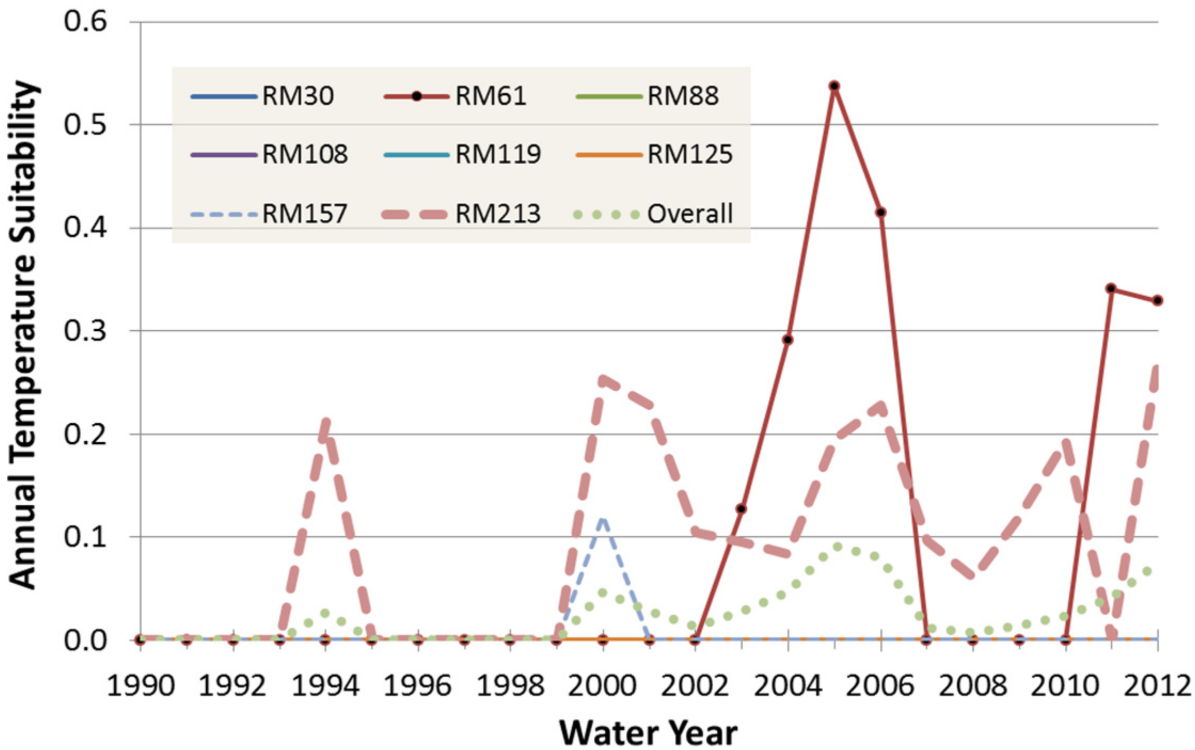

FIGURE F-22 Output from the Temperature Suitability Model for Humpback Chub Aggregation Locations Based on Modeled Water Temperatures for Water Years 1990–2012

chub in the downstream-most aggregation sites is anticipated to be negatively affected compared to current conditions under Alternative F; however, for the other long-term strategies, suitability would remain similar to the low historic levels, as represented by the suitability under Alternative A (the no-action alternative). It should be noted that, historically, there have been years where the magnitude and timing of mainstem water temperatures have likely coincided to allow spawning and egg incubation to occur in some of the downstream aggregation areas; however, the overall average suitability has likely been low (Figure F-22).

F.4.3 Other Native Fish

The temperature suitability model for native fish evaluates how well alternatives provide mainstem water temperatures suitable for spawning, egg incubation, and growth of four species of warmwater native fish other than humpback chub (speckled dace [*Rhinichthys osculus*], razorback sucker [*Xyrauchen texanus*], flannelmouth sucker [*Catostomus latipinnis*], and bluehead sucker [*C. discobolus*]). In order to account for changes in water temperatures as water released from Glen Canyon Dam travels downstream, evaluations of temperature suitability were conducted for five mainstem Colorado River locations (Table F-11). As described in Section F.4.1, the time series of water temperatures used as inputs for these locations are generated using a water temperature model developed by Wright et al. (2009).

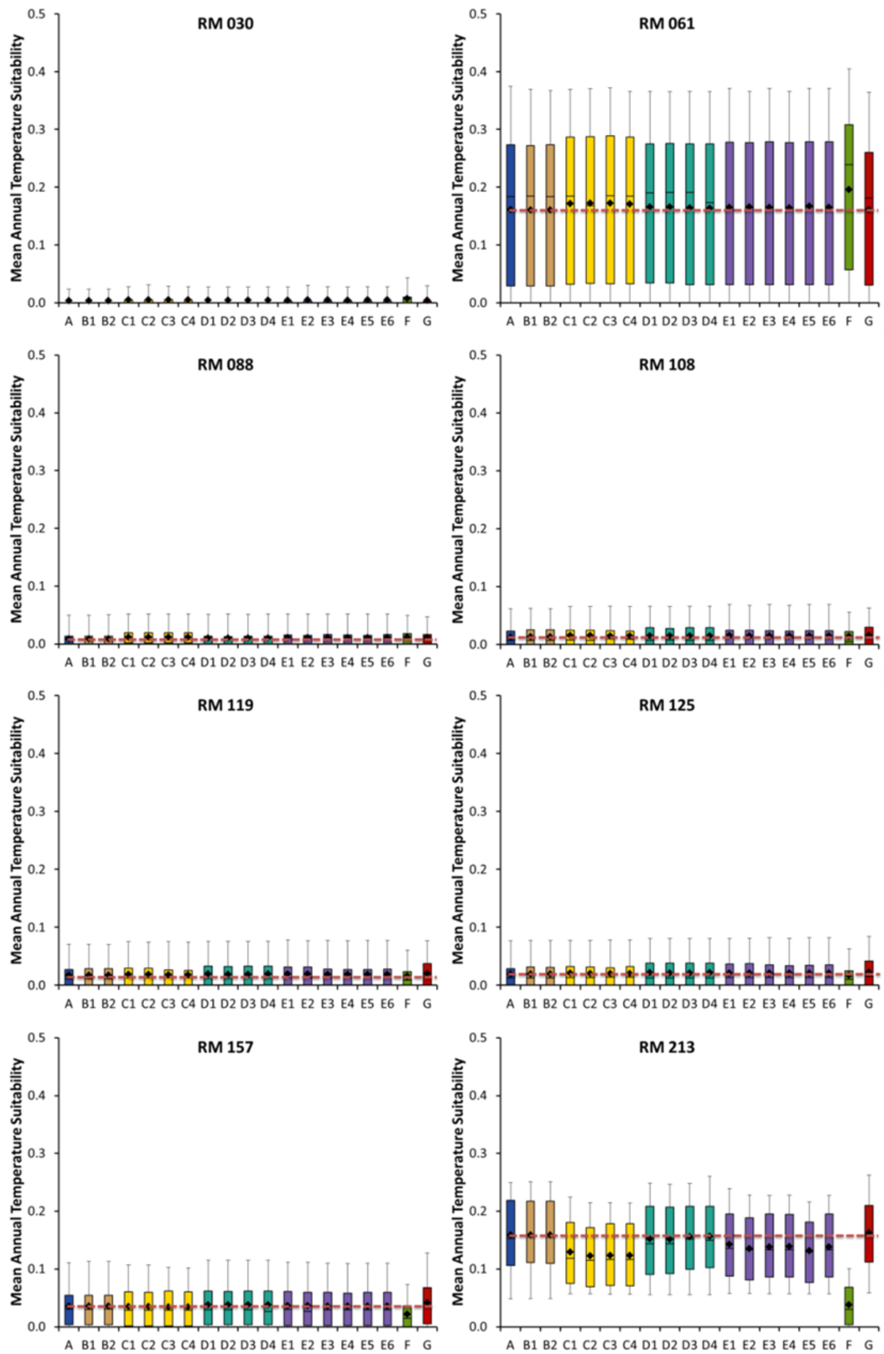

FIGURE F-23 Mainstem Temperature Suitability for Humpback Chub Aggregation Locations under LTEMP Alternatives and Long-Term Strategies (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

TABLE F-11 Locations Used for Temperature Suitability Modeling of Native Fish, Nonnative Fish, and Parasites

Aggregation Location	River Mile (RM)[a]
Glen Canyon Dam	RM –15
Paria River/Lee Ferry	RM 0
Little Colorado River confluence	RM 61
Havasu Creek	RM 157
Diamond Creek	RM 225

[a] River mile distances are calculated as the distance downstream from the Lee Ferry Gage. Glen Canyon Dam is indicated as being at RM –15, since it is located upstream of Lee Ferry.

The calculated suitability of various water temperatures for successful spawning, egg incubation, and growth of the four native species is depicted in Figure F-24. The months encompassing the spawning, egg incubation, and growth periods for each of the four native fish species are indicated in Figure F-25.These time periods were identified by reviewing the scientific literature pertaining to each of the species.

F.4.3.1 Historic Temperature Suitability for Native Fish

Historic temperature suitability of mainstem water temperatures for the four native fish species was examined using modeled water historic temperatures at five evaluation locations for a 23-year period from October 1, 1989, through September 30, 2012 (water years 1990–2012) as the temperature inputs (Figure F-26). Figure F-27 presents the annual temperature suitability scores for spawning, incubation, and growth of the four native fish species based upon the modeled historic temperatures for water years 1990–2012 at RM 225 (Diamond Creek). Figure F-28 presents the annual temperature suitability scores for the five river locations and a combined overall score for all locations based on the modeled historic temperature suitability for the various assessment locations. The overall means of annual suitability scores at each river location for native fish over the 1990–2012 water years are presented in Figure F-29.

F.4.3.2 Results for LTEMP Alternatives

The temperature suitability for the four native fish at multiple downstream locations under the LTEMP alternatives and long-term strategies is summarized in Figure F-30. Modeled main channel water temperature suitability for native fish species was relatively low and similar to Alternative A under all long-term strategies at RM 61, reflecting the prevalence of coldwater releases from Glen Canyon Dam throughout the year and the limited effect that the long-term strategies would have on mainstem water temperature regimes at RM 61. Because the water

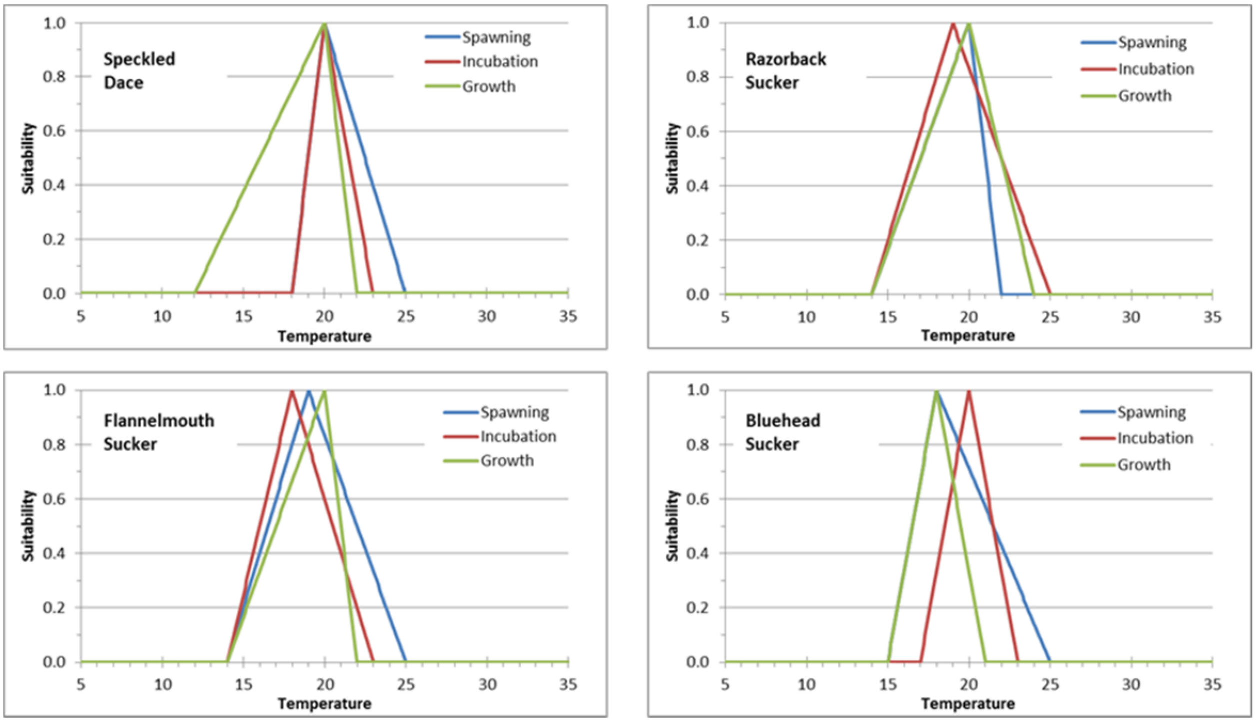

FIGURE F-24 Suitability of Water Temperatures (°C) for Spawning, Egg Incubation, and Growth of Native Fish Species (Source: Valdez and Speas 2007)

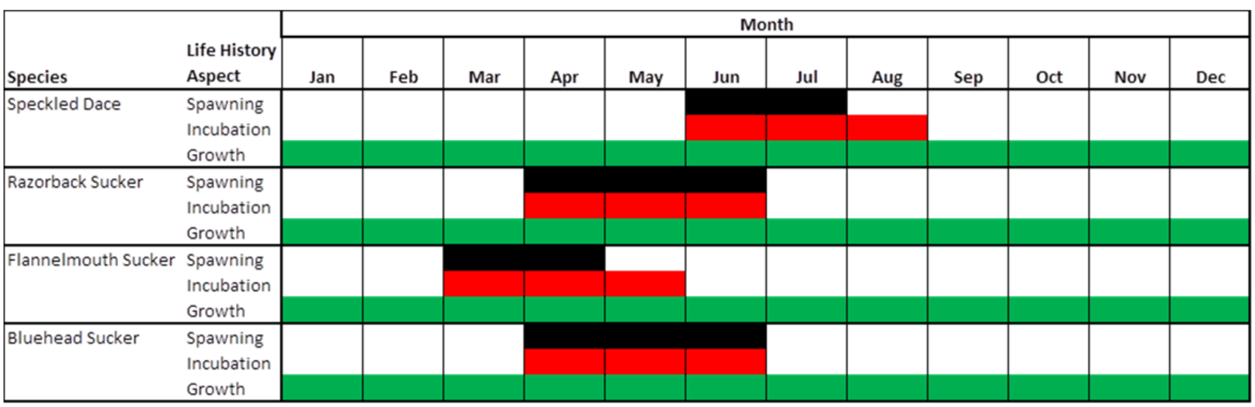

FIGURE F-25 Months for Which Temperature Suitability for Specific Life History Aspects Were Considered for Native Fish Species

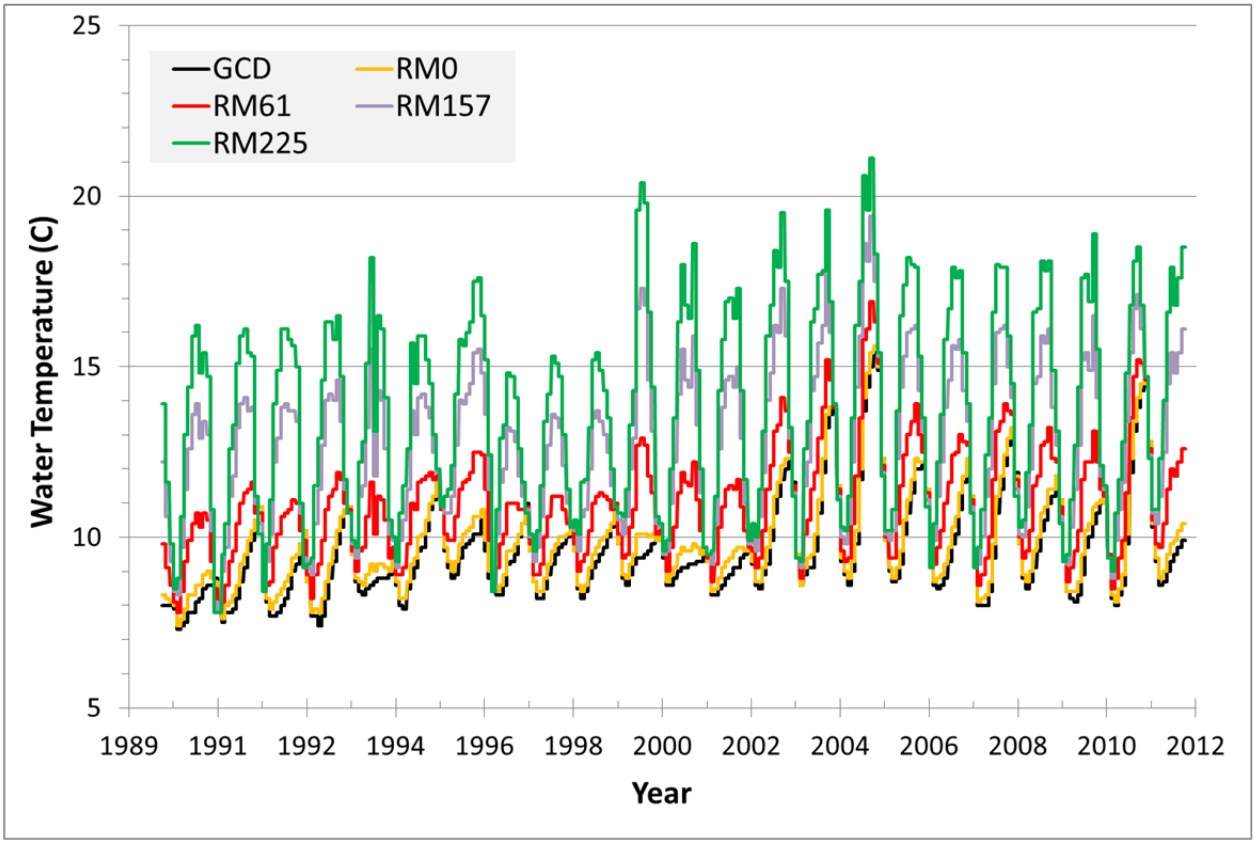

FIGURE F-26 Modeled Historic Water Temperatures in the Colorado River Downstream of Glen Canyon Dam, Water Years 1990–2012 (Source: Williams 2013)

warms as it travels downstream from the dam (for spring through fall months), temperature suitability improves with increasing downstream distance, and differences in suitability among the long-term strategies begin to appear. Whereas suitability for most long-term strategies remain similar to, or lower than, the modeled suitability under Alternative A at these downstream locations, temperature suitability for native fish improves somewhat under long-term strategies D1, D2, D3, and D4 (Figure F-30). It should be noted that there is little difference in temperature suitability among the long-term strategies specific to Alternatives B, C, D, and E, suggesting that experimental elements identified in Table 4.1-1 such as HFEs, low summer flows, TMFs, and hydropower improvement flows would have little effect on mainstem water temperature regimes during periods of the year considered most important for spawning and egg incubation by native species. Rather, differences in temperature suitability for native fish under the various long-term strategies appear to be more related to differences in the seasonal patterns of releases and the effects of those patterns on seasonal temperatures. Thus, the reduction in modeled temperature suitability under Alternative F at RM 225 reflects the higher flows expected to occur under this alternative during spring and early summer months when native fish are expected to spawn; those higher flows would result in temperatures less suitable for spawning and egg incubation.

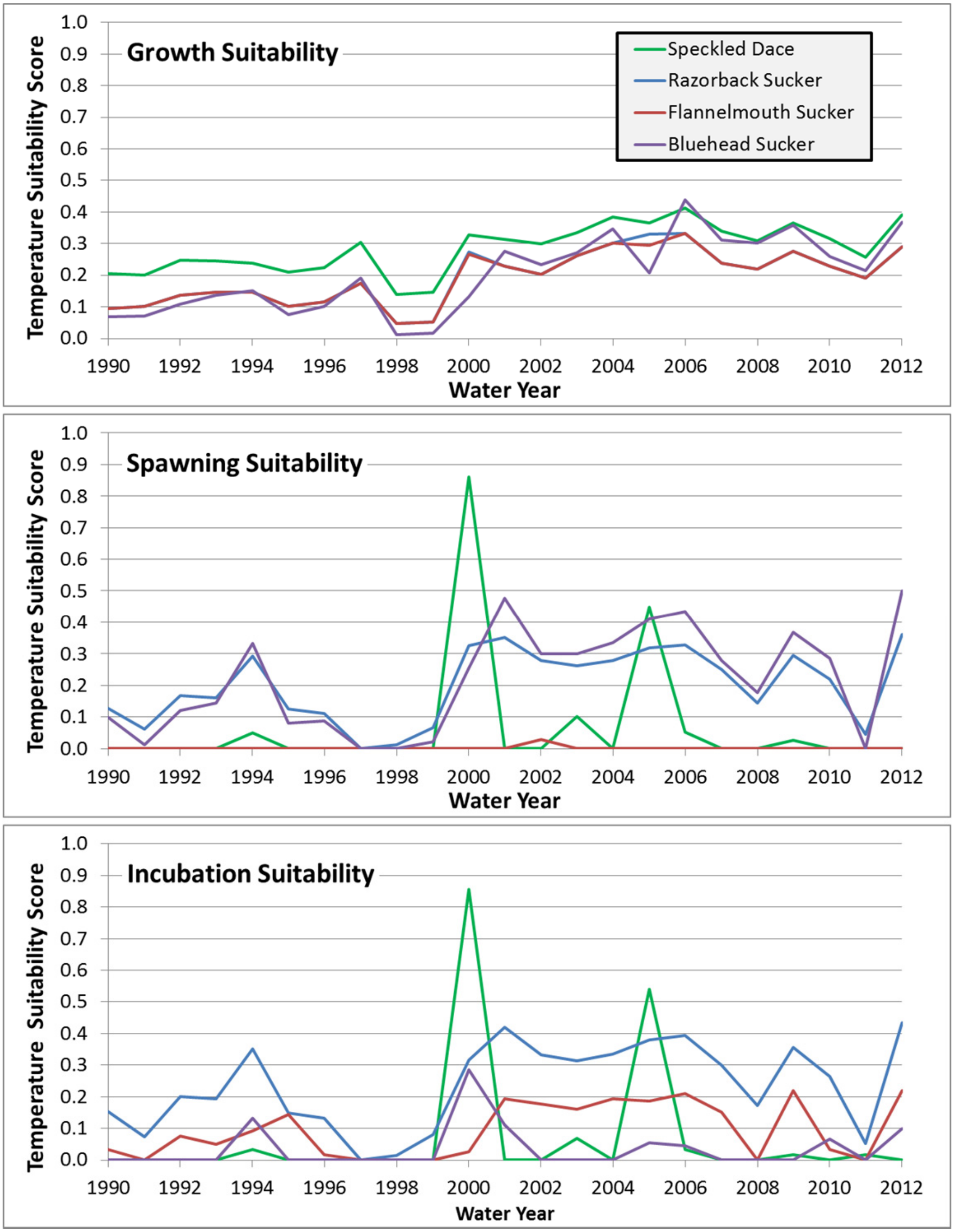

FIGURE F-27 Annual Temperature Suitability Scores for Growth, Spawning, and Egg Incubation of Native Fish Species at RM 225 Based on Modeled Water Temperatures for Water Years 1990–2012

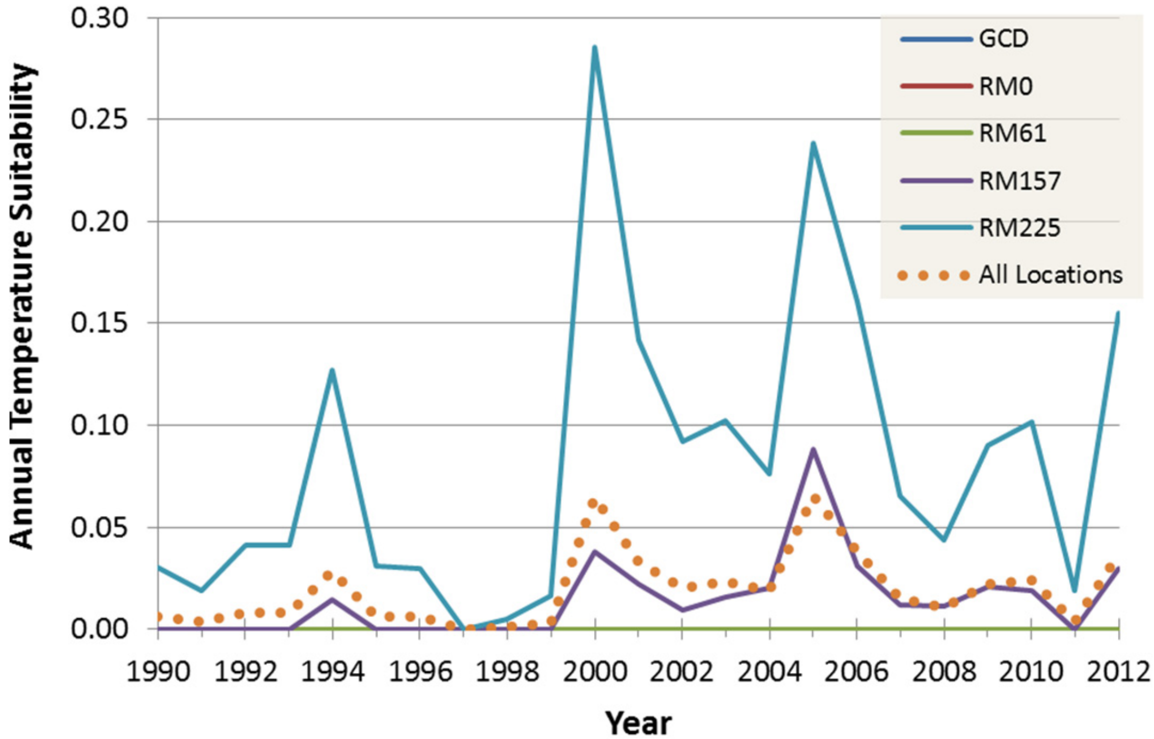

FIGURE F-28 Annual Temperature Suitability Scores for Native Fish by Assessment Location Based on Modeled Water Temperatures for Water Years 1990–2012

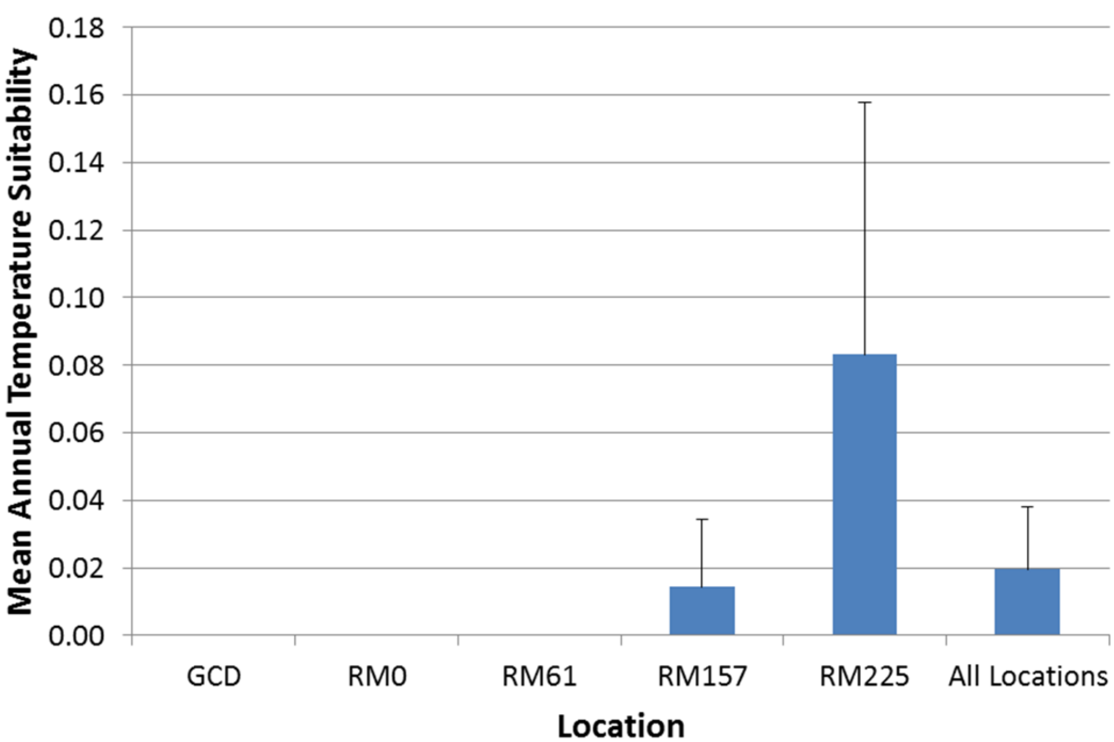

FIGURE F-29 Mean (±SD) Annual Overall Temperature Suitability for Native Fish by Assessment Location Based on Modeled Water Temperatures for Water Years 1990–2012

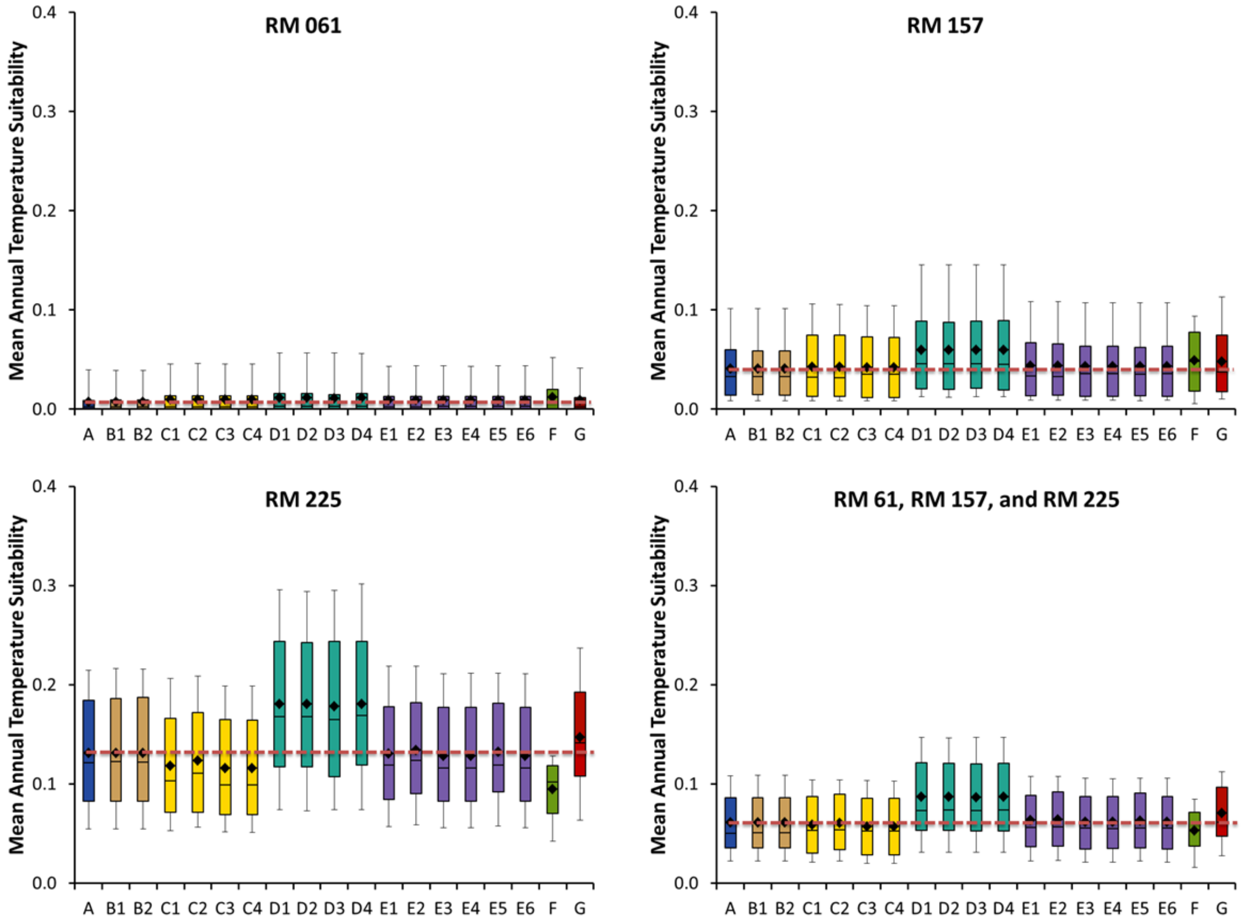

FIGURE F-30 Mean Annual Mainstem Temperature Suitability for Native Fish under LTEMP Alternatives and Long-Term Strategies at RM 61, RM 157, and RM 225, and Overall Mean for RM 61–225 (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

F.4.4 Nonnative Fish

The temperature suitability model for nonnative fish evaluates how well alternatives provide mainstem water temperatures suitable for spawning, egg incubation, and growth of six species of coldwater (brown trout [*Salmo trutta*], rainbow trout) and warmwater (channel catfish [*Ictalurus punctatus*], green sunfish [*Lepomis cyanellus*], smallmouth bass [*Micropterus dolomieu*], and striped bass [*Morone saxatilis*]) nonnative fish. In order to account for changes in water temperatures as water released from Glen Canyon Dam travels downstream, evaluations of temperature suitability were conducted for the five mainstem Colorado River locations identified in Table F-11. As described in Section F.4.1, the time series of water temperatures used as inputs for these locations were generated using a water temperature model developed by Wright et al.

(2009). The calculated suitability values for various water temperatures for successful spawning, egg incubation, and growth of the six nonnative fish species are depicted in Figure F-31. The months encompassing the spawning and egg incubation periods for each of the six nonnative fish species are indicated in Figure F-32. The annual suitability of daily temperatures for growth is calculated as the mean of daily suitability values for the entire water year (October through September). The overall means of temperature suitability values for the coldwater and warmwater nonnative species groups were examined separately.

F.4.4.1 Historic Temperature Suitability for Nonnative Fish

Historic suitability of mainstem water temperatures for the six nonnative fish species was examined using modeled water historic temperatures at the five evaluation locations for a 23-year period from October 1, 1989, through September 30, 2012 (water years 1990–2012), as the temperature inputs (Figure F-26). Figure F-33 presents the annual temperature suitability scores for spawning, incubation, and growth of the six nonnative fish species based upon the modeled historic temperatures for water years 1990–2012 at RM 225 (Diamond Creek). The mean annual temperature suitability scores for each species and temperature group for the five river locations are presented in Figure F-34. The overall means of annual suitability scores for the coldwater and warmwater nonnative fish species across all river locations during the 1990–2012 water years are presented in Figure F-35.

F.4.4.2 Results for LTEMP Alternatives

In general, temperature suitability for coldwater nonnative species (i.e., brown and rainbow trout) would be similar among most of the long-term strategies at most locations downstream of Glen Canyon Dam and would be remain similar to current conditions based on comparisons to Alternative A (Figure F-36). Because of the effects of the timing and magnitude of peak and base flow releases on water temperatures, temperature suitability would be slightly greater under Alternative F than under the other long-term strategies at the confluence with the Little Colorado River (RM 61), and lower under Alternative F than under the other long-term strategies for locations farther downstream; however, those differences are very small and may not be biologically significant. Although main channel temperature regimes at and downstream of RM 61 appear to become more suitable for trout species than at locations closer to the dam (Figure F-36), the abundance of trout is known to be lower at those locations (based on sampling), suggesting that other habitat characteristics (e.g., substrate composition and water clarity) may be less suitable at these downstream locations. Because inclusion of flow actions—such as HFEs, TMFs, and low summer flows—had only minor influences on modeled monthly mainstem water temperatures during periods of the year considered most important for spawning and egg incubation by trout, these flow actions have little effect on modeled mainstem temperature suitability and would not alter relative suitability for coldwater nonnative species among the long-term strategies (Figure F-36).

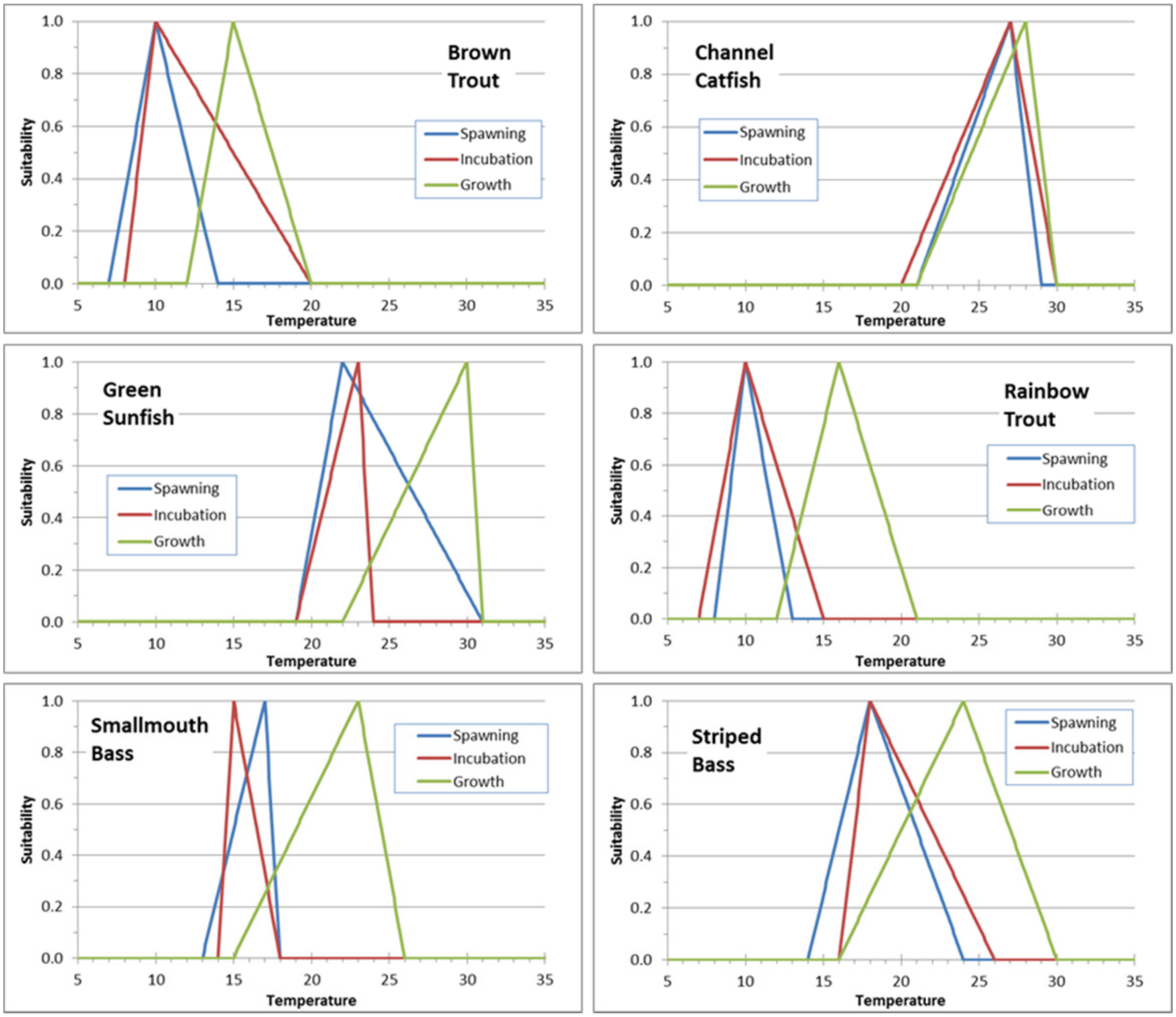

FIGURE F-31 Suitability of Water Temperatures (°C) for Spawning, Egg Incubation, and Growth of Nonnative Fish Species (Source: Valdez and Speas 2007)

Temperature suitability at the various main channel locations was modeled for the four nonnative warmwater species considered to be representative of the warmwater nonnative fish community (smallmouth bass, green sunfish, channel catfish, and striped bass). In general, the estimated average main-channel temperature suitability for these nonnative fish did not differ greatly among the long-term strategies, and was low under all long-term strategies (Figure F-37). The modeled temperature suitability indicated that temperature conditions would be most suitable for warmwater nonnative species at locations farther downstream from Glen Canyon Dam (e.g., RM 157 and RM 225) compared to upstream locations (e.g., RM 0 and RM 61); this agrees with past surveys that have found more warmwater nonnative fish species in those areas. Relative to current conditions (as exemplified by Alternative A), the temperature suitability model indicated that the long-term strategies for Alternative C (i.e., long-term strategies C1, C2, C3, and C4) and Alternative F have the greatest potential to improve conditions for warmwater nonnative fish at locations downstream of RM 157, which could result in increased numbers

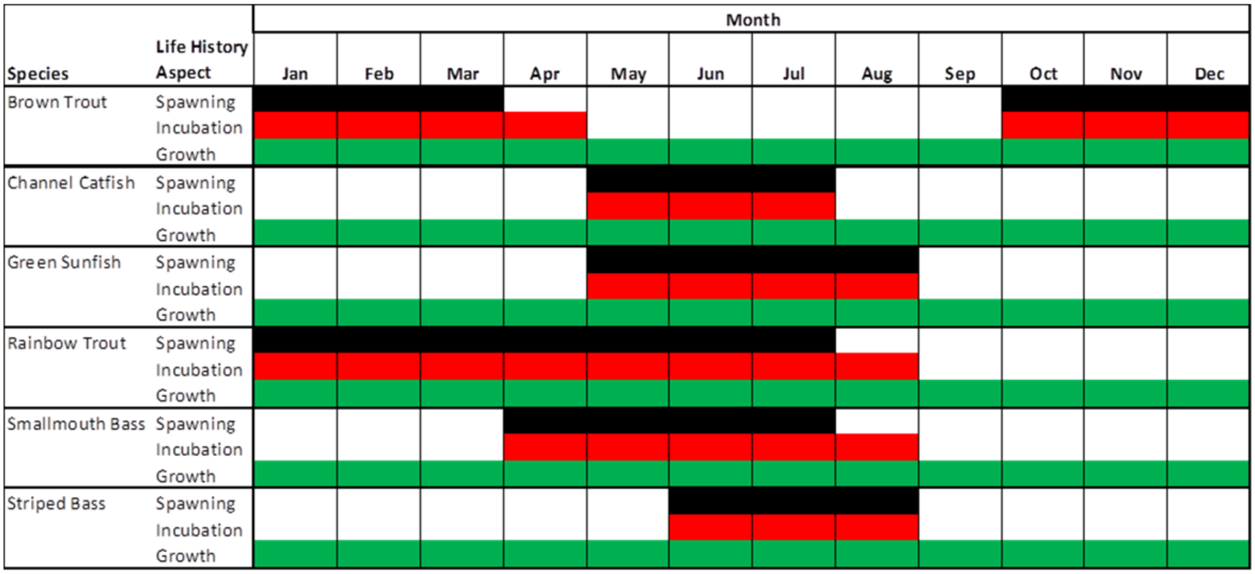

FIGURE F-32 Months during Which Temperature Suitability for Specific Life History Aspects Were Calculated for Nonnative Fish Species

and a greater potential for upstream spread of warmwater nonnative fish species. As described above for coldwater fish species, inclusion of flow actions such as HFEs, TMFs, and low summer flows had only minor influences on modeled monthly mainstem water temperatures during periods of the year considered most important for spawning and egg incubation by nonnative warmwater species. As a consequence, the various experimental elements associated with the long-term strategies (Table 4.1-1) would be expected to have little effect on mainstem temperature suitability for warmwater nonnative species (Figure F-37). Rather, as identified for native fish in Section F.4.3.2, differences among alternatives appear to be more related to differences in the seasonal patterns of releases and the effects of those patterns on seasonal temperatures.

F.4.5 Aquatic Parasites

The temperature suitability model for aquatic parasite species evaluates how well alternatives provide mainstem water temperatures suitable for host activity for and infestation by four species (Asian tapeworm [*Bothriocephalus acheilognathi*], anchor worm [*Lernaea cyprinacea*], trout nematode [*Truttaedacnitis truttae*], and whirling disease [*Myxobolus cerebralis*]) that could parasitize fish in the Colorado River downstream of Glen Canyon Dam. In order to account for changes in water temperatures as water released from Glen Canyon Dam travels downstream, evaluations of temperature suitability were conducted for the mainstem Colorado River locations identified in Table F-11. As described in Section F.4.1, the time series of water temperatures used as inputs for these locations were generated using a water temperature model developed by Wright et al. (2009).

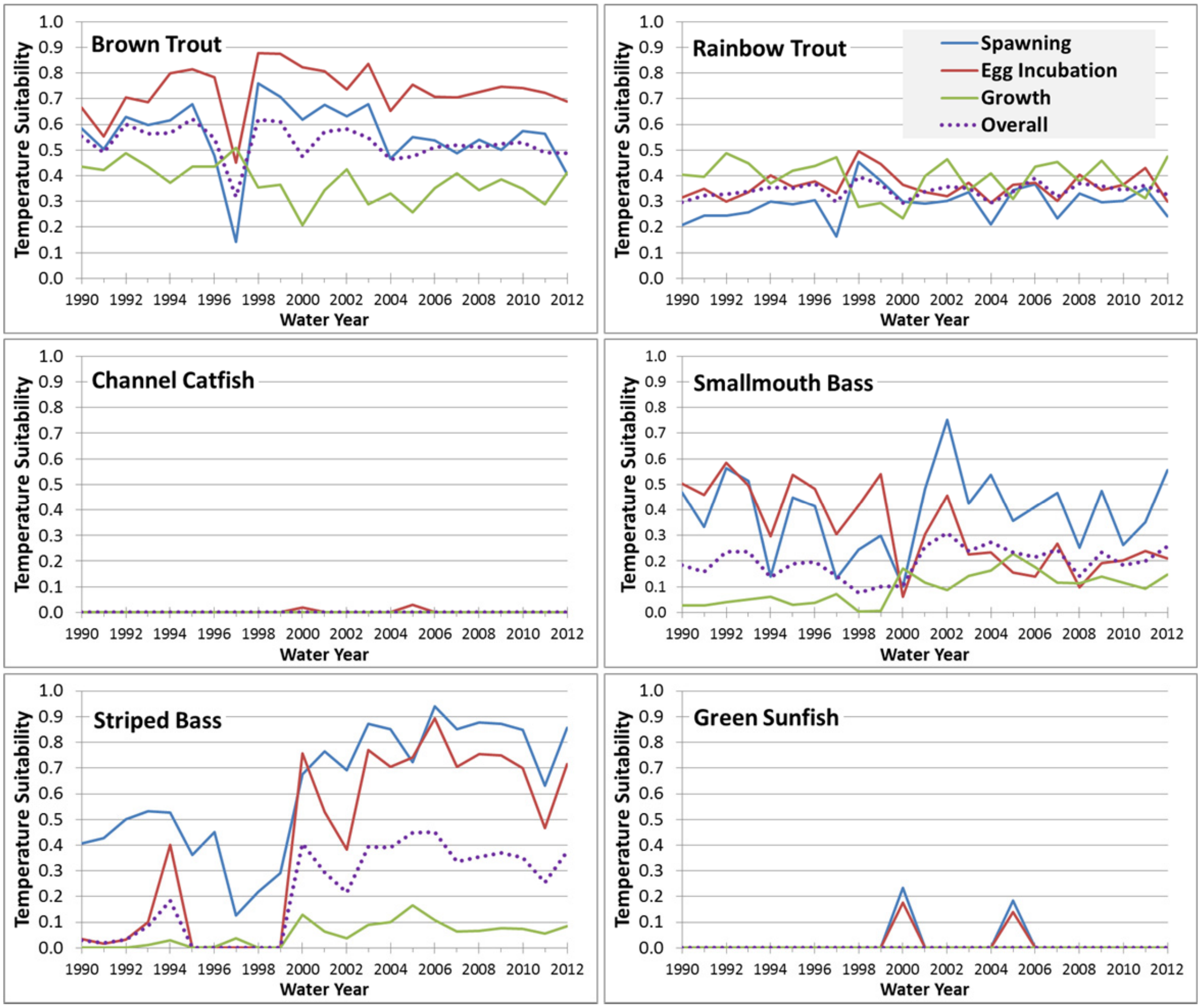

FIGURE F-33 Annual Temperature Suitability Scores for Spawning, Incubation, and Growth of Nonnative Fish Species at RM 225 (Diamond Creek) Based on Modeled Temperatures for Water Years 1990 to 2012

The calculated suitability values at various water temperatures for host activity and infestation rates of the four parasite species is depicted in Figure F-38. It was assumed that evaluation of temperature suitability across the entire water year (rather than just a portion of the year) was relevant for both of the parasite life history components. The geometric mean of the annual temperature suitability values for host activity and infestation was used as an indicator of the annual overall suitability for each parasite species and served as the indicator of the potential for each of the parasite species to become problematic at a particular downstream location. The combined mean of the annual suitability scores for all four parasite species was used as an indication of the overall suitability of each year's temperature regime for the group of parasite species at each downstream location. The mean of the group means for all of the downstream locations was calculated as an indication of overall relative suitability of the temperature regime

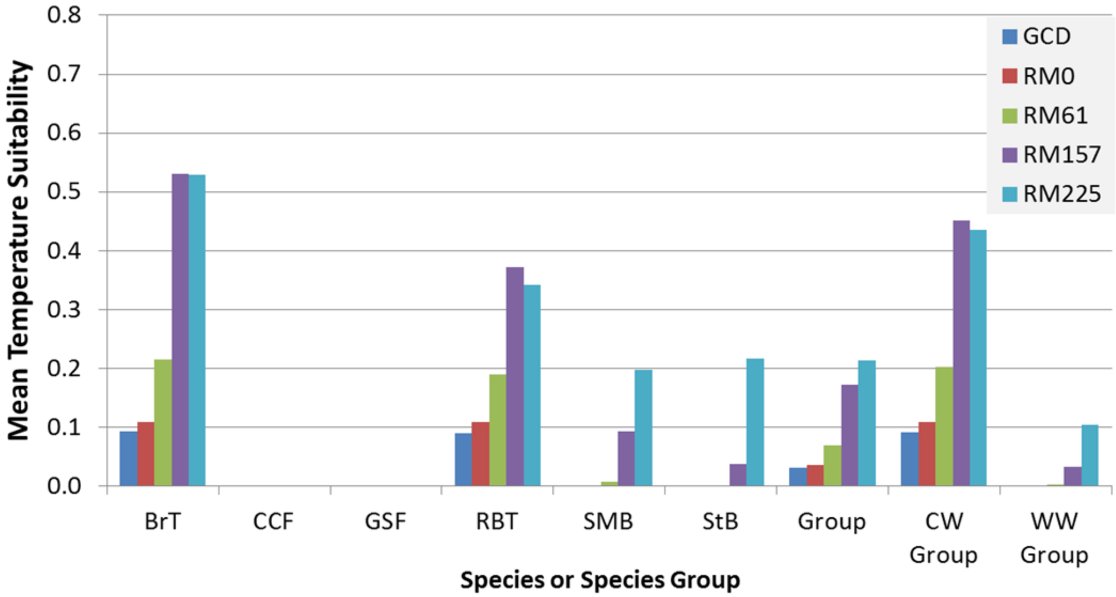

FIGURE F-34 Mean Annual Temperature Suitability Scores for Nonnative Fish Species and for Temperature Groups by River Location Based on Modeled Water Temperatures for Water Years 1990–2012 (BrT = brown trout; CCF = channel catfish; GSF = green sunfish; RBT = rainbow trout; SMB = smallmouth bass; StB = striped bass; Group = combined coldwater and warmwater; CW = coldwater; WW = warmwater)

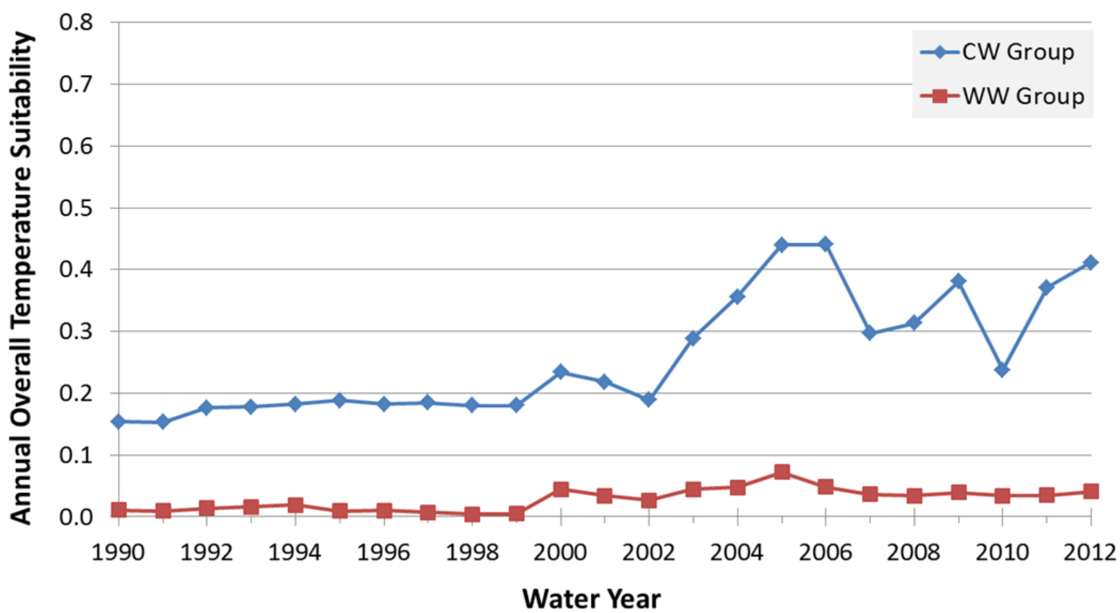

FIGURE F-35 Mean Annual Overall Temperature Suitability Scores for Coldwater (CW) and Warmwater (WW) Nonnative Fish Species Groups Based on Modeled Historic Temperatures for Water Years 1990–2012

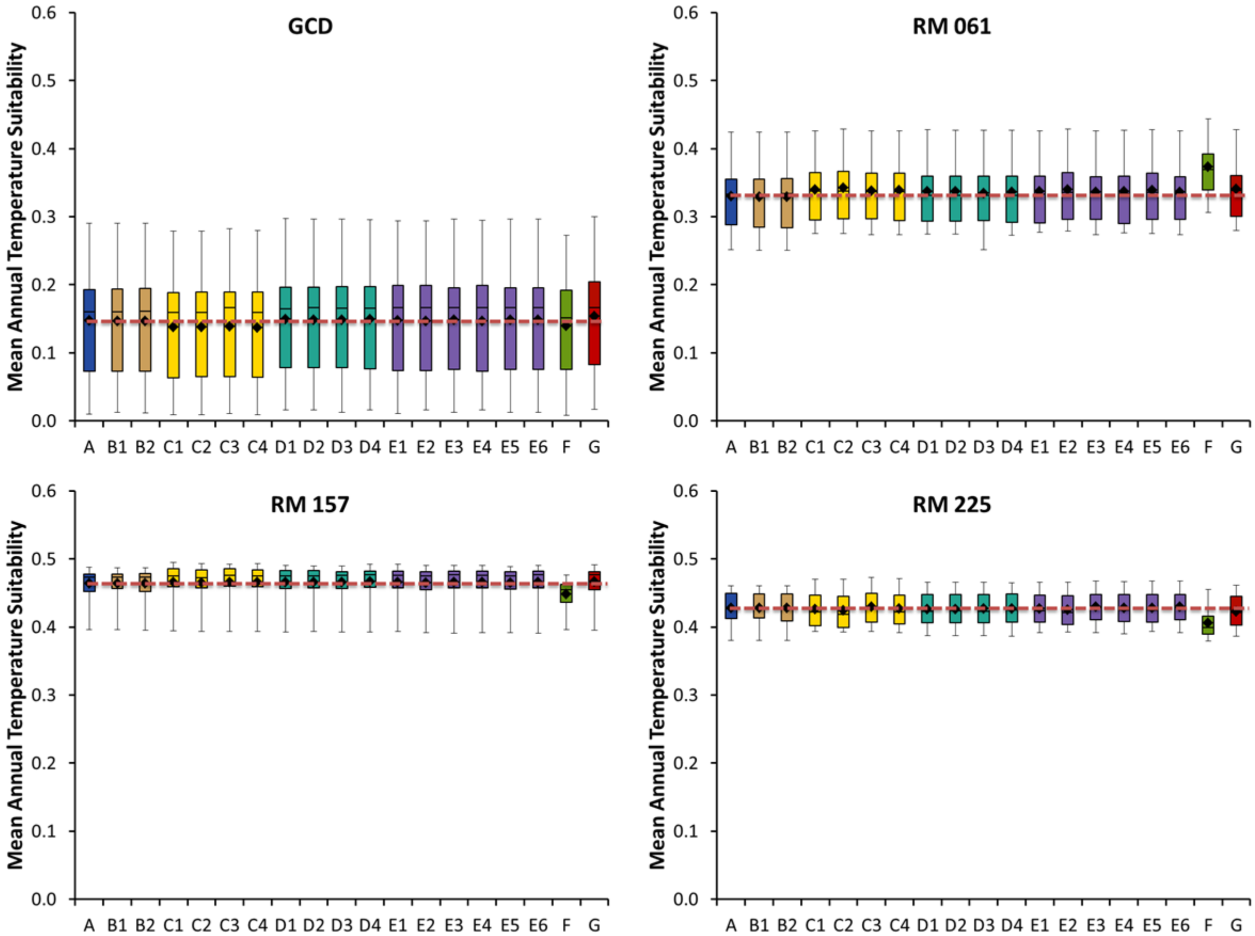

FIGURE F-36 Mean Annual Mainstem Temperature Suitability for Coldwater Nonnative Fish (brown trout and rainbow trout) under LTEMP Alternatives and Long-Term Strategies at RM 15 (Glen Canyon Dam, GCD), RM 61, RM 157, and RM 225 (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

within a particular year for parasite species in the mainstem Colorado River downstream of Glen Canyon Dam.

F.4.5.1 Historic Temperature Suitability for Aquatic Parasites

Historic suitability of mainstem water temperatures for the four aquatic parasite species was examined using modeled water historic temperatures at the five evaluation locations for a 23-year period from October 1, 1989, through September 30, 2012 (water years 1990–2012), as the temperature inputs (Figure F-26). Figure F-39 presents the annual temperature suitability

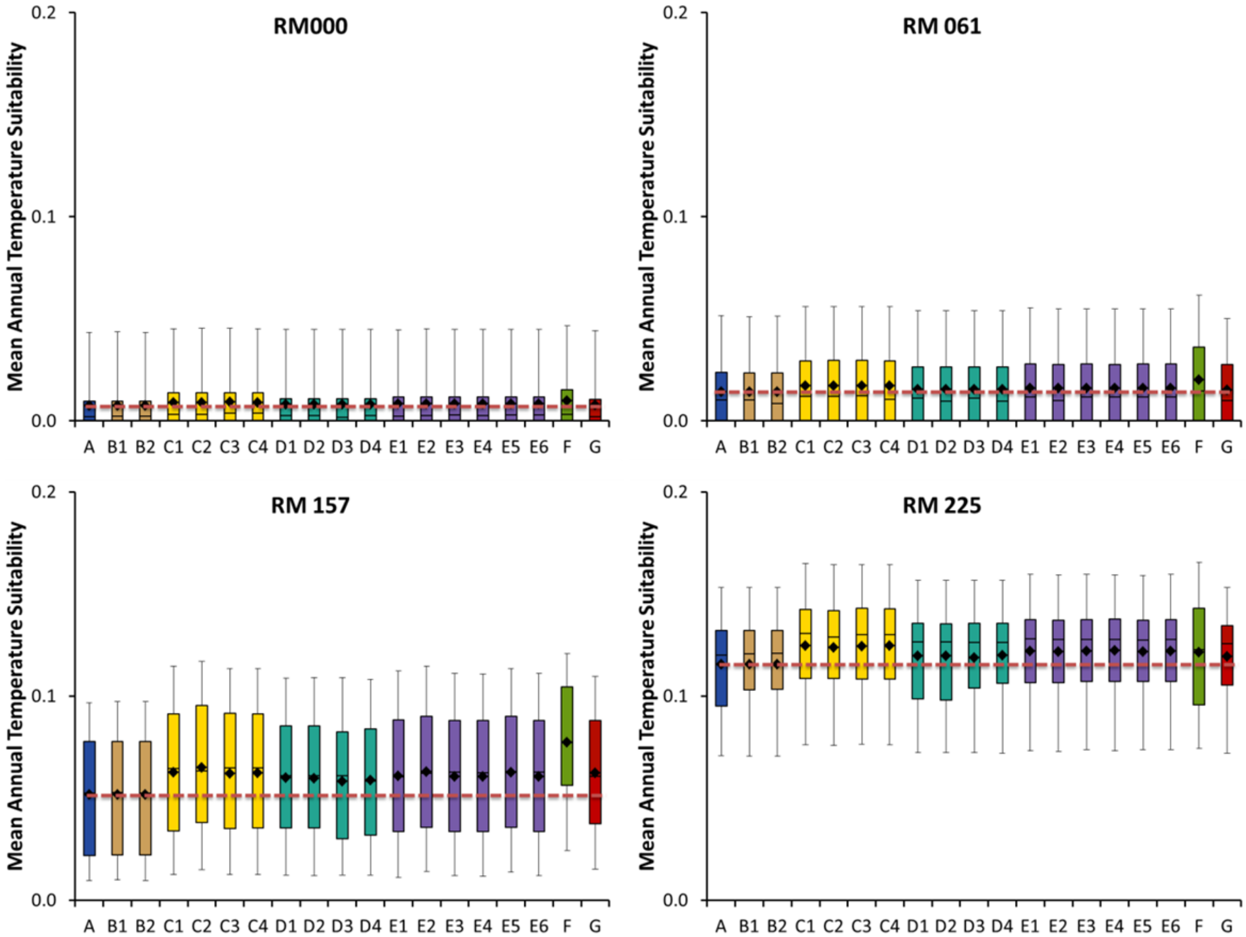

FIGURE F-37 Mean Annual Mainstem Temperature Suitability for Warmwater Nonnative Fish (channel catfish, green sunfish, smallmouth bass, and striped bass) under LTEMP Alternatives and Long-Term Strategies at RM 0, RM 61, RM 157, and RM 225 (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

scores for host activity and infestation rates of the parasite species based upon the modeled historic temperatures for water years 1990–2012 at RM 225 (Diamond Creek). The mean annual temperature suitability scores for each species and temperature group for the five river locations are presented in Figure F-40. The overall means of modeled annual suitability scores for the coldwater and warmwater nonnative fish species groups across all river locations during the 1990–2012 water years are presented in Figure F-41.

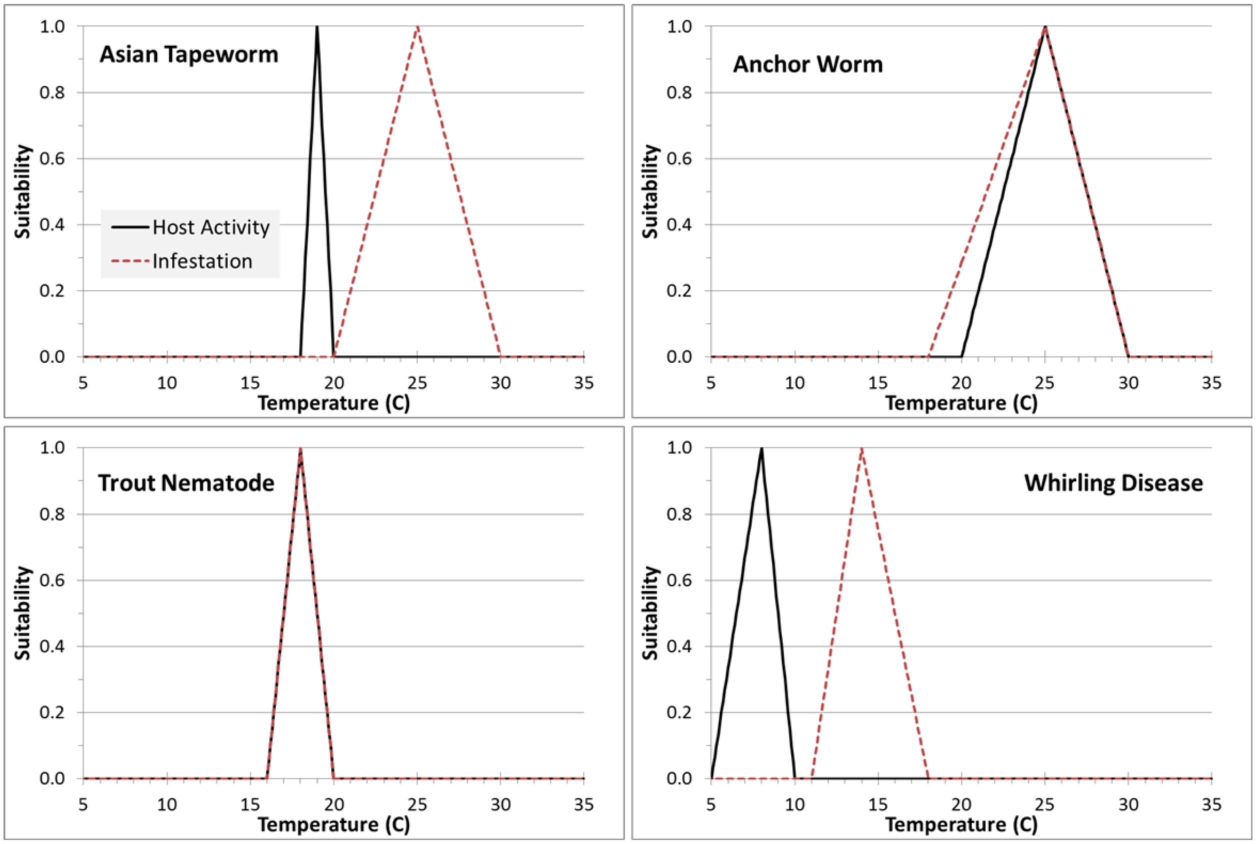

FIGURE F-38 Suitability of Various Water Temperatures for Host Activity and Infestation Rates of Parasite Species (Source: Valdez and Speas 2007)

F.4.5.2 Results for LTEMP Alternatives

Temperature suitability for the four aquatic parasite species (Asian tapeworm, anchor worm, trout nematode, and whirling disease) under the LTEMP alternatives and long-term strategies was modeled for various locations downstream from Glen Canyon Dam. Modeling indicated that temperature suitability for the aquatic parasite species would generally be very low under all long-term strategies and would be comparable to the suitability under current operations as represented by Alternative A (no-action alternative; Figure F-42). As a consequence, the relative distributions of aquatic parasites or the effects of aquatic parasites on survival and growth of native fish or trout species would not be expected to change relative to current conditions under any of the long-term strategies. Under current conditions, population-level effects of parasites on survival and growth of native fish or trout have not been observed. Inclusion of flow actions such as HFEs, TMFs, and low summer flows had only minor influences on modeled monthly mainstem water temperatures during periods of the year considered most important for spawning and egg incubation by native fish. As a consequence, these flow actions are expected to have minor effects on temperature suitability for the parasite species group and would not alter the relative suitability among the long-term strategies.

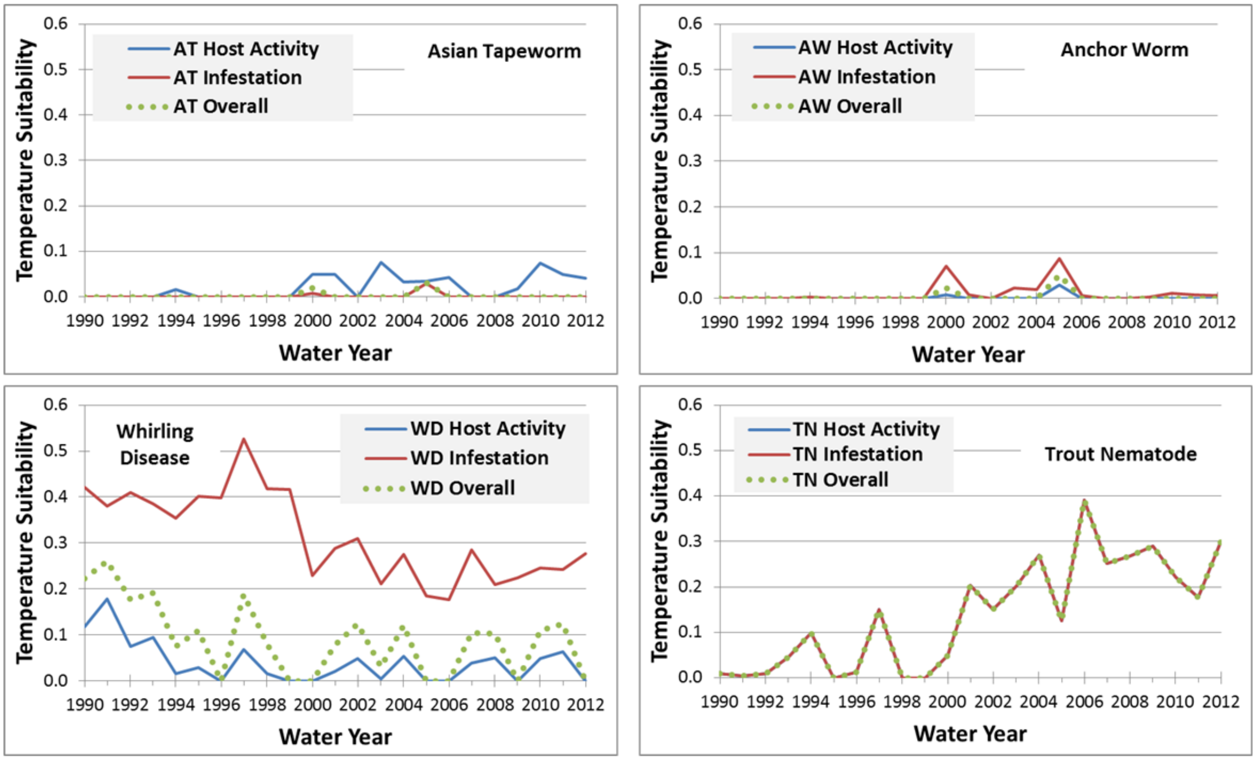

FIGURE F-39 Annual Temperature Suitability Scores for Parasite Species at RM 225 (Diamond Creek) Based on Modeled Water Temperatures for Water Years 1990–2012 (AT = Asian tapeworm; AW = anchor worm; TN = trout nematode; and WD = whirling disease)

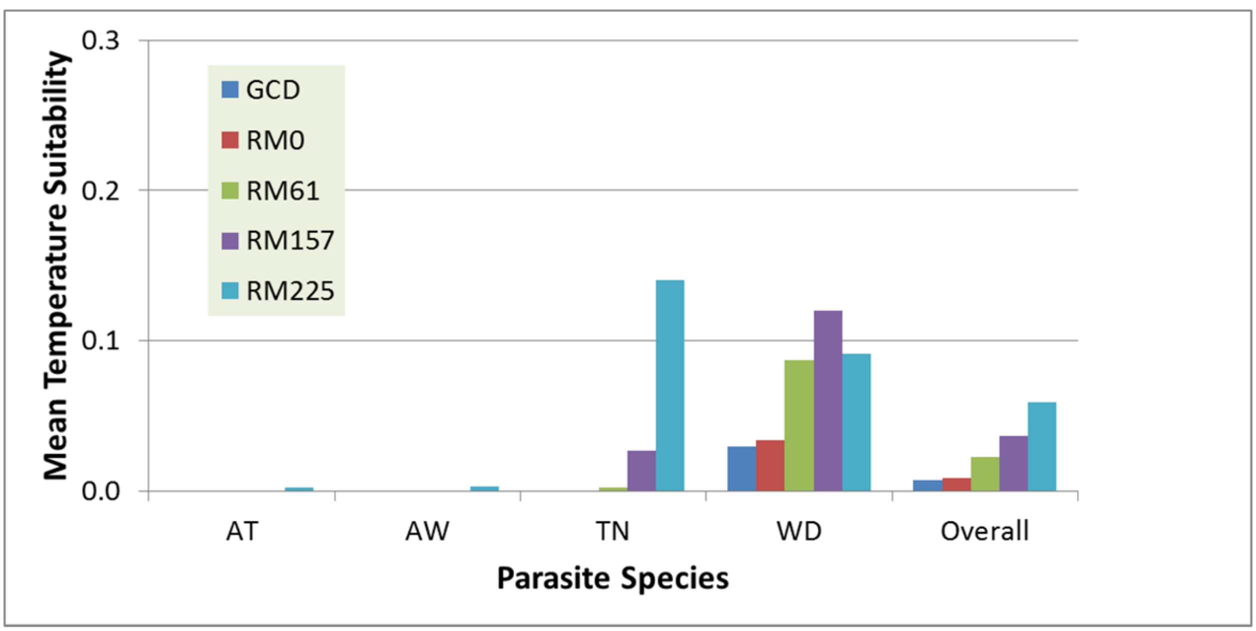

FIGURE F-40 Mean Annual Temperature Suitability Scores for Parasite Species by River Location Based on Modeled Water Temperatures for Water Years 1990–2012 (AT = Asian tapeworm; AW = anchor worm; TN = trout nematode; and WD = whirling disease)

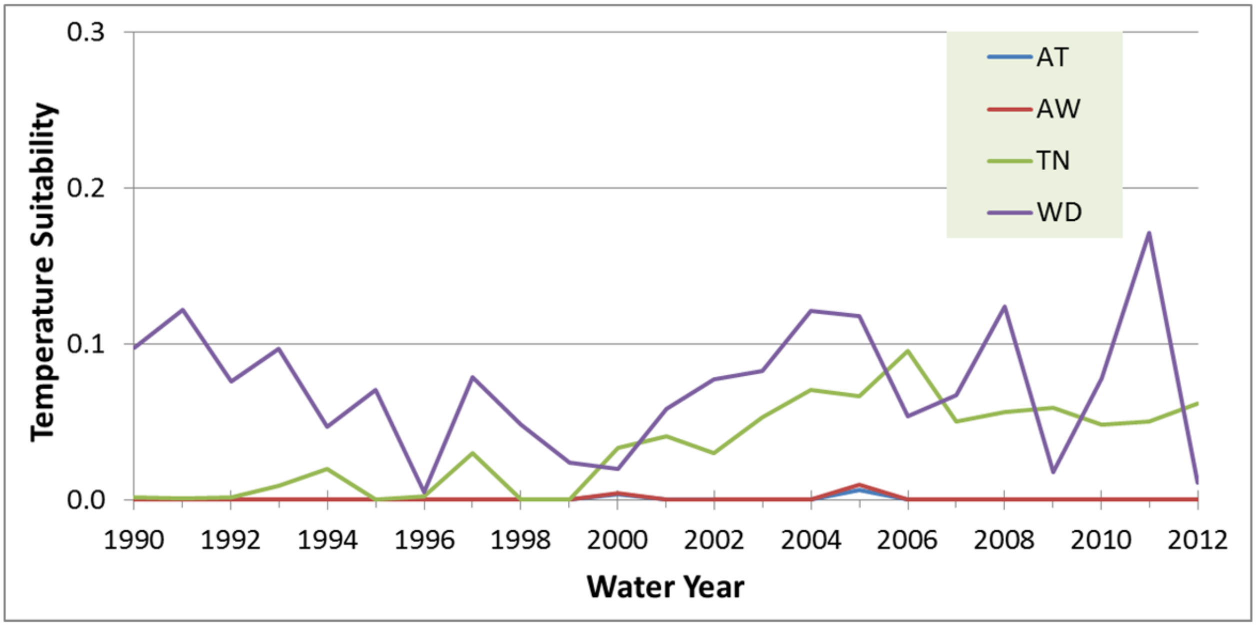

FIGURE F-41 Overall Means of Annual Suitability Scores for Parasite Species across All River Locations during the 1990–2012 Water Years (AT = Asian tapeworm; AW = anchor worm; TN = trout nematode; and WD = whirling disease)

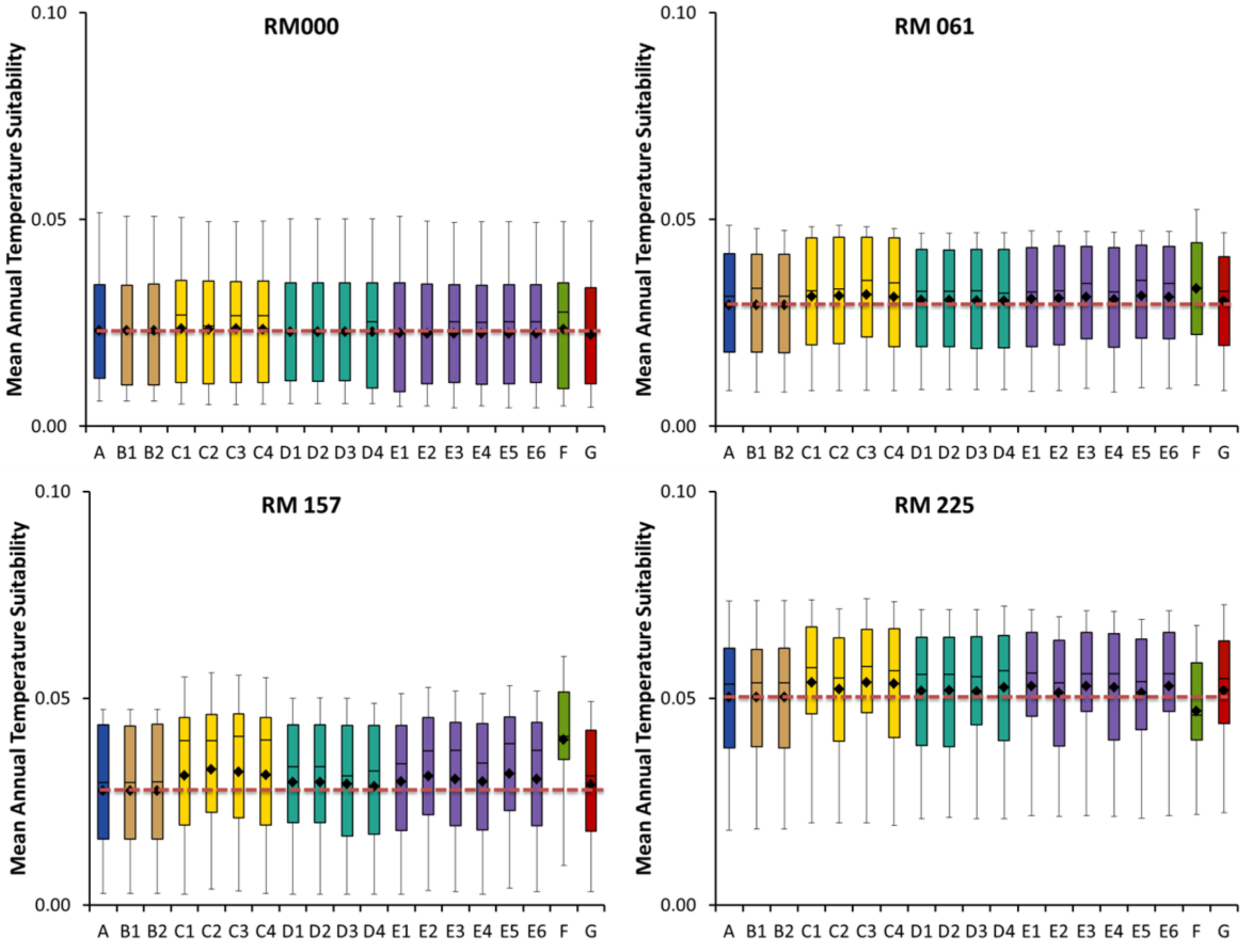

FIGURE F-42 Overall Modeled Mean Annual Temperature Suitability under LTEMP Alternatives and Long-Term Strategies for Aquatic Fish Parasites (Asian tapeworm, anchor worm, trout nematode, and whirling disease) at Four Locations Downstream of Glen Canyon Dam (The graph shows the mean, median, 75th percentile, 25th percentile, minimum, and maximum values for 21 hydrology scenarios and three sediment scenarios. Means were calculated as the average for all years within each of the 21 hydrology runs. Note that diamond = mean; horizontal line = median; lower extent of box = 25th percentile; upper extent of box = 75th percentile; lower whisker = minimum; upper whisker = maximum; horizontal dashed line identifies mean value for Alternative A.)

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