Los Angeles Department of Water & Power
BUILDING A STRONGER L.A.

May 12, 2023

Mr. Erik Ekdahl
Deputy Director
Division of Water Rights
State Water Resources Control Board
1001 I Street, 14th Floor
Sacramento, California 95814

Dear Mr. Ekdahl:

Subject: Annual Monitoring Reports for Runoff Year 2022-23

Pursuant to the Los Angeles Department of Water and Power (LADWP) Mono Basin Water Rights Amended License Nos. 10191 and 10192 (Licenses), please find the enclosed Annual Monitoring Reports, which includes the following reports required by the Licenses:

- Section 1: Mono Basin Fisheries Monitoring Report Rush, Lee Vining, and Walker Creeks 2022
- Section 2: Mono Basin Waterfowl Habitat Restoration Program 2022 Monitoring Report

Note that a Stream Monitoring Report is not included this year. An update on limnology monitoring and results is included in the Waterfowl Monitoring Report.

The submission of these reports fulfills the requirements of Section 20.g(2) and Section 22.c of the Licenses.

If you have any questions, please contact Mr. Mark Y. Ching, Civil Engineering Associate, at (213) 367-2132.

Sincerely,

Adam Perez
Manager of Aqueduct

MYC:jm
Enclosures
c/enc: Mr. Mark Y. Ching

Mono Basin
Annual Monitoring Reports

May 2023

Prepared for State Water Resources Control Board

In Accordance with the Terms and Conditions of Amended License Nos. 10191 and 10192

<u>Contents</u>

Section I. Mono Basin Fisheries Monitoring Report Rush, Lee Vining, and Walker Creeks 2022

Prepared by Ross Taylor and Associates

Mono Basin Fisheries Monitoring Report
Rush, Lee Vining, and Walker Creeks
2022

Prepared by Ross Taylor and Associates for

The State Water Resources Control Board, the Los Angeles Department of Water and Power
and the Mono Basin Monitoring Administration Team

May 8, 2023

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Executive Summary

The 2022 fisheries sampling was the first of ten years of biological monitoring of the Stream Ecosystem Flows (SEF), with oversight from the newly formed Mono Basin Monitoring Administration Team (MAT) as directed by the California State Water Resource Control Board's (SWRCB) amended Licenses 10191 and 10192. This monitoring continues a 25-year history of monitoring ordered by the SWRCB under Orders 98-05 and 98-07. The six reaches sampled in 2022 were similar in length to those sampled between 2009 and 2021, except the Lee Vining Creek side channel was dropped from the sample sites. Sample site selection has evolved over time, with more sites annually sampled in 1999 through 2008 as the Fisheries Team was investigating potential differences in fish production based on proximity to the Grant Lake Reservoir (GLR) dam and varying channel slopes and confinement. For example, in Rush Creek, the Mono Gate One Return Ditch (MGORD) was selected because of its tailwater condition below the dam and its propensity to support older and larger Brown Trout. Upper Rush was selected for its moderately-sloped channel and its location just downstream of a confined, gorge-like section. Finally, the Bottomlands section was selected for its location in a low-gradient area that historically had more potential for the formation of meanders, deeply scoured pools, and side-channels.

The 2022 Runoff Year (RY) was 60% of normal and classified as a Dry RY type, as measured on April 1st. The range of runoff that defines a Dry RY is ≤ 68.5% (80% - 100% exceedance). The preceding 10 years included a Dry RY of 58% in 2021, Dry-Normal-1 RY of 71% in 2020, a Wet RY of 140% in 2019, a Normal RY of 85% in 2018, a record Extreme-wet RY of 206% in RY 2017 and five consecutive below "Normal" RY's (RY 2016 was 74% of normal, RY 2015 was 25% of normal, RY 2014 was 48% of normal, RY 2013 was 66% of normal and RY 2012 was 55% of normal).

Two-pass electrofishing for generating mark-recapture population estimates was conducted in the Lee Vining Creek main channel section and in three sections of Rush Creek – the MGORD, Upper Rush and the Bottomlands. Multiple-pass depletion electrofishing was conducted in Walker Creek.

Annual Monitoring Metrics

As in previous years, the annual monitoring metrics included population estimates of three size classes of trout (<125 mm, 125-199 mm, and ≥200 mm in total length), density estimates (# fish/ha) of age-0 and age-1+ trout, total standing crop estimates (kg/ha), condition factors, growth rates (in length and weight) from recaptures of previously PIT tagged trout, apparent survival of age-1 trout (from PIT tag data), and relative stock densities (RSD) for three size classes of catchable trout (Table 1).

Table 1 provides a concise view of comparisons of the 2022 monitoring metrics versus 2021 results. The Results section provides more detail regarding the percentages of the decreases and increases of all monitoring metrics (Table 1). Several of the reported increases and decreases in Table 1 may fail to make sense without the details presented in the Results section. For example, in Walker Creek, two of three size classes of Brown Trout experienced decreases, yet the density estimate of age-1+ fish increased as did the total standing crop estimate (Table 1). The surface area of the Walker Creek section decreased between 2021 and 2022; also, the increases in the density and standing crop estimates from 2021 to 2022 were relatively small.

PIT Tagging – New Tags and Recaptures

In 2022, a total of 697 trout received PIT tags and adipose fin clips in Rush, Lee Vining, and Walker Creeks. In addition, five recaptured adipose fin-clipped fish had shed their original tags and were re-tagged, thus a total of 702 PIT tags were implanted during the 2022 fisheries sampling. Of the 702 trout tagged, 610 were age-0 Brown Trout and 54 were age-1 and older Brown Trout. For Rainbow Trout, 37 age-0 fish and one older fish were tagged. Forty-nine of the age-1+ Brown Trout tagged in the MGORD section were up to 225 mm in total length and were presumed to be age-1 fish. In addition, 26 age-0 Brown Trout were tagged in the MGORD.

One-hundred-seven previously tagged trout (that retained their tags) were recaptured in the Rush Creek watershed: 23 of the recaptures occurred in the Bottomlands section, followed by 22 recaptures in the Upper Rush section (including one Rainbow Trout), 54 recaptures in Walker Creek, and eight recaptures in the MGORD. In Lee Vining Creek, 18 previously tagged Brown Trout (that retained their tags) were recaptured.

Although the growth rates of most PIT-tagged recaptured age-1 and age-2 Brown Trout increased from 2021 to 2022 (Table 1), these two Dry RYs experienced relatively low growth compared to most previous years where recapture data were available.

Summer Water Temperatures in Rush Creek

In 2022, a Dry RY with GLR storage levels either below or only 3.4 feet above the Synthesis Report recommended minimum summer storage elevation of 7,100 feet in July-September resulted in mostly unfavorable summer thermal conditions, with peak water temperatures >70°F at all six Rush Creek temperature monitoring locations. At all six of these monitoring locations, the numbers of days with water temperatures >70°F were either the highest or second-highest ever recorded at these stations. In July and August, three of the temperature monitoring locations recorded peak temperatures >75°F. In 2022, daily mean temperatures and average daily maximum temperatures were either the highest or second-highest recorded at all Rush Creek temperature monitoring locations since these data were collected.

The diurnal fluctuations throughout the summer of 2022 were relatively low at the Top of MGORD and Bottom of MGORD temperature monitoring locations, but diurnal fluctuations increased at the downstream monitoring locations, most likely due to effects of daily warming and nightly cooling of air temperatures. Over the 21-day durations, these larger diurnal fluctuations were above the threshold of 12.6°F considered detrimental to trout growth

(Werley et al. 2007) during the summer of 2022 as recorded at the Above Parker, Below Narrows, and County Road temperature monitoring locations. These same three temperature monitoring locations also had 21-day durations with diurnal fluctuations exceeding 12.6°F during the summers of 2020 and 2021.

Because solar radiation has a large effect on water temperature, 33 years of summer air temperatures in the Mono Basin are presented in the Discussion section. Broken down by decades (1990's, 2000's, 2010's and 2020's), the metrics of average daily maximum and number of days with peak temperatures ≥90°F have all increased. The average daily maximum air temperature in the 1990's equaled 80.4°F and in the first three years of the 2020's, the average maximum air temperature was 86.0°F. The number of days with peak air temperatures ≥90°F has recently experienced the biggest increase. In the first 25 years of the 33-year dataset, there were four years (1994, 2002, 2007 and 2012) where at least 10 days had maximum air temperatures ≥90°F versus in the most recent seven years (2016-2022), six of the years experienced at least 10 days with maximum air temperatures ≥90°F. The past three years have had the highest totals of days with maximum air temperatures ≥90°F (17 days in 2020, 24 days in 2021, and 22 days in 2022).

Proposed Fisheries Sampling for 2023 Season

During the development of the post-settlement monitoring scope and budget, RTA proposed that the annual fisheries sampling was reduced to conducting population estimate sampling every other year. In the other years, single-pass electrofishing sampling would occur to collect data to evaluate population age-class structure, compute condition factors, generate growth data from recaptures of previously tagged fish, and implant PIT tags in new cohorts of fish. We intend to conduct single-pass sampling in the fall of 2023. In addition to conducting single-pass sampling at the annually sampled locations, RTA proposes sampling several locations in Rush Creek in conjunction with field observations made by MLC staff of side-channel and off-channel habitats. Summer water temperatures would also be collected within these specific habitats.

Because the 2023 record snowpack in the Mono Basin and an Extremely-wet RY forecast, the 2023 fisheries sampling will occur during early October, as opposed to its usual scheduling in mid-September. Similar to 2017, this later start date for the fisheries sampling will allow for an extended receding limb in Rush Creek that releases water stored in GLR. Then LADWP has the flexibility to divert flow from Lee Vining Creek into GLR to allow for safe wading and effective electrofishing in Lee Vining Creek without causing a spill from GLR and excessively high flows (for fisheries sampling) to occur in Rush Creek.

With the record Extemely-wet RY, we expect that Rush Creek will experience elevated, yet cooler, flows during the summer of 2023; which should translate into higher growth rates and better condition factors in Brown Trout.

Table 1. Summary of Mono Basin Brown Trout annual monitoring metrics; changes between sampling years 2021 and 2022. N/A = not applicable or not available. For applicable metrics, the percent increases/decreases between 2021 and 2022 are provided in parentheses. For growth rates, increases/decreases between 2021 and 2022 are provided in millimeters or grams.

Annual Monitoring Metrics	Rush Creek - MGORD	Rush Creek - Upper	Rush Creek - Bottomlands	Walker Creek	Lee Vining Creek
Population Estimate <125 mm	N/A	Increase (62%)	Decrease (9%)	Decrease (43%)	Decrease (2%)
Population Estimate 125-199 mm	N/A	Decrease (56%)	Decrease (42%)	Increase (3%)	Decrease (70%)
Population Estimate ≥200 mm	Decrease (20%)	Increase (152%)	Increase (59%)	Decrease (13%)	Increase (90%)
Density Estimate (fish/ha) Age-0	N/A	Increase (66%)	Decrease (15%)	Decrease (38%)	Increase (8%)
Density Estimate (fish/ha) Age-1+	Decrease (32%)	Decrease (33%)	Decrease (28%)	Increase (11%)	Decrease (50%)
Standing Crop (kg/ha)	Decrease (19%)	Increase (17%)	Decrease (4%)	Increase (1%)	Decrease (7%)
Condition Factor	Increase	Increase	Increase	Decrease	Increase
Growth Rate (mm) of Age-1 Recaptures	Increase (23 mm)	Increase (15 mm)	Increase (13 mm)	Increase (6 mm)	Increase (10 mm)
Growth Rate (g) of Age-1 Recaptures	Increase (32 g)	Increase (14 g)	Increase (15 g)	Same	Increase (6 g)
Growth Rate (mm) of Age-2 Recaptures	Increase (26 mm)	Increase (14 mm)	Increase (46 mm)	Increase (12 mm)	Increase (4 mm)
Growth Rate (g) of Age-2 Recaptures	Increase (79 g)	Increase (15 g)	Increase (51 g)	Increase (4 g)	Increase (7 g)
Apparent Survival of Age-1	N/A	Increase	Increase	Increase	Increase
RSD-225	Increase	Increase	Increase	N/A	Increase
RSD-300	Increase	Same	Same	N/A	Same
RSD-375	Same	N/A	N/A	N/A	N/A

Introduction

Study Area

Between September 13th and 23rd 2022, Ross Taylor (the SWRCB's Fisheries Scientist) and a staff of five fisheries biologists conducted the annual fisheries monitoring surveys in six reaches along Rush, Lee Vining, and Walker Creeks in the Mono Lake Basin. The 2022 fisheries sampling was the first of ten post-settlement years of biological monitoring of the Stream Ecosystem Flows (SEF), with oversight from the newly formed Mono Basin Monitoring Administration Team (MAT). The SEFs are an integral part of the amended water licenses in SWRCB's Order WR-2021-0086. The six reaches sampled in 2022 were similar in length to those sampled between 2009 and 2021, except the Lee Vining Creek side channel was dropped from the sample sites (Figure 1). Aerial photographs of the sampling reaches are provided in Appendix A.

Hydrology

The 2022 RY was 60% of normal and classified as a Dry RY type, as measured on April 1st. The range of runoff that defines a Dry RY is ≤ 68.5% (80% - 100% exceedence). The preceding 10 years included a Dry RY of 58% in 2021, Dry-Normal-1 RY of 71% in 2020, a Wet RY of 140% in 2019, a Normal RY of 85% in 2018, a record Extreme-wet RY of 206% in RY 2017 and five consecutive below "Normal" RY's (RY 2016 was 74% of normal, RY 2015 was 25% of normal, RY 2014 was 48% of normal, RY 2013 was 66% of normal and RY 2012 was 55% of normal).

Following the flow regimes developed for Los Angeles Department of Water and Power's (LADWP) State Water Board Order WR-2021-0086, in Dry runoff years, SEFs in Rush Creek are 30 cfs April 1-30, rising to 70 cfs May 18-July 6, descending to a summer baseflow of 30 cfs by July 28, and holding until a winter baseflow of 27 cfs beginning October 1st (Figure 2). The blue line in Figure 2 depicts both snowmelt runoff and Southern Cal Edison (SCE) ramping in Rush Creek upstream of Grant Lake Reservoir (GLR). The red line depicts the SEF releases by LADWP into the top end of the MGORD (Figure 2). The green line in Figure 2 depicts the flows released below GLR with unregulated accretions from Parker and Walker Creeks.

For RY 2022 in Lee Vining Creek, LADWP followed the diversion rate table and fall/winter baseflows consistent with the SEF regime defined in WR-2021-0086 (Figure 3). In 2022, multiple, small peaks occurred in Lee Vining Creek above the intake, with a peak of 144 cfs on May 27th (Figure 3). Consistent with the SEF diversion rate table, LADWP diverted flows from Lee Vining Creek to GLR when flows above the intake were >30 cfs (Figure 3). Flows in Lee Vining Creek were also diverted in September to provide for safer electrofishing, resulting in flows of approximately 25-28 cfs for the duration of the fisheries sampling (Figure 3).

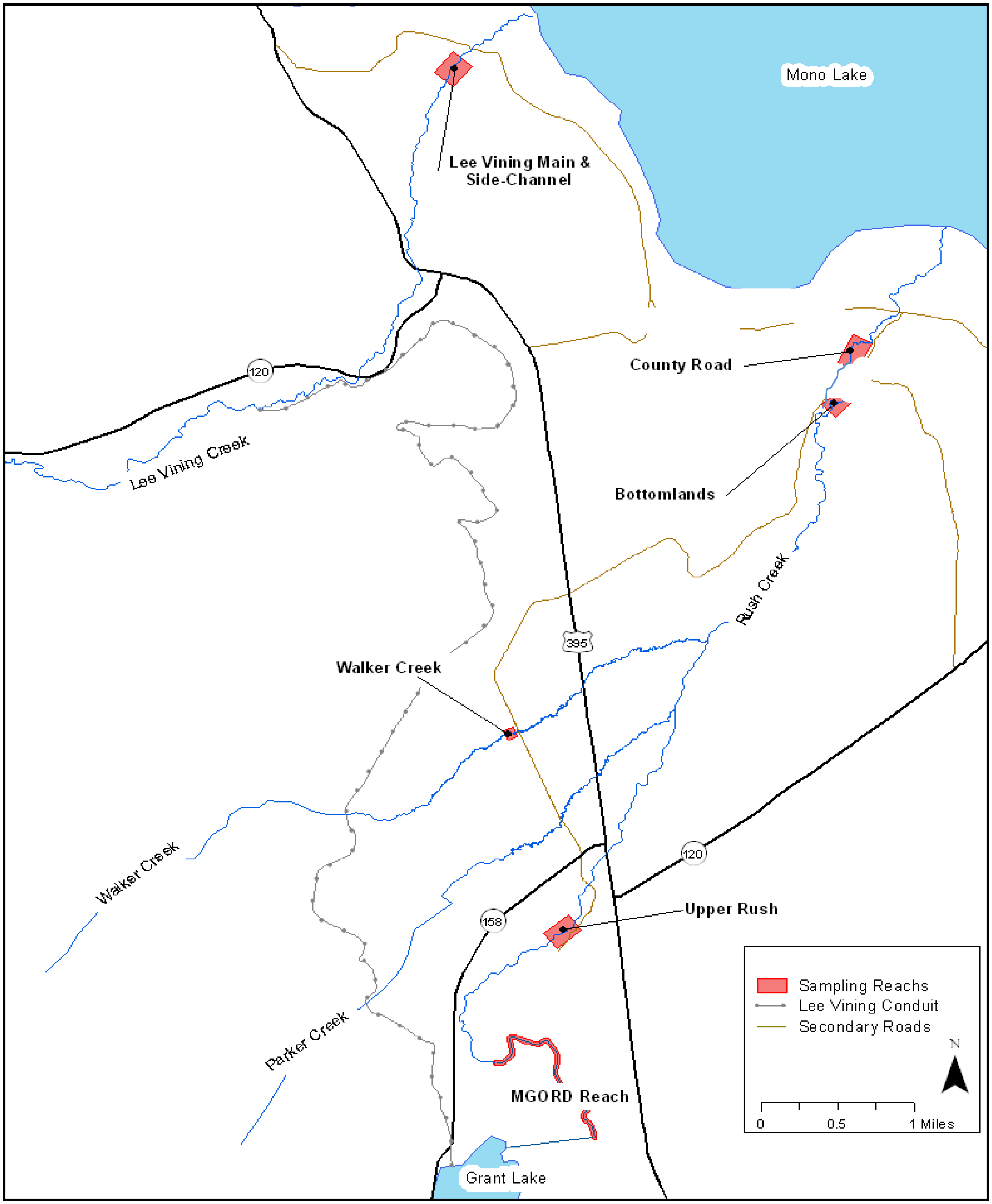

Figure 1. Annual fisheries sampling sites within Mono Basin study area, September 2022.

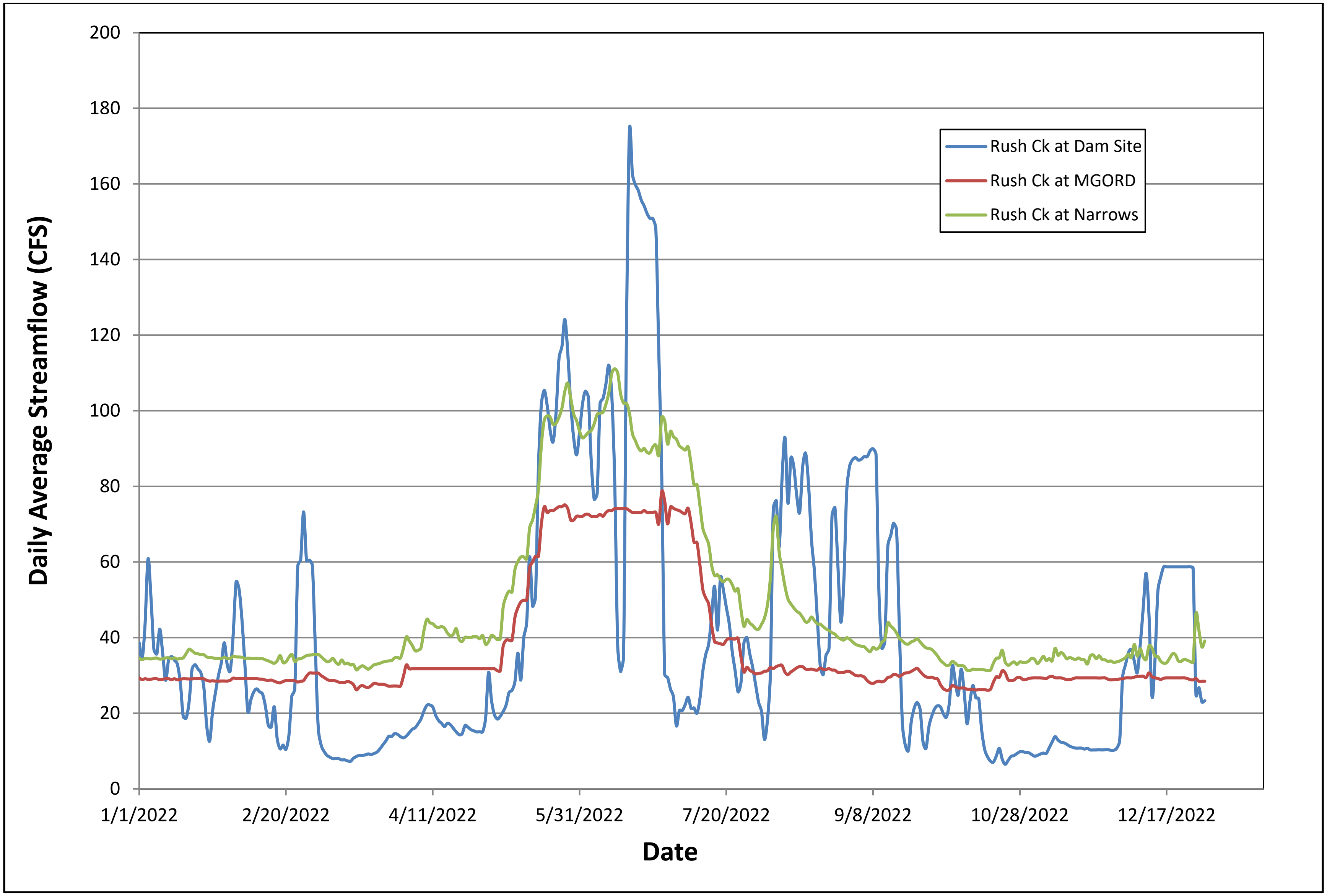

Figure 2. Rush Creek hydrographs between January 1st and December 31st of 2022.

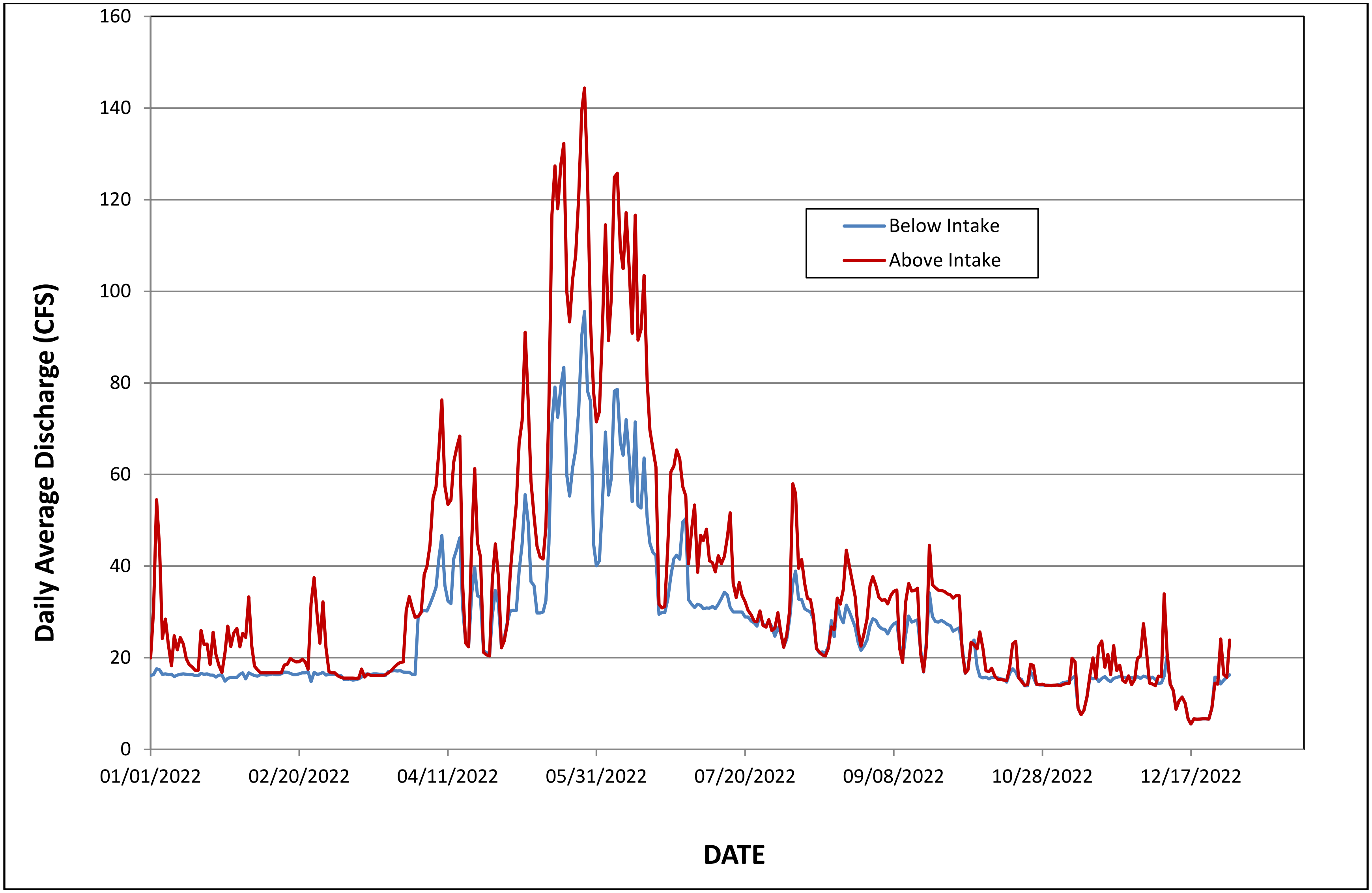

Figure 3. Lee Vining Creek hydrograph between January 1st and December 31st of 2022.

Grant Lake Reservoir (GLR)

In 2022, storage elevation levels in GLR fluctuated from a high of 7,103.4 ft on September 17th to a low of 7,089.4 ft on April 4th (Figure 4). The low level in 2022 of 7,089.4 ft was the lowest level in GLR since 2015 and 2016, the final two years of the five-year drought (Figure 4). In 2022, GLR dropped from January until late April, filled until early June, dropped slightly in June and July, then continued to rise to its 2022 peak in mid-September, and then dropped throughout the remainder of the year (Figure 4).

During the summer months of RY2022, GLR's elevation was below the "low" GLR level for 41 days (7/1-8/10), as defined in the Synthesis Report by the Stream Scientists as a level where warm water temperatures should be a concern (<20,000 AF storage or approximately 7,100 ft elevation) (red horizontal line in Figure 4). From mid-August through mid-September, GLR was 0.1 ft to 3.4 ft above the 7,100 ft elevation. As would be expected, the 2022 summer water temperature monitoring documented concerningly warm water temperatures with sometimes large diurnal fluctuations, leading to less than favorable conditions for Brown Trout growth and survival, at all Rush Creek locations downstream of GLR for variable lengths of the summer period, defined as July through September.

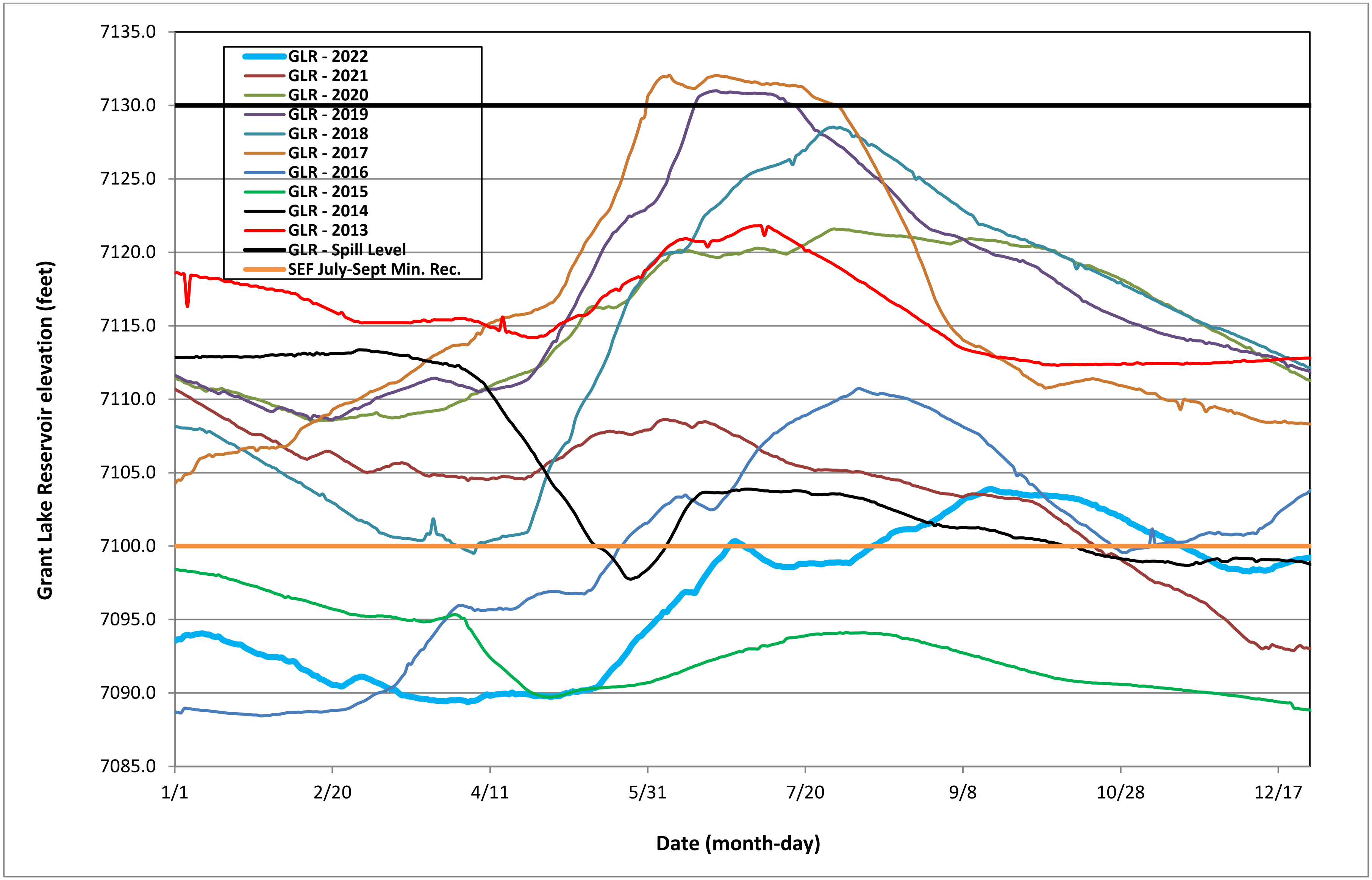

Figure 4. Grant Lake Reservoir's elevation between January 1st and December 31st 2013 - 2022.

Methods

The annual fisheries monitoring was conducted between September 13^{th} and 23^{rd} of 2022. The sampling was conducted by Ross Taylor of Ross Taylor and Associates (RTA), Lawrance Vernallis (RTA employee), Gavin Bandy (RTA employee), and three sub-consultants to RTA: Beth Chasnoff-Long, John Lang, and Tyler Rose. Closed population mark-recapture and depletion methods were utilized to estimate trout abundance. The mark-recapture method was used on the MGORD, Upper and Bottomlands sections of Rush Creek and on the Lee Vining Creek main channel section. The multiple-pass depletion method was used on the Walker Creek section. As previously mentioned, the Lee Vining Creek side-channel section was dropped from the set of sampling sites in 2022.

For the mark-recapture method to meet the assumption of a closed population, semi-permanent block fences were installed at the upper and lower ends of each section. The semi-permanent fences were 48 inches tall, constructed with ½-inch mesh hardware cloth, t-posts, and rope. Hardware cloth was stretched across the entire width of the creek and t-posts were then driven at roughly five-foot intervals through the cloth on the upstream side approximately one foot from the edge. Rocks were placed on the upstream (lower) edge of the fence to prevent trout from swimming underneath the fence. Rope was secured across the tops of the t-posts and anchored to both banks upstream of the fence. The hardware cloth downstream of the t-posts was raised and secured to the rope with bailing wire. Fences were raised the morning of the mark run and left in place for seven days until the recapture run was finished. To prevent failure, all fences were cleaned of leaves, twigs, and checked for mortalities at least twice daily (morning and evening).

Depletion estimates only required a temporary blockage to prevent fish movement in and out of the study area while conducting the survey. Temporary blockage of the sections was achieved with 3/16 inch-mesh nylon seine nets installed across the channel at the upper and lower ends of the study areas. Rocks were placed on the lead line to prevent trout from swimming underneath the seine net. A long T-post was laid across the channel, on top of the banks, and then the cork line of the seine net was zip-tied to the T-post. Both ends of the seine net were weighted with rocks to hold it in place.

Equipment used to conduct mark-run electrofishing on Rush Creek included a seven-foot plastic barge that contained the Smith-Root© 2.5 GPP electrofishing system, an insulated cooler, and battery powered aerators. The Smith-Root© 2.5 GPP electrofishing system included a 5.5 horsepower Honda© generator which powered the 2.5 GPP control box. Electricity from the 2.5 GPP control box was introduced into the water via two anodes. The electrical circuit was completed by the metal plate cathode attached to the bottom of the barge.

Mark-recapture runs on Rush Creek consisted of a single downstream pass starting at the upper block fence and ending at the lower block fence. In 2022, the field crew consisted of a barge operator, two anode operators, and three netters; one for each anode and a “rover” netter. The barge operator’s job consisted of carefully maneuvering the barge down the creek and ensuring overall safety of the entire crew. The anode operators' job was to safely shock and hold trout until they were netted. The netters' job was to net and transport fish to the insulated cooler and monitor trout for signs of stress. Once the cooler was full of fish, electrofishing was temporarily stopped to process the trout. The trout were then transferred from the cooler to live cars and placed back in the creek. The trout were then processed in small batches and then returned to a recovery live car in the creek. Once all the trout were processed at a sub-stop, the crew resumed electrofishing until the cooler was once again full.

The mark-recapture runs on the Lee Vining Creek main channel consisted of an upstream pass starting at the lower block fence to the upper block fence, a short 15–20-minute break, and then a downstream pass back down to the lower fence. The electrofishing crew consisted of two crew members operating Smith-Root© LR-20B and Model-12 POW backpack electrofishers, three netters, and one bucket carrier who transported the captured trout.

Due to the depth of the MGORD, all electrofishing and netting was done from inside a drift boat. The drift boat was held perpendicular to the flow by two crew members who walked it down the channel. The electrofishing barge was tied off to the upstream side of the drift boat and a single throw anode was used. A single netter used a long handled dipnet to net the stunned trout, which were then placed in an insulated cooler equipped with aerators. A safety officer sat at the stern of the drift boat whose job was to monitor the trout in the cooler, the electrofishing equipment, the electrofishing crew, and shut off the power should the need arise. Once the cooler was full, the trout were moved to a live car and placed back in the creek for the shore-based crew to process before continuing the electrofishing effort. Any time the electrofishing crew unloaded fish into a live car ahead of the processing crew, the live car's location was marked with bright-colored survey flagging at the edge of the MGORD road.

For the Walker Creek depletion, a single pass was considered an upstream pass from the lower seine net to the upper seine net followed by a downstream pass back to the lower seine net. One member of the electrofishing crew operated a LR-20B electrofisher; another member was the primary netter and a third member was the backup netter/bucket carrier. The other crew members processed the trout captured during the first pass while the electrofishing crew was conducting the second pass. Processed first pass fish were temporarily held in a live car until the second pass was completed. If it was determined that only two passes were required to generate suitable estimates, all fish were then released. If additional passes were needed, fish from each pass were held in live cars until we determined that no additional electrofishing passes were required to generate reasonable population estimates.

To process trout during the mark-run, small batches of fish from the live car were transferred to a five-gallon bucket equipped with aerators. Trout were then anesthetized, identified as either Brown Trout or Rainbow Trout, measured to the nearest millimeter (total length), and weighed to the nearest gram on an electronic balance. Trout were then "marked" with a small (< 3 mm) fin clip for identification during the recapture run. Trout captured in the Rush Creek Bottom-lands and MGORD sections received anal fin clips and trout captured in the Upper Rush section received lower caudal fin clips. Before placing trout into the aerated recovery bucket, each fish was examined for a missing adipose fin. Trout missing their adipose fin were then scanned for their Passive Integrated Transponder (PIT) tag number. Any trout missing their adipose fin that failed to produce a tag number when scanned were recorded as having “shed” the PIT tag; in most instances these fish were retagged. Partially regenerated adipose fins of fish with PIT tags were reclipped for ease of future identification. Once recovered, fish were then moved from the recovery bucket to a live car to be held until the day’s sampling effort was completed; this was done to prevent captured fish from potentially moving downstream into the actively sampled section. At the end of the electrofishing effort, fish were released from the live cars back into the sub-sections they had been captured in. Fish were then provided a seven-day period to remix back into the section’s population prior to conducting the recapture-run.

Between 2009 and 2012, PIT tags were implanted in most age-0 trout in Rush and Lee Vining Creeks and in all ages of trout in the MGORD. No PIT tags were deployed in 2013; however, the tagging program was resumed during the 2014-2022 field seasons. Starting in 2017, PIT tags implanted in trout caught in the MGORD were focused primarily on fish up to 225 mm in length, with the intent being to tag only age-0 and presumed age-1 trout.

All data collected in the field were written on data sheets and entered into Excel spreadsheets using a field laptop computer. Hard copy data collection was used to provide a crucial back-up in case of in-field technical issues with the laptop. These data sheets were then used to proof the Excel spreadsheets.

Calculations

To calculate the area of each sample section, channel lengths and wetted widths were measured within the sample reaches. Wetted widths were measured at approximately 10-meter intervals to 0.1-meter accuracy within each reach. Average wetted widths and reach lengths were used to generate sample section areas (in hectares), which were then used to calculate each section’s estimates of trout biomass (kg/ha) and density (# of fish/ha).

Mark-recapture population estimates were derived from the Chapman modification of the Petersen equation (Ricker 1975 as cited in Taylor and Knudson 2012. Depletion estimates and condition factors were derived from MicroFish 3.0 software program. Estimates were generated for three size groups of trout: <125 mm in length, 125-199 mm in length, and ≥200 mm in length (200 mm is approximately eight inches).

Mortalities

For the purpose of conducting the mark-recapture methodology, accounting for fish that died during the sampling process was important. Depending on when the fish died (i.e. whether or not they were sampled during the mark-run), dictated how these fish were treated within the estimation process.

All fish that died during the mark-run, and were consequently unavailable for sampling during the recapture-run, were considered as "morts" in the mark-run for the purposes of mark-recapture estimates. These fish were removed from the mark-run data, and then were added back into the total estimate after computing the mark-recapture estimate.

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During the seven-day period between the mark-run and the recapture-run, when the block fences were cleaned twice daily, fence cleaners also looked for additional dead fish, primarily on the lower fences, inside the bounded study sections. When "marked" morts were found on the fences, we went back into the mark-run data and assigned block-fence morts on a one-to-one basis as "morts" to individual fish on the mark-run based on species and size. When this occurred, a comment was added to the individual fish, such as "assigned as fence mort". These marked morts were then removed from the mark-run data since they were unavailable for sampling during the recapture-run. Because of fin deterioration on some morts, exact lengths were not always available. Fortunately, it was not critical to match the exact length when assigning these marked fence morts to fish from the mark-run, but it was important that the fence morts were placed within the proper "length group" for which estimates were computed. As with fish that died during the mark-run, these marked fence morts were added back into the total estimate after the mark-recapture estimate was computed.

Unmarked fence morts (dead fish in the block fences that had not been caught and clipped during the mark-run) were measured and tallied by the three length groups for which estimates were computed. These fish were then added to the total number of morts (for each length group), which were then added back into the mark-recapture estimates to provide unbiased total estimates for each length group.

PIT tags were removed from all morts with previously implanted tags. The PIT tag database was updated to confirm these morts and "tag pulled" was noted, because these tags were reused.

Length-Weight Relationships

Length-weight regressions (Cone 1989 as cited in Taylor and Knudson 2012) were calculated for all Brown Trout greater than 100 mm in all sections of Rush Creek. Regressions using Log10 transformed data were used to compare length-weight relationships by year and by section.

Fulton-type condition factors were computed in MicroFish 3.0 using methods previously reported (Taylor and Knudson 2012) for Brown Trout 150 to 250 mm. A trout condition factor of 1.00 was considered the demarcation between poor and average condition (Reimers 1963; Barnham and Baxter 1998; Blackwell et al. 2000). The literature considers a trout condition factor of <1.00 as poor, a condition factor of 1.00-1.19 as average, and a condition factor >1.20 as good (Barnham and Baxter 1998).

Relative Stock Density (RSD) Calculations

Relative stock density (RSD) is a numerical descriptor of length frequency data (Hunter et al. 2007; Gabelhouse 1984). RSD values are the proportions (percentage x 100) of the total number of Brown Trout ≥150 mm in length that are also ≥225 mm or (RSD-225), ≥300 mm (RSD-300) and ≥375 mm or (RSD-375). A primary purpose of generating RSD values is to describe the structure of a fish population in terms of recreational fishing satisfaction. For Rush and Lee Vining Creeks this would be a descriptor of an eastern Sierra trout stream; as in, out of the estimated numbers of catchable trout (≥150 mm or ≈6 inches) what proportion are "stock" length (≥225 mm or ≈9 inches), "memorable" length (≥300 mm or ≈12 inches), or "trophy" length (≥375 mm or ≈15 inches). These three RSD values are calculated by the following equations:

RSD-225 = [(# of Brown Trout ≥225 mm) ÷ (# of Brown Trout ≥150 mm)] x 100
RSD-300 = [(# of Brown Trout ≥300 mm) ÷ (# of Brown Trout ≥150 mm)] x 100
RSD-375 = [(# of Brown Trout ≥375 mm) ÷ (# of Brown Trout ≥150 mm)] x 100

Water Temperature Monitoring

Water temperatures were recorded (in degrees Fahrenheit) at various locations within Rush and Lee Vining Creeks as part of the Fisheries Monitoring Program. Data loggers were deployed by Robbie Di Paolo of the Mono Lake Committee (MLC) in January and recorded data throughout the year in one-hour time intervals. Data loggers were downloaded at the end of the year and the data were summarized in spreadsheets. Water temperature data loggers were deployed at the following locations in 2022:

1. Rush Creek – top of MGORD.
2. Rush Creek – bottom of MGORD.
3. Rush Creek – at Upper Rush/Old Highway 395 Bridge.
4. Rush Creek – above Parker Creek.
5. Rush Creek – below Narrows.
6. Rush Creek – at County Road crossing.
7. Lee Vining Creek – at County Road crossing.

For the Fisheries Monitoring Program, the year-long data sets were edited to focus on the 2022 summer water temperature regimes (July – September) in Rush Creek. Analysis of summer water temperature included the following metrics:

1. Daily mean temperature.
2. Average daily minimum temperature.
3. Average daily maximum temperature.
4. Number of days with daily maximums exceeding 70°F.
5. Number of hours with temperatures exceeding 66.2°F.
6. Number of good/fair/poor potential growth days, based on daily average temperatures.
7. Number of bad thermal days based on daily average temperatures.
8. Maximum diurnal fluctuations.
9. Average maximum diurnal fluctuations for a consecutive 21-day period.

Results

Channel Lengths and Widths

Differences in wetted widths between years can be due to several factors such as; magnitude of spring peak flows, stream flows at time of measurements, and locations of where the measurements were taken. Lengths, widths, and areas from 2021 were provided for comparisons (Table 2).

Table 2. Total length, average wetted width, and total surface area of sample sections in Rush, Lee Vining, and Walker Creeks sampled between September 13-23, 2022. Values from 2021 provided for comparisons.

Sample Section	Length (m) 2021	Width (m) 2021	Area (m^2) 2021	Length (m) 2022	Width (m) 2022	Area (m^2) 2022	Area (ha) 2022
Rush – Upper	381	7.4	2,819.4	381	7.2	2,743.2	0.2743
Rush - Bottomlands	437	6.6	2,884.2	437	6.6	2,884.2	0.2884
Rush – MGORD	2,230	8.5	18,955.0	2,230	8.5	18,955.0	1.8955
Lee Vining – Main	255	5.3	1,351.5	255	5.1	1,300.5	0.1301
Walker Creek	210	2.1	441.0	202	2.0	399.6	0.0400

Trout Population Abundance

In 2022, a total of 501 Brown Trout ranging in size from 74 mm to 340 mm were captured on the two mark-recapture electrofishing passes in the Upper Rush section (Figure 5); 273 of these fish were caught on the mark-run (Table 3). For comparison, in 2021 a total of 319 Brown Trout were caught on the mark-run. In 2022, age-0 Brown Trout comprised 53% of the total catch (compared to 28% in 2021, 56% in 2020 and 62% in 2019). The Upper Rush section supported an estimated 757 age-0 Brown Trout in 2022 compared to 467 age-0 Brown Trout in 2021 (a 62% increase) (Table 3).

In 2022, the 146 Brown Trout captured in the 125-199 mm size class comprised 29% of the total catch in the Upper Rush section (compared to 64% in 2021). The Upper Rush section supported an estimated 262 Brown Trout in the 125-199 mm size class in 2022, compared to 586 fish in 2021 (a 55% decrease) (Table 3).

Brown Trout ≥200 mm in length comprised 18% of the Upper Rush total catch in 2022 (compared to 8% in 2021). In 2022, Upper Rush supported an estimated 159 Brown Trout ≥200 mm in length compared to an estimate of 63 fish in 2021 (a 152% increase) (Table 3). In 2022, two Brown Trout ≥300 mm in length were captured in the Upper Rush section (Figure 5).

A total of 66 Rainbow Trout were captured in the Upper Rush section comprising 12% of the section's total catch in 2022 (Table 3); Rainbow Trout comprised 11% of the total catch in 2021. The 66 Rainbow Trout ranged in length from 69 mm to 274 mm and 43 of these were age-0 fish (Figure 6). All of the Rainbow Trout appeared to be of naturally produced origin and sufficient numbers of fish were caught to generate an unbiased population estimate of Rainbow Trout <125 mm in length (Table 3). Because of the low numbers of recaptures (<7 fish), estimates for Rainbow Trout 125-199 mm in length and >200mm in length were biased (Table 3). In 2022, the Upper Rush section supported an estimated 118 Rainbow Trout <125 mm in length (77 in 2021, 253 in 2020, and 418 in 2019), an estimated 22 Rainbow Trout 125-199 mm in length (25 in 2021, and 119 in 2020), and an estimated 13 Rainbow Trout ≥200 in length (nine in 2021) (Table 3).

In 2022, a total of 383 Brown Trout ranging in size from 69 mm to 265 mm were captured on the two mark-recapture electrofishing passes in the Bottomlands section of Rush Creek (Figure 7); 154 of these fish were caught on the mark-run (Table 3). For comparison, in 2021 a total of 167 Brown Trout were caught on the mark-run. Brown Trout <125 mm in length comprised 60% of the total catch in 2022 versus 34% of the total catch in 2021. The Bottomlands section supported an estimated 617 Brown Trout <125 mm in length in 2022 versus 677 fish in 2021 (a 9% decrease).

Brown Trout 125-199 mm in length comprised 31% of the total catch in the Bottomlands section in 2022 versus 55% of the total catch in 2021. This section supported an estimated 200 Brown Trout 125-199 mm in length in 2022 (Table 3) compared to 345 fish in 2021 (a 42% decrease).

Brown Trout ≥200 mm in length comprised 10% of the total catch in the Bottomlands section in 2022 (versus 11% in 2021) with the largest trout 265 mm in length (Figure 7). The Bottomlands section supported an estimated 65 Brown Trout ≥200 mm in 2022 compared to 41 trout in 2021 (a 59% increase).

Within the MGORD section of Rush Creek a total of 234 Brown Trout were captured in 2022, with 95 fish caught on the mark-run (Table 3). In comparison, a total of 556 Brown Trout were caught in two passes in 2021. In 2022, Brown Trout ranged in size from 92 mm to 464 mm (Figure 8). A total of 27 Brown Trout <125 mm in length were captured in 2022, which comprised 9% of the total catch of Brown Trout (123 age-0 fish were caught in 2021). No estimate of Brown Trout <125 mm was possible due to the lack of recaptures (Table 3).

In 2022, a total of 28 Brown Trout 125-199 mm in length were caught during the mark-recapture sampling and comprised 12% of the total Brown Trout catch in the MGORD section (82 fish were caught in 2021). No estimate of Brown Trout 125-199 mm was possible due to the lack of recaptures (Table 3).

In 2022, a total of 179 Brown Trout ≥200 mm in length were caught during the mark-recapture sampling and comprised of 77% of the total catch in the MGORD section (315 fish were caught in 2021). The MGORD supported an estimated 498 Brown Trout in the ≥200 mm size class in 2022 (Table 3), compared to 625 fish in 2021, a decrease of 20%.

In 2022, 28 Brown Trout ≥300 mm were captured in the MGORD (47 fish ≥300 mm were captured in 2021). Six Brown Trout ≥375 mm in length were captured in 2022 (compared to 12 fish in 2021, six fish in 2020, four fish in 2019, 15 fish in 2018, 11 fish in 2017 and 20 fish in 2016). In 2022, four of these Brown Trout were >400 mm in length (Figure 8).

In 2022, 14 Rainbow Trout were captured in the MGORD section (Figure 9). In the previous nine years, the Rainbow Trout catch in the MGORD has ranged from zero to 40 fish. Most of the Rainbow Trout captured in 2022 appeared to be of natural origin, with several larger fish exhibiting signs of hatchery origin.

For the past 17 sampling years, electrofishing passes through the MGORD have produced the following total catch values (all size classes of Brown and Rainbow Trout):

- 2022 – Mark run = 100 trout. Recapture run = 148 trout. Two pass average = 124 fish.
- 2021 – Mark run = 273 trout. Recapture run = 387 trout. Two pass average = 330 fish.
- 2020 – Single pass = 457 trout.
- 2019 – Single pass = 361 trout.
- 2018 – Mark run = 233 trout. Recapture run = 188 trout. Two-pass average = 210.5 fish.
- 2017 – Single pass = 203 trout.
- 2016 – Mark run = 121 trout. Recapture run = 110 trout. Two-pass average = 115.5 fish.
- 2015 – Single pass = 176 trout.
- 2014 – Mark run = 206 trout. Recapture run = 268 trout. Two-pass average = 237 fish.
- 2013 – Single pass = 451 trout.
- 2012 – Mark run = 606 trout. Recapture run = 543 trout. Two-pass average = 574.5 fish.
- 2011 – Single pass = 244 trout.
- 2010 – Mark run = 458 trout. Recapture run = 440 trout. Two-pass average = 449 fish.
- 2009 – Single pass = 649 trout.
- 2008 – Mark run = 450 trout. Recapture run = 419 trout. Two-pass average = 434.5 fish.
- 2007 – Single pass = 685 trout.
- 2006 – Mark Run = 283 trout. Recapture run = 375 trout. Two-pass average = 329 fish.

Table 3. Rush Creek mark-recapture estimates for 2022 showing total number of trout marked (M), total number captured on the recapture run (C), total number recaptured on the recapture run (R), and total estimated number and its associated standard error (S.E.) by stream, section, date, species, and size class. Mortalities (Morts) were those trout that were captured during the mark run, but died prior to the recapture run. Mortalities were not included in mark-recapture estimates and were added to estimates for accurate total estimates. NP = estimate not possible. BNT = Brown Trout. RBT = Rainbow Trout. * = biased due to low recaps.

Stream / Section / Species / Date	Size Class (mm)	M	C	R	Morts	**Estimate**	S.E.
Rush Creek							
Upper Rush - BNT							
9/14/2022 & 9/21/2022							
	0 - 124 mm	145	138	26	6	**757**	115
	125 - 199 mm	72	99	27	2	**262**	32
	≥200 mm	47	62	18	1	**159**	23
Upper Rush - **RBT**							
9/14/2022 & 9/21/2022							
	0 - 124 mm	21	26	6	0	**118**	39
	125 - 199 mm	9	8	3	0	**22***	34
	≥200 mm	6	5	2	0	**13***	14
Bottomlands - BNT							
9/13/2022 & 9/20/2022							
	0 - 124 mm	85	164	22	1	**617**	100
	125 - 199 mm	51	87	22	2	**200**	26
	≥200 mm	15	28	6	0	**65**	15
MGORD - BNT							
9/15/2022 & 9/22/2022							
	0 - 124 mm	6	21	0	0	**NP**	NP
	125 - 199 mm	11	18	1	0	**NP**	NP
	≥200 mm	78	119	18	0	**498**	89
Lee Vining Creek							
Main Channel - BNT							
9/16/2022 & 9/23/2022							
	0 - 124 mm	16	19	5	0	**56**	14
	125 - 199 mm	39	38	12	0	**119**	22
	≥200 mm	47	40	19	0	**97**	12

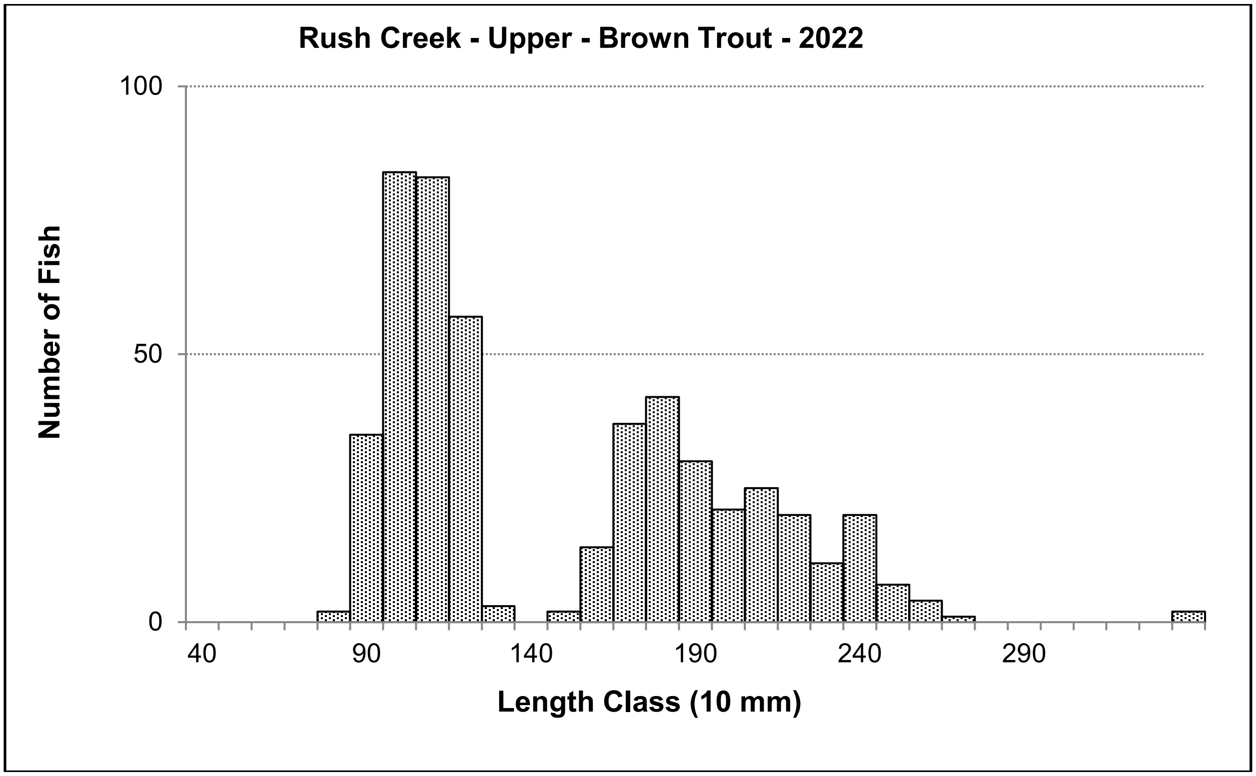


Figure 5. Length-frequency histogram of Brown Trout captured in Upper Rush, September 14th and 21st, 2022.

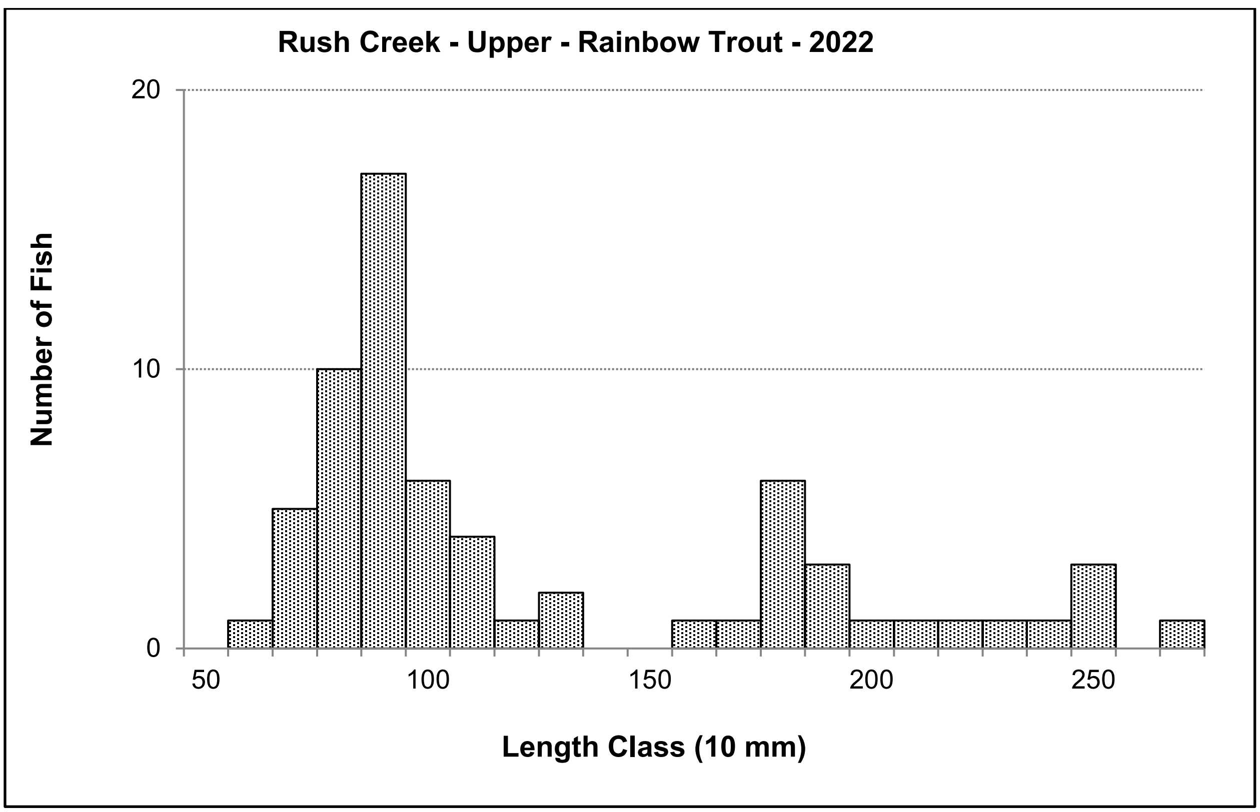


Figure 6. Length-frequency histogram of Rainbow Trout captured in Upper Rush, September 14th and 21st, 2022.

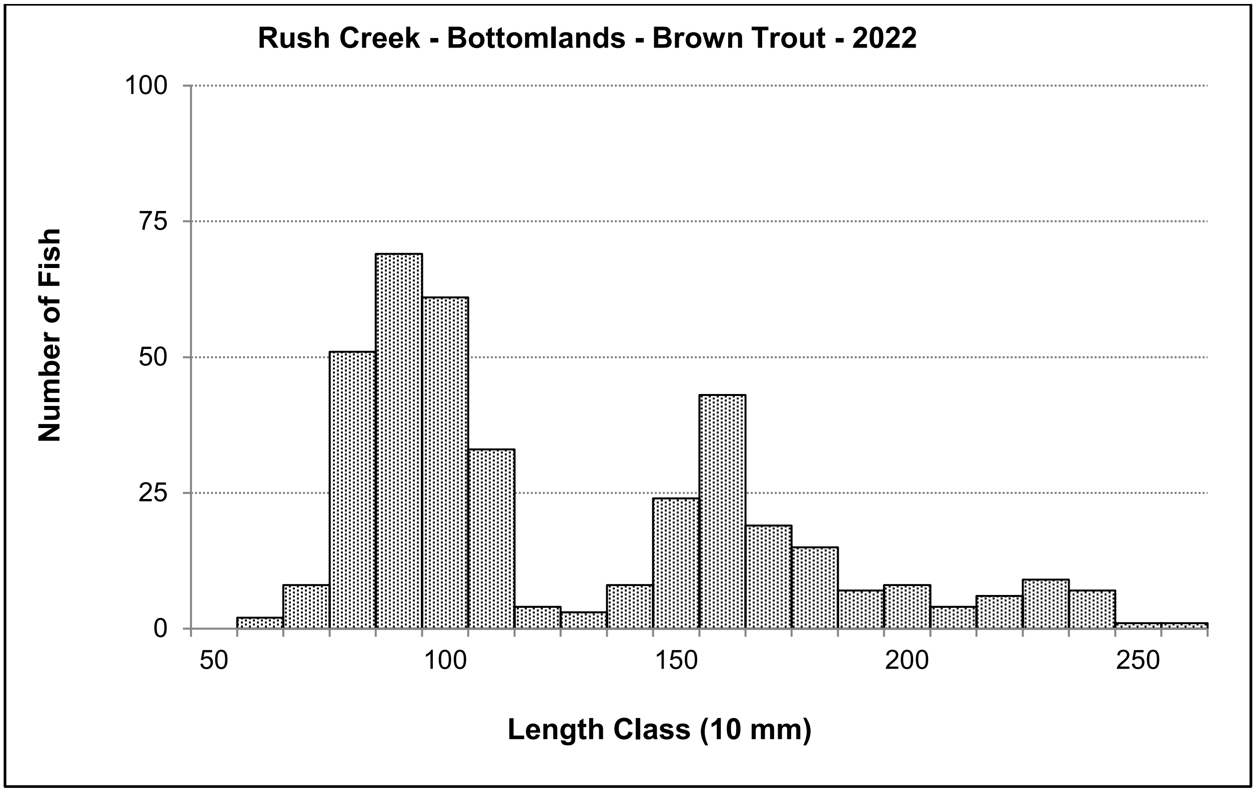


Figure 7. Length-frequency histogram of Brown Trout captured in the Bottomlands section of Rush Creek, September 13th and 20th, 2022.

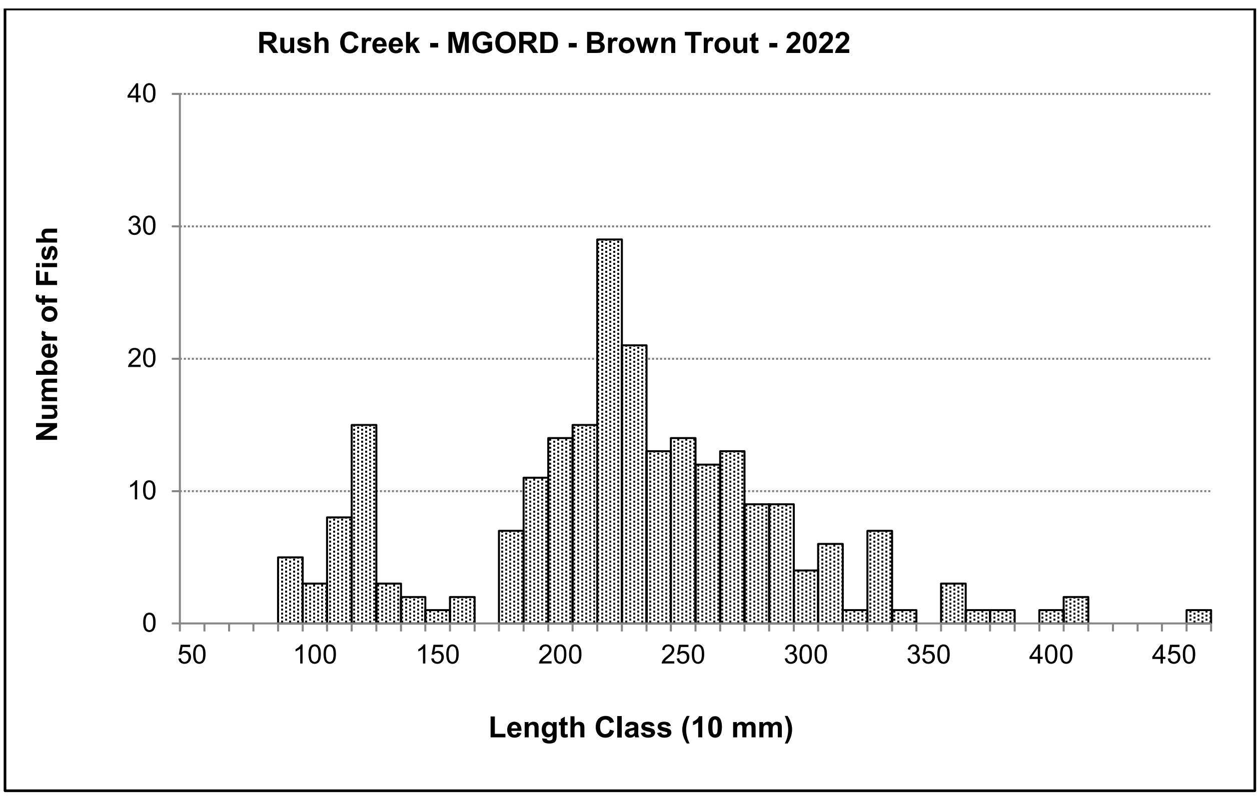


Figure 8. Length-frequency histogram of Brown Trout captured in the MGORD section of Rush Creek, September 15th and 22nd, 2022.

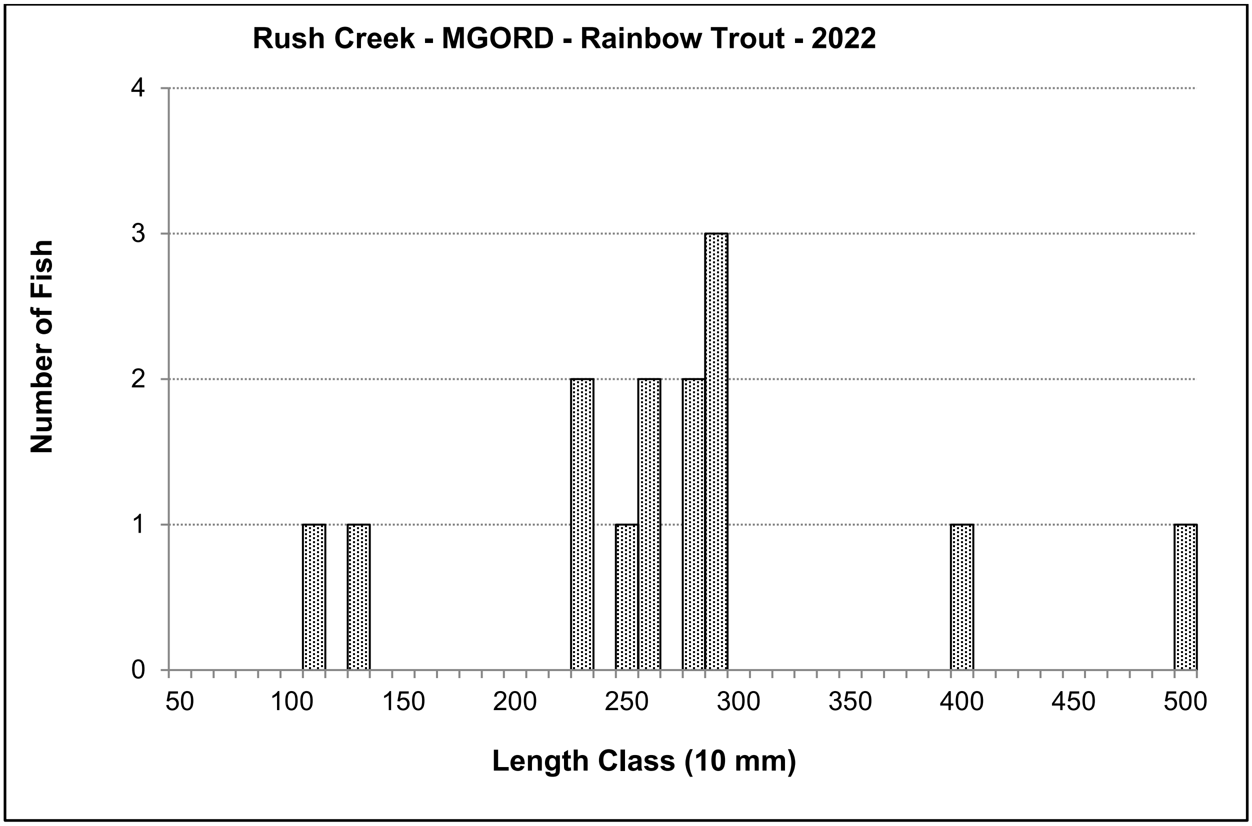


Figure 9. Length-frequency histogram of Rainbow Trout captured in the MGORD section of Rush Creek, September 15th and 22nd, 2022.

__Lee Vining Creek__

In 2022, a total of 165 trout were captured on the mark-recapture electrofishing passes made in the Lee Vining Creek main channel section (Table 3) versus 319 trout in 2021 . Nearly all of the trout captured in 2022 were Brown Trout (163 fish). In 2022, Brown Trout ranged in size from 76 mm to 277 mm in length (Figure 10). Fish <125 mm in length comprised 18% of the total Brown Trout catch in 2022, compared to 12% in 2021, 60% in 2020, 63% in 2019 and 62% in 2018. In 2022, the Lee Vining Creek's main channel section supported an estimated 56 Brown Trout in the <125 mm size class (Table 3), compared to an estimated 57 fish in 2021; two extremely low estimates following an estimate of 449 Brown Trout <125 mm in 2020.

In 2022, Brown Trout 125-199 mm in length comprised 40% of the total Brown Trout catch in Lee Vining Creek's main channel section (versus 76% in 2021). This section supported an estimated 119 Brown Trout 125-199 mm in length in 2022 (Table 3) compared to 402 fish in 2021 (a 70% decrease).

In 2022, the population estimate of Brown Trout ≥200 mm in Lee Vining Creek's main channel was 97 fish (versus 51 fish in 2021, 24 fish in 2020 and 48 fish in 2019) (Table 3). No Brown Trout captured in 2022 were >300 mm in length (Figure 10).

No population estimate was generated for Rainbow Trout in Lee Vining Creek due to insufficient numbers of fish, with only two captured during the mark-recapture electrofishing passes made in 2022. These fish were 97 and 251 mm in length.

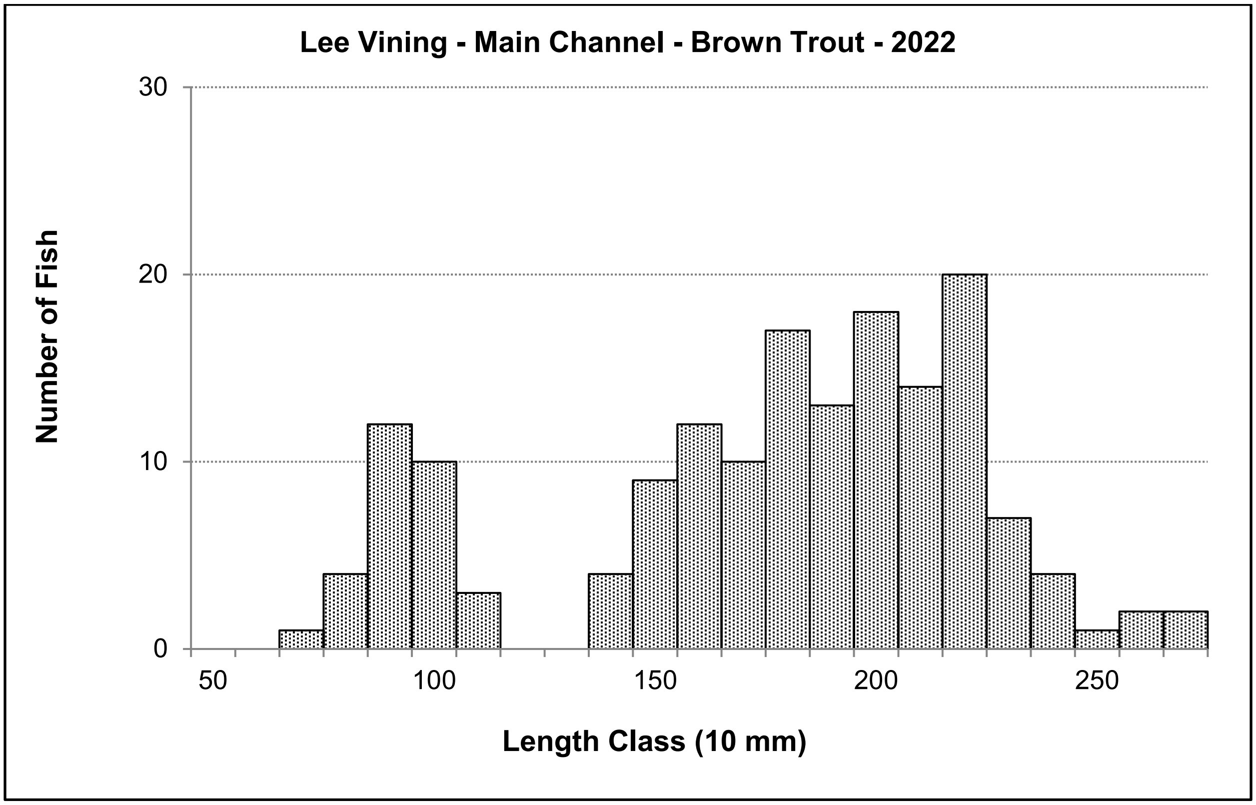


Figure 10. Length-frequency histogram of Brown Trout captured in the main channel section of Lee Vining Creek, September 16th and 23rd, 2022.

__Walker Creek__

In 2022, 249 Brown Trout were captured in two electrofishing passes in the Walker Creek section (Table 4) (versus 356 caught in 2021, 362 caught in 2020, 278 caught in 2019, and 175 caught in 2018). Ninety-eight of these captured fish, or 39%, were age-0 fish ranging in size from 70 mm to 101 mm in length (Figure 11). The break in the length-frequency histogram (Figure 11) and one PIT tag recapture suggest that age-1 trout were as small as 114 mm in 2022. The 2022 estimated population of Brown Trout <125 mm in length was 129 fish (Table 3), compared to 227 fish in 2021, a 43% decrease. For trout <125 mm in length, the probability of capture in 2022 equaled 0.67 (Table 4).

Brown Trout in the 125-199 mm size class (120 fish) accounted for 48% of Walker Creek's total catch in 2022. The 2022 population estimate for Brown Trout in the 125-199 mm size class was 123 trout (a 3% increase from the 2021 estimate) with a probability of capture of 0.83 (Table 4).

Brown Trout ≥200 mm in length (14 fish caught) accounted for 6% of the total catch in 2022. The 2022 population estimate for this size class was 14 Brown Trout with a probability of capture of 0.88 (Table 4). The largest Brown Trout captured in Walker Creek in 2022 was 232 mm in length (Figure 11).

In 2022, one Rainbow Trout was captured in Walker Creek and this fish was 167 mm in length.

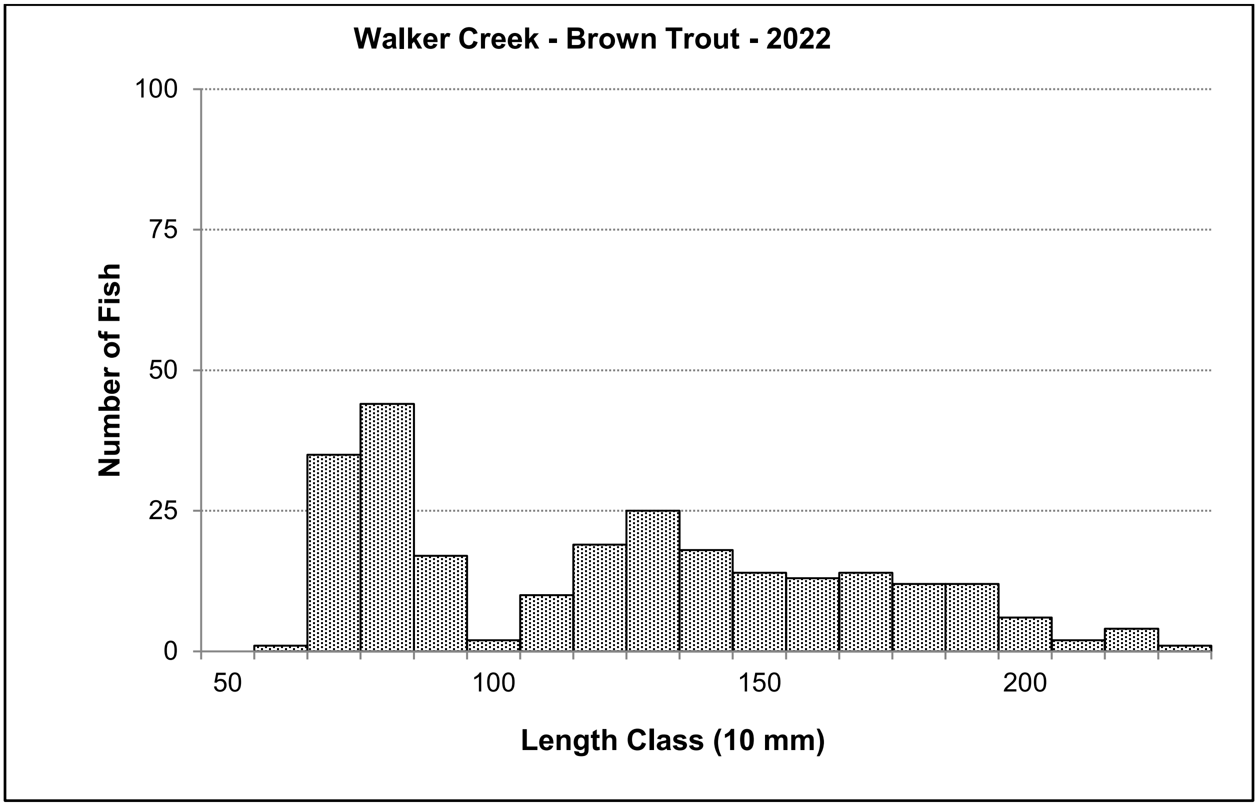


Figure 11. Length-frequency histogram of Brown Trout captured in Walker Creek, September 19th, 2022.

Table 4. Depletion estimates made in Walker Creek during September 2022 showing number of trout captured in each pass, estimated number, probability of capture (P.C.) by size class.

Stream - Section Date **Species**	Size Class (mm)	Removals	Removal Pattern	**Estimate**	P.C.
Walker Creek - above old Hwy 395 - 9/19/2022					
Brown Trout					
	0 - 124 mm	2	85 30	**129**	0.67
	125 - 199 mm	2	101 19	**123**	0.83
	200 + mm	2	12 2	**14**	0.88

Relative Condition of Brown Trout

Linear regressions of log-length to log-weight for captured Brown Trout ≥ 100 mm indicated strong correlations between length and weight (r^2 values 0.98 and greater; Table 5). Slopes of these relationships were near 3.0 indicating isometric growth, which was assumed to compute fish condition factors, and were reasonable (Table 5).

Table 5. Regression statistics for $\log_{10}$ transformed length (L) to weight (WT) for Brown Trout 100 mm and longer captured in Rush Creek by sample section and year. The 2022 regression equations are in ***bold*** type.

Section	Year	N	Equation	r^2	P
Bottomlands	***2022***	***253***	***$Log_{10}(WT) = 3.1013*Log_{10}(L) – 5.2251$***	***0.98***	***<0.01***
	2021	205	$Log_{10}(WT) = 3.0091*Log_{10}(L) – 5.0526$	0.98	<0.01
	2020	223	$Log_{10}(WT) = 2.9792*Log_{10}(L) – 4.9754$	0.98	<0.01
	2019	310	$Log_{10}(WT) = 2.9631*Log_{10}(L) – 4.9409$	0.99	<0.01
	2018	226	$Log_{10}(WT) = 2.9019*Log_{10}(L) – 4.8059$	0.99	<0.01
	2017	160	$Log_{10}(WT) = 3.0398*Log_{10}(L) – 5.0998$	0.99	<0.01
	2016	132	$Log_{10}(WT) = 3.0831*Log_{10}(L) – 5.2137$	0.99	<0.01
	2015	301	$Log_{10}(WT) = 3.0748*Log_{10}(L) – 5.1916$	0.99	<0.01
	2014	238	$Log_{10}(WT) = 3.0072*Log_{10}(L) – 5.0334$	0.98	<0.01
	2013	247	$Log_{10}(WT) = 2.7997*Log_{10}(L) – 4.5910$	0.98	<0.01
	2012	495	$Log_{10}(WT) = 2.8149*Log_{10}(L) – 4.6206$	0.98	<0.01
	2011	361	$Log_{10}(WT) = 2.926*Log_{10}(L) – 4.8580$	0.99	<0.01
	2010	425	$Log_{10}(WT) = 2.999*Log_{10}(L) – 5.0050$	0.99	<0.01
	2009	511	$Log_{10}(WT) = 2.920*Log_{10}(L) – 4.8210$	0.99	<0.01
	2008	611	$Log_{10}(WT) = 2.773*Log_{10}(L) – 4.5240$	0.99	<0.01
Upper Rush	***2022***	***392***	***$Log_{10}(WT) = 2.9632*Log_{10}(L) – 4.9305$***	***0.99***	***<0.01***
	2021	441	$Log_{10}(WT) = 2.9851*Log_{10}(L) – 4.9837$	0.98	<0.01
	2020	426	$Log_{10}(WT) = 2.9187*Log_{10}(L) – 4.8382$	0.99	<0.01
	2019	686	$Log_{10}(WT) = 2.9667*Log_{10}(L) – 4.9298$	0.99	<0.01
	2018	391	$Log_{10}(WT) = 2.9173*Log_{10}(L) – 4.8237$	0.99	<0.01
	2017	309	$Log_{10}(WT) = 3.0592*Log_{10}(L) – 5.1198$	0.99	<0.01
	2016	176	$Log_{10}(WT) = 3.0702*Log_{10}(L) – 5.1608$	0.99	<0.01
	2015	643	$Log_{10}(WT) = 2.9444*Log_{10}(L) – 4.8844$	0.99	<0.01
	2014	613	$Log_{10}(WT) = 2.9399*Log_{10}(L) – 4.8705$	0.99	<0.01
	2013	522	$Log_{10}(WT) = 2.9114*Log_{10}(L) – 4.8160$	0.99	<0.01
	2012	554	$Log_{10}(WT) = 2.8693*Log_{10}(L) – 4.7210$	0.99	<0.01

Table 5 (continued).

Section	Year	N	Equation	r^2	P
Upper Rush	2011	547	$Log_{10}(WT) = 3.006*Log_{10}(L) – 5.0140$	0.99	<0.01
	2010	420	$Log_{10}(WT) = 2.995*Log_{10}(L) – 4.9941$	0.99	<0.01
	2009	612	$Log_{10}(WT) = 2.941*Log_{10}(L) – 4.8550$	0.99	<0.01
	2008	594	$Log_{10}(WT) = 2.967*Log_{10}(L) – 4.9372$	0.99	<0.01
	2007	436	$Log_{10}(WT) = 2.867*Log_{10}(L) – 4.7150$	0.99	<0.01
	2006	485	$Log_{10}(WT) = 2.99*Log_{10}(L) – 4.9802$	0.99	<0.01
	2005	261	$Log_{10}(WT) = 3.02*Log_{10}(L) – 5.0203$	0.99	<0.01
	2004	400	$Log_{10}(WT) = 2.97*Log_{10}(L) – 4.9430$	0.99	<0.01
	2003	569	$Log_{10}(WT) = 2.96*Log_{10}(L) – 4.8920$	0.99	<0.01
	2002	373	$Log_{10}(WT) = 2.94*Log_{10}(L) – 4.8670$	0.99	< 0.01
	2001	335	$Log_{10}(WT) = 2.99*Log_{10}(L) – 4.9630$	0.99	< 0.01
	2000	309	$Log_{10}(WT) = 3.00*Log_{10}(L) – 4.9610$	0.98	< 0.01
	1999	317	$Log_{10}(WT) = 2.93*Log_{10}(L) – 4.8482$	0.98	< 0.01
MGORD	***2022***	***229***	$\boldsymbol{Log_{10}(WT) = 3.1344*Log_{10}(L) – 5.3145}$	***0.99***	***<0.01***
	2021	498	$Log_{10}(WT) = 2.9447*Log_{10}(L) – 4.8871$	0.99	<0.01
	2020	383	$Log_{10}(WT) = 3.0144*Log_{10}(L) – 5.0575$	0.98	<0.01
	2019	314	$Log_{10}(WT) = 2.9774*Log_{10}(L) – 4.9282$	0.98	<0.01
	2018	350	$Log_{10}(WT) = 3.0023*Log_{10}(L) – 5.0046$	0.98	<0.01
	2017	159	$Log_{10}(WT) = 3.0052*Log_{10}(L) – 5.0205$	0.99	<0.01
	2016	183	$Log_{10}(WT) = 3.0031*Log_{10}(L) – 5.3093$	0.99	<0.01
	2015	172	$Log_{10}(WT) = 3.131*Log_{10}(L) – 5.0115$	0.99	<0.01
	2014	399	$Log_{10}(WT) = 2.9805*Log_{10}(L) – 4.9827$	0.98	<0.01
	2013	431	$Log_{10}(WT) = 2.8567*Log_{10}(L) – 4.6920$	0.98	<0.01
	2012	795	$Log_{10}(WT) = 2.9048*Log_{10}(L) – 4.8081$	0.99	<0.01
	2011	218	$Log_{10}(WT) = 2.917*Log_{10}(L) – 4.8230$	0.98	<0.01
	2010	694	$Log_{10}(WT) = 2.892*Log_{10}(L) – 4.7563$	0.98	<0.01
	2009	689	$Log_{10}(WT) = 2.974*Log_{10}(L) – 4.9330$	0.99	<0.01
	2008	862	$Log_{10}(WT) = 2.827*Log_{10}(L) – 4.6020$	0.98	<0.01
	2007	643	$Log_{10}(WT) = 2.914*Log_{10}(L) – 4.8254$	0.98	<0.01
	2006	593	$Log_{10}(WT) = 2.956*Log_{10}(L) – 4.8722$	0.98	<0.01
	2004	449	$Log_{10}(WT) = 2.984*Log_{10}(L) – 4.9731$	0.99	<0.01
	2001	769	$Log_{10}(WT) = 2.873*Log_{10}(L) – 4.7190$	0.99	<0.01

Condition factors of Brown Trout 150 to 250 mm in length in 2022 increased from 2021 values in three Rush Creek sections, increased in the Lee Vining Creek main channel section, and decreased in the Walker Creek section (Figures 12 and 13). In 2022, two sections had Brown Trout condition factors ≥1.00, after two consecutive years that no sections supported Brown Trout with condition factors ≥1.00 (Figures 12 and 13).

Brown Trout in the Upper Rush section had a condition factor of 0.97 in 2022 a slight increase from 0.96 in 2021 and 0.95 in 2020 (Figure 12). The Upper Rush section has had Brown Trout condition factors ≥1.00 in 11 of 23 sampling seasons (Figure 12).

Brown Trout in the Bottomlands section of Rush Creek had a condition factor of 1.01 in 2022, an increase from 0.93 in 2021 (Figure 12). In 15 years of sampling the Bottomlands section, the 2022 sampling was the first time that the Brown Trout condition factor was ≥1.00 (Figure 12).

The MGORD's 2022 Brown Trout condition factor was 1.01, a slight increase from 0.98 in 2021 (Figure 12). In 2022, condition factors for larger Brown Trout in the MGORD were also computed: fish ≥300 mm had a condition factor of 1.09 (0.94 in 2021) and fish ≥375 mm had a condition factor of 1.28 (0.97 in 2021).

In 2022, the condition factor for Brown Trout in Lee Vining Creek's main channel was 0.98, an increase from 0.94 in 2021 (Figure 13). The main channel's 2022 value was the fourth straight year that Brown Trout condition factors were less than 1.00 (Figure 13). In 2022, a Rainbow Trout condition factor was not computed for the Lee Vining Creek main channel because of the extremely small sample size (one fish between 150 to 250 mm in length).

In Walker Creek, Brown Trout had a condition factor of 0.88 in 2022, a decrease from 0.94 in 2021, 0.96 in 2020 and 0.98 in 2019 (Figure 12). Brown Trout condition factors in Walker Creek have been ≥1.00 in 12 of the 23 sampling years; however, in the past eight years only the 2018 sampling year had a condition factor ≥1.00 (Figure 12).

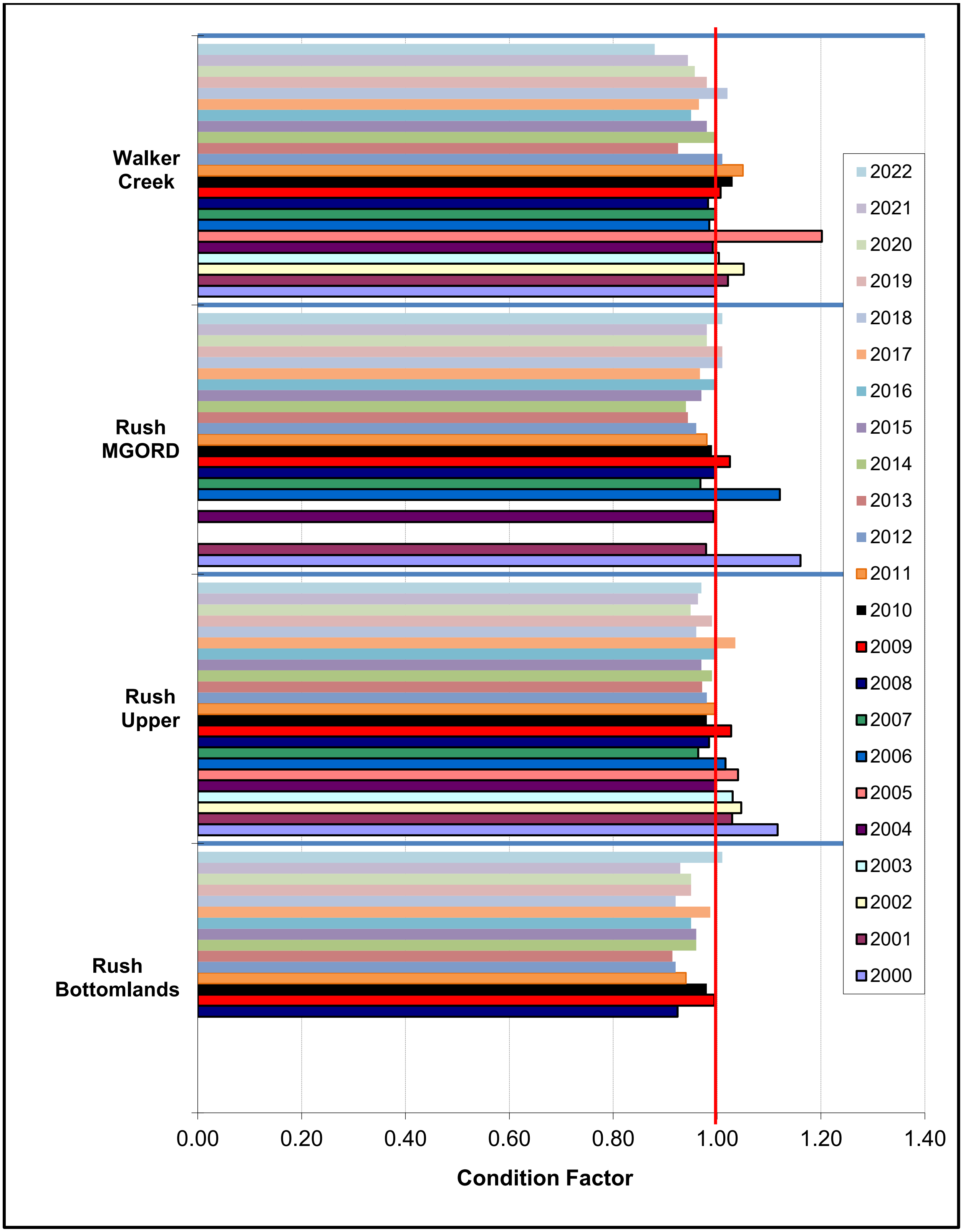


Figure 12. Condition factors for Brown Trout 150 mm to 250 mm in length from sample sections of Rush Creek and Walker Creeks from 2000 to 2022.

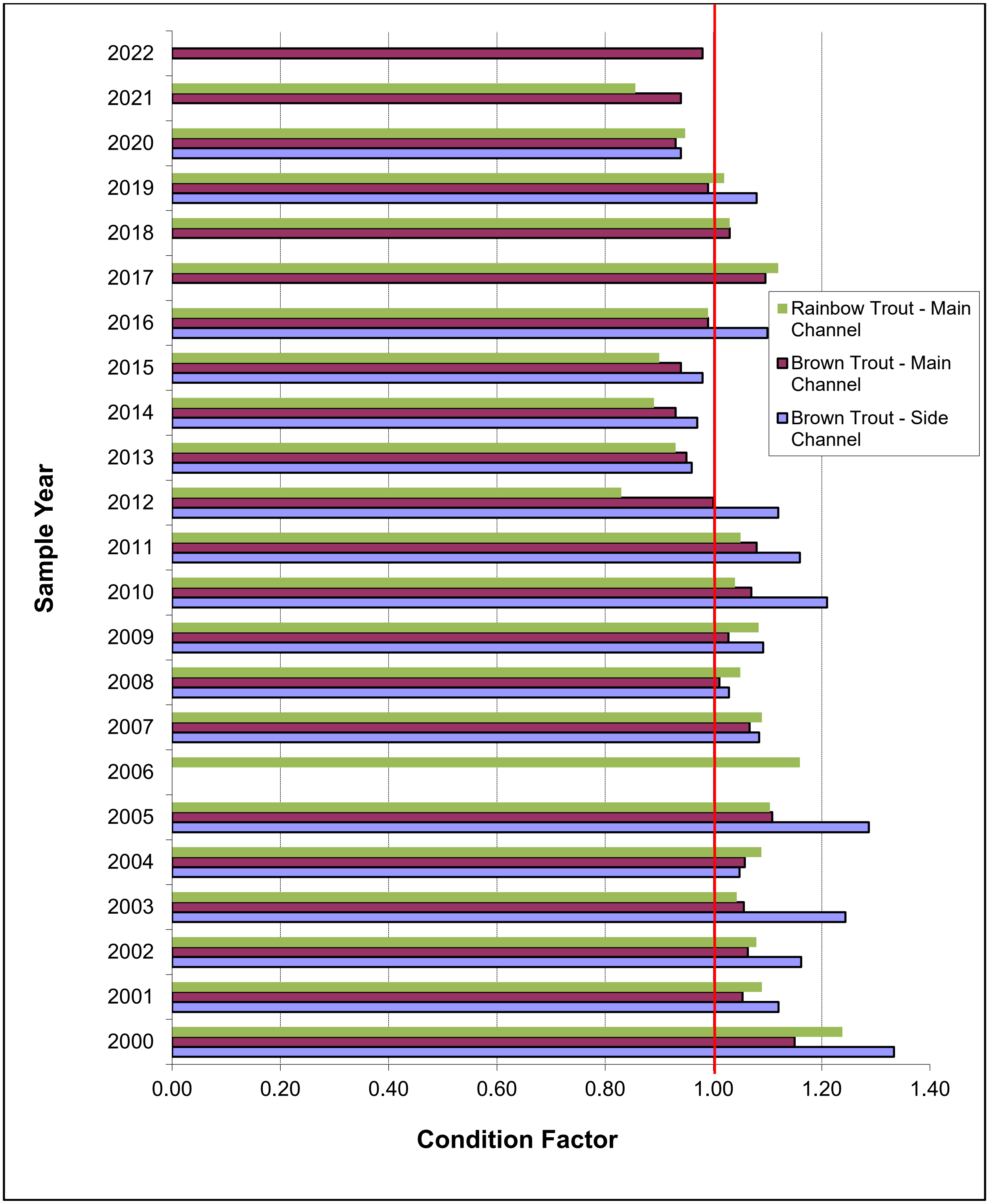


Figure 13. Condition factors for Rainbow Trout and Brown Trout 150 to 250 mm in length from the main channel and side channel sections of Lee Vining Creek from 2000 to 2022. Main channel was not sampled in 2006 due to high flows. The side channel was dropped from the annual sampling in 2022.

Estimated Trout Densities Expressed in Numbers per Hectare

Age-0 Brown Trout

The Upper Rush section had an estimated density of 2,760 age-0 Brown Trout/ha in 2022, an increase of 66% from 2021's estimate of 1,657 age-0 Brown Trout/ha (Figure 14). The Upper Rush section has a 23-year average of 5,521 age-0 Brown Trout/ha.

The Bottomlands section of Rush Creek had a density estimate of 1,988 age-0 Brown Trout/ha in 2022, a 15% decrease from 2021's estimate of 2,347 age-0 trout/ha (Figure 14). The Bottomlands section has a 15-year average of 2,111 age-0 Brown Trout/ha.

In Walker Creek, the 2022 density estimate of 3,193 age-0 Brown Trout/ha was a 38% decrease from the 2021 estimate of 5,147 age-0 trout/ha (Figure 14). This section has a 24-year average of 3,558 age-0 Brown Trout/ha.

In 2022, the estimated density of age-0 Brown Trout in the main channel of Lee Vining Creek was 454 age-0 trout/ha, which was an 8% increase from the 2021 density estimate of 419 age-0 trout/ha (Figure 15). Sample years 2021 and 2022 were two relatively low recruitment years for age-0 Brown Trout after two relatively high years in 2019 and 2020 (Figure 15).

Age-1 and older (aka Age-1+) Brown Trout

The Upper Rush section had an estimated density of 1,535 age-1+ Brown Trout/ha in 2022, a decrease of 33% from the 2021 estimate of 2,302 trout/ha (Figure 16). For the Upper Rush section, the 24-year long-term average equaled 1,536 age-1+ Brown Trout/ha.

The Bottomlands section of Rush Creek had an estimated density of 966 age-1+ Brown Trout in 2022, a 28% decrease from the 2021 estimate of 1,338 age-1+trout/ha (Figure 16). For the Bottomlands section, the 15-year long-term average equaled 1,116 age-1+ Brown Trout/ha.

The estimated density of age-1+ Brown Trout in the MGORD section of Rush Creek in 2022 was 278 fish/ha, a 32% decrease from the 2021 estimate of 411 age-1+trout/ha (Figure 16). Since 2001, for the 12 seasons where density estimates were generated for the MGORD, the long-term density estimate of age-1+ Brown Trout averaged 442 fish/ha.

The 2022 density estimate for age-1+ Brown Trout in Walker Creek was 3,391 age-1+ trout/ha which was an 11% increase from the 2021 estimate of 3,061 age-1+ trout/ha (Figure 16). For Walker Creek, the 24-year long-term average equaled 2,043 age-1+ Brown Trout/ha.

The 2022 density estimate for age-1+ Brown Trout in the Lee Vining Creek main channel section was 1,660 trout/ha, a 50% decrease from the 2021 estimate of 3,350 age-1+ trout/ha (Figure 17). For the Lee Vining Creek main channel section, the 23-year long-term average equaled 1,241 age-1+ Brown Trout/ha.

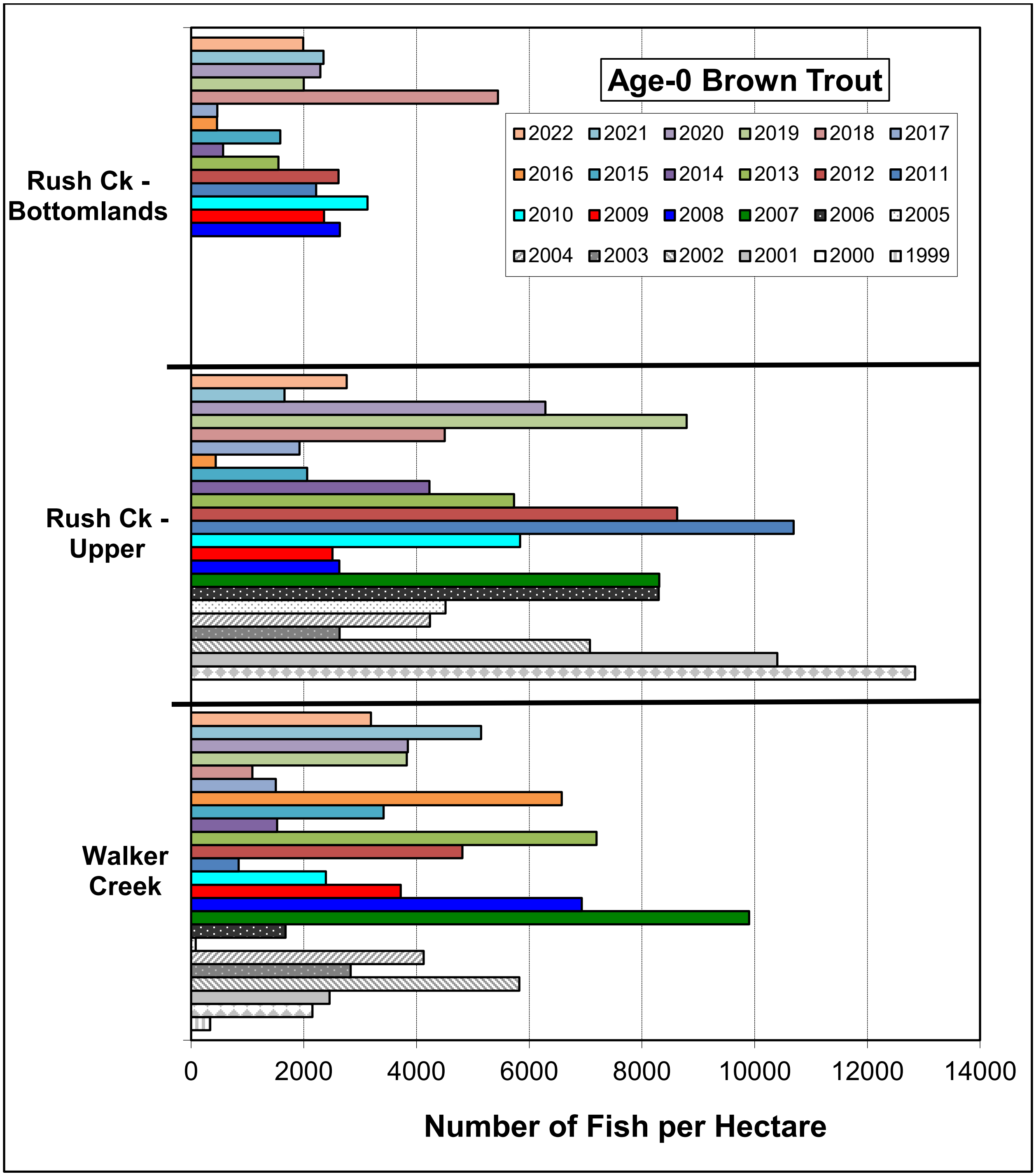


Figure 14. Estimated number of age-0 Brown Trout per hectare in Rush Creek and Walker Creek from 1999 to 2022.

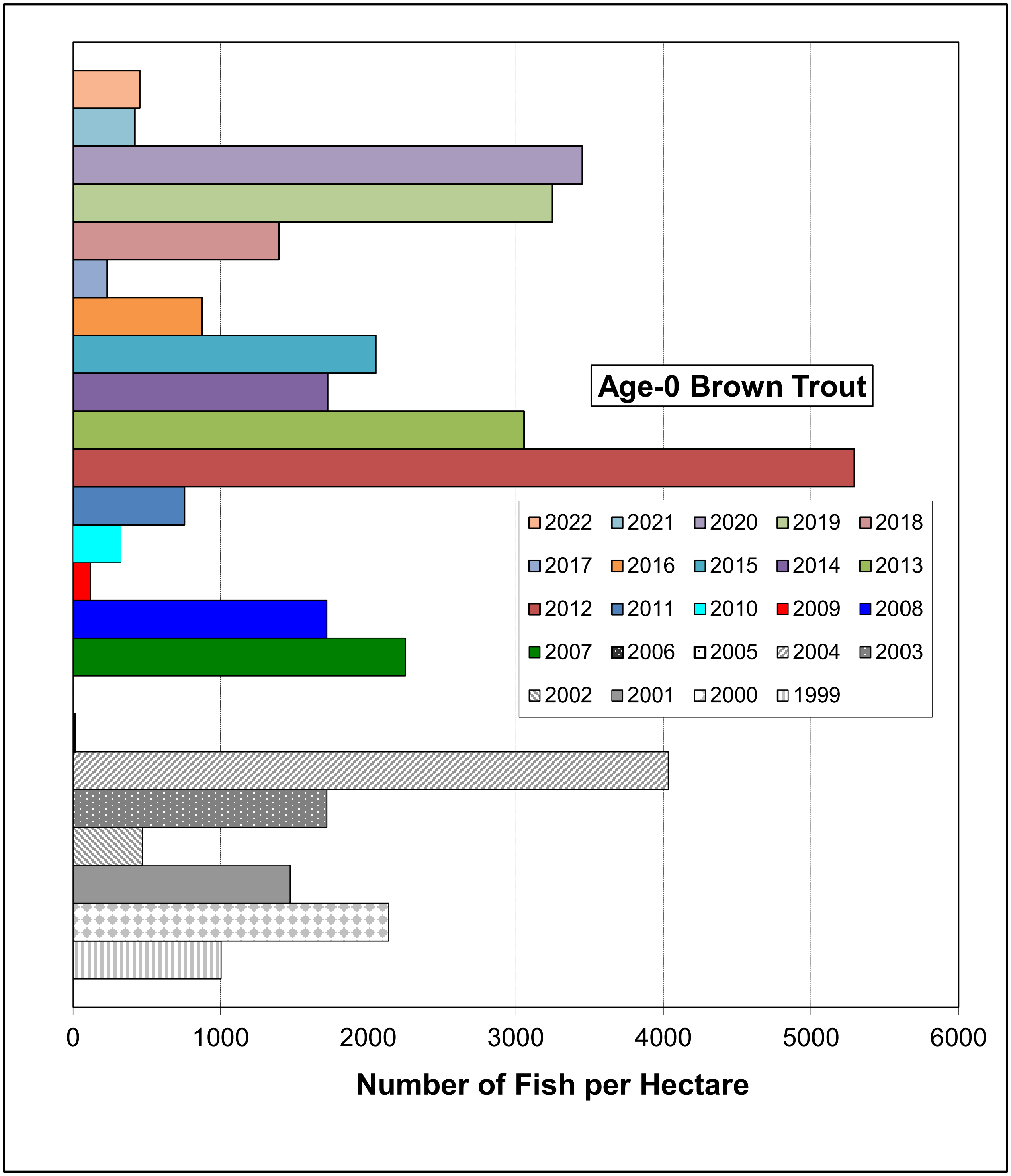


Figure 15. Estimated number of age-0 Brown Trout per hectare in the Lee Vining Creek main channel from 1999 to 2022.

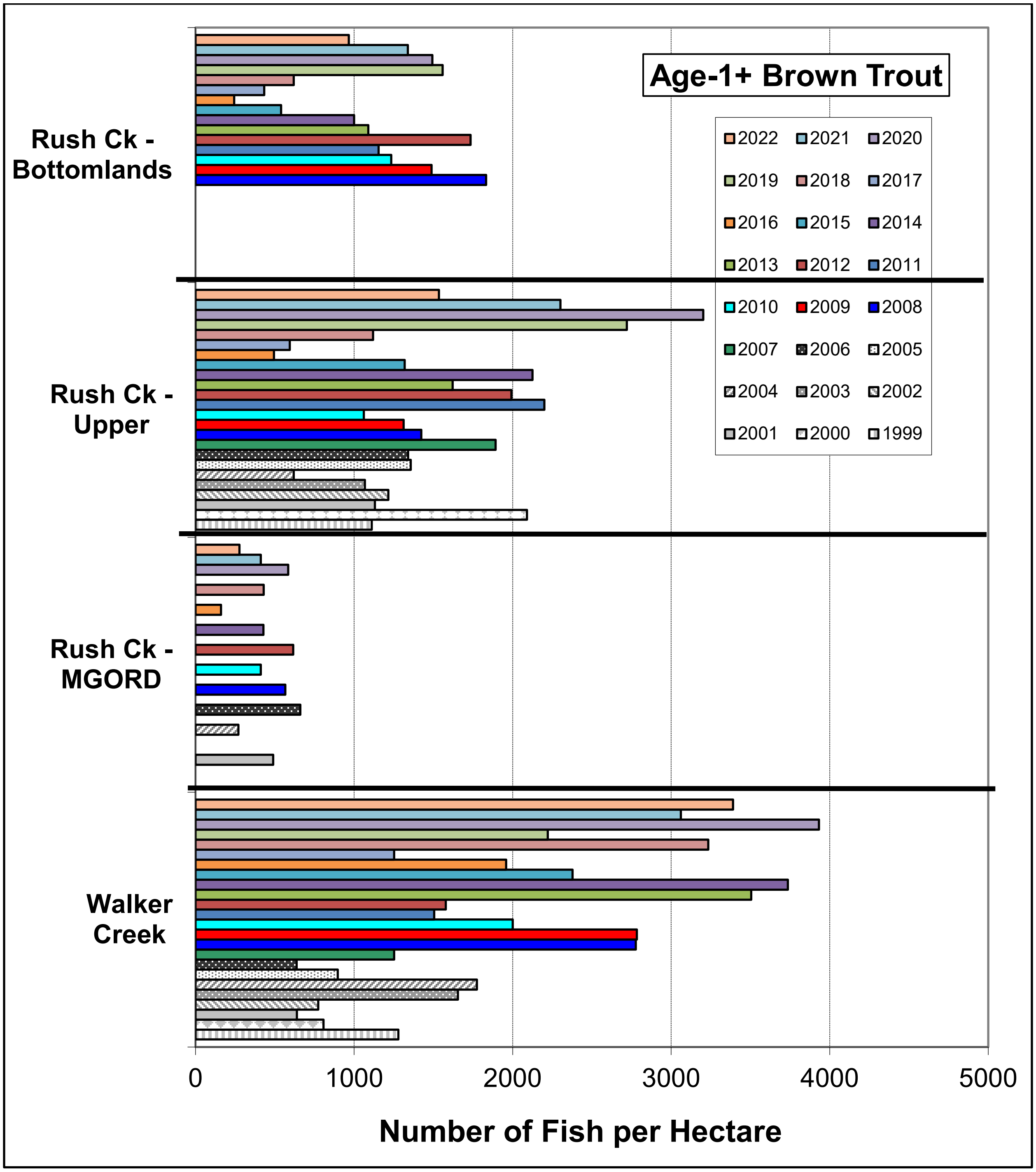


Figure 16. Estimated number of age-1 and older Brown Trout per hectare in sections of Rush and Walker Creeks from 1999 to 2022.

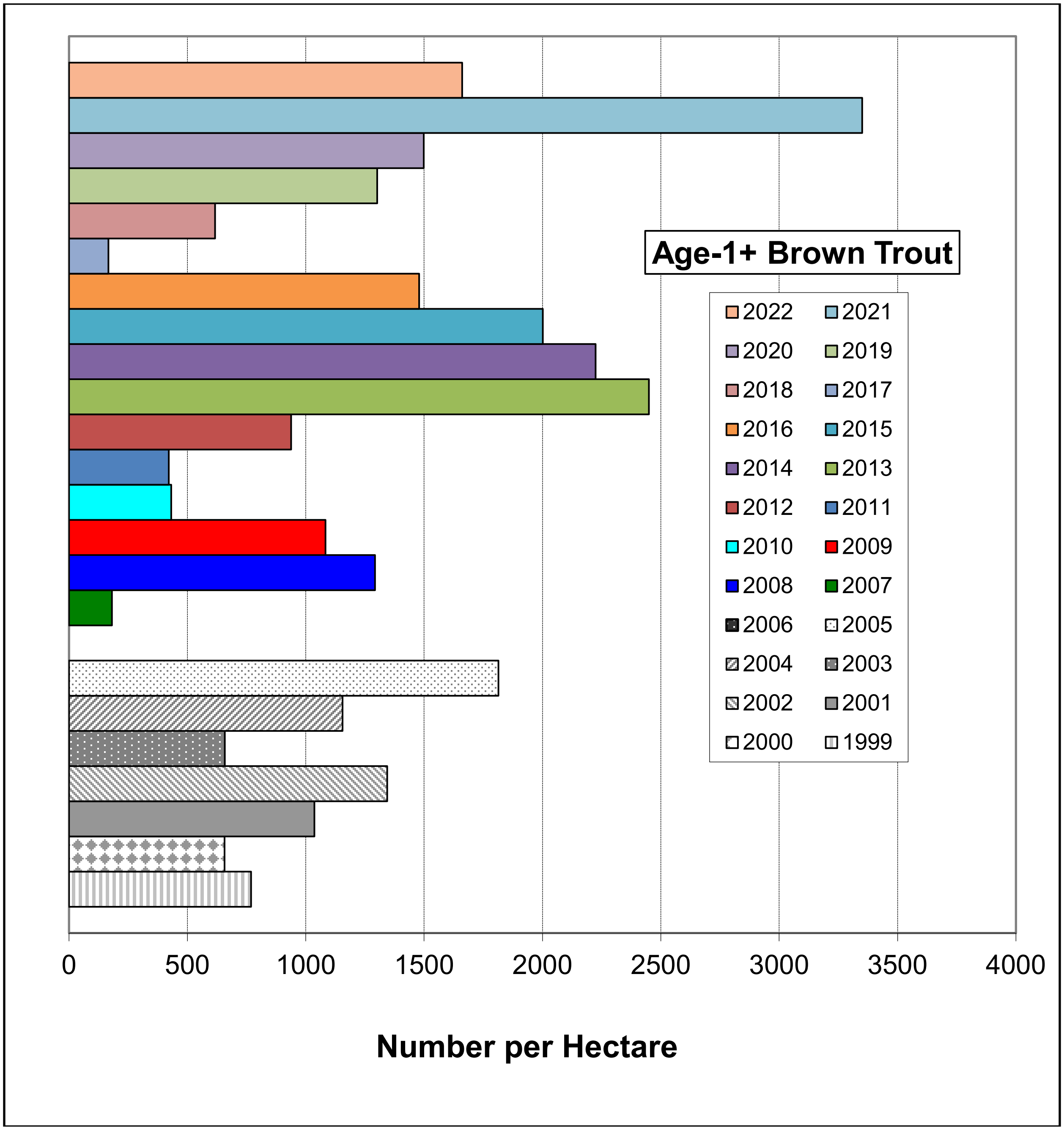


Figure 17. Estimated number of age-1 and older Brown Trout per hectare in the Lee Vining Creek main channel from 1999 to 2022.

Age-0 Rainbow Trout

In the Lee Vining Creek main channel, only one age-0 Rainbow Trout was captured during the 2022 sampling.

The Upper Rush section supported an estimated density of 430 age-0 Rainbow Trout/ha in 2022, an increase of 58% from the 2021 estimate of 273 age-0 Rainbow Trout/ha.

Age-1 and older (aka Age-1+) Rainbow Trout

In 2022, only one age-1 and older Rainbow Trout was captured in the Lee Vining Creek main channel.

The Upper Rush section supported an estimated density of 128 age-1+ Rainbow Trout/ha in 2028, an increase of 6% from the 2021 estimate of 121 age-1+ Rainbow Trout/ha.

Estimated Trout Standing Crops (kg/ha)

The total (Brown and Rainbow Trout) estimated standing crop in the Upper Rush section was 162 kg/ha in 2022, a 17% increase from 138 kg/ha in 2021 (Table 6 and Figure 18). Rainbow Trout comprised 15.9 kg/ha of the 2022 standing crop estimate (Figure 18). For the Upper Rush section, the 24-year average standing crop of Brown and Rainbow Trout equaled 156 kg/ha.

The estimated standing crop for Brown Trout in the Bottomlands section of Rush Creek was 75 kg/ha in 2022, a 4% decrease from 78 kg/ha in 2021 (Table 6 and Figure 19). For the Bottomlands section of Rush Creek, the 15-year average standing crop of Brown Trout equaled 81 kg/ha.

The estimated standing crop for Brown Trout in the MGORD section of Rush Creek was 54 kg/ha in 2022, a 19% decrease from 67 kg/ha in 2021 (Figure 18). For the 12 seasons where Brown Trout standing crop estimates were generated for the MGORD; the average value equaled 83 kg/ha.

The estimated standing crop for Brown Trout in Walker Creek was 160 kg/ha in 2022, a 1% increase from the 2021 estimate of 158 kg/ha (Table 6 and Figure 19). For Walker Creek, the 24-year average standing crop of Brown Trout equaled 145 kg/ha.

The estimated total standing crop for Brown Trout in the Lee Vining Creek main channel in 2022 was 136 kg/ha; a decrease of 7% from the 2021 estimate of 146 kg/ha (Table 7 and Figure 20). The long-term average for the 23-year sampling period is 125 kg/ha.

Table 6. Comparison of Brown Trout standing crop (kg/ha) estimates between 2017 and 2022 for Rush Creek sections and Walker Creek section. These six years include one extremely wet RY 2017, followed by the normal RY 2018, the wet RY 2019, dry-normal-1 RY 2020, dry RY 2021 and dry RY 2022.

Collection Location	2017 Total Standing Crop (kg/ha)	2018 Total Standing Crop (kg/ha)	2019 Total Standing Crop (kg/ha)	2020 Total Standing Crop (kg/ha)	2021 Total Standing Crop (kg/ha)	2022 Total Standing Crop (kg/ha)	Percent Change Between 2021 and 2022
Rush Creek – MGORD	N/A	95	N/A	81	67	54	-19%
Rush Creek – Upper	123	188*	291**	195***	138#	162##	+17%
Rush Creek - Bottomlands	50	103	91	84	78	75	-4%
Walker Creek	85	245	179	240	158	160	+1%

*Includes 18.7 kg/ha of Rainbow Trout **includes 36.5 kg/ha of Rainbow Trout ***Includes 24.4 kg/ha of Rainbow Trout
#Includes 10.8 kg/ha of Rainbow Trout ##Includes 15.9 kg/ha of Rainbow Trout

Table 7. Comparison of total (Brown and Rainbow Trout) standing crop (kg/ha) estimates between 2017 and 2022 for the Lee Vining Creek main channel section. These six years include one extremely wet RY 2017, followed by the normal RY 2018, the wet RY 2019, dry-normal-1 RY 2020, dry RY 2021 and dry RY 2022. The Rainbow Trout portion of the main channel's total estimated biomass is provided within the parentheses.

Collection Location	2017 Total Standing Crop (kg/ha)	2018 Total Standing Crop (kg/ha)	2019 Total Standing Crop (kg/ha)	2020 Total Standing Crop (kg/ha)	2021 Total Standing Crop (kg/ha)	2022 Total Standing Crop (kg/ha)	Percent Change Between 2021 and 2022
Lee Vining Creek - Main Channel	21 (0)	70 (0)	192 (4.6)	96 (0.6)	146 (0)	136 (0)	-7%

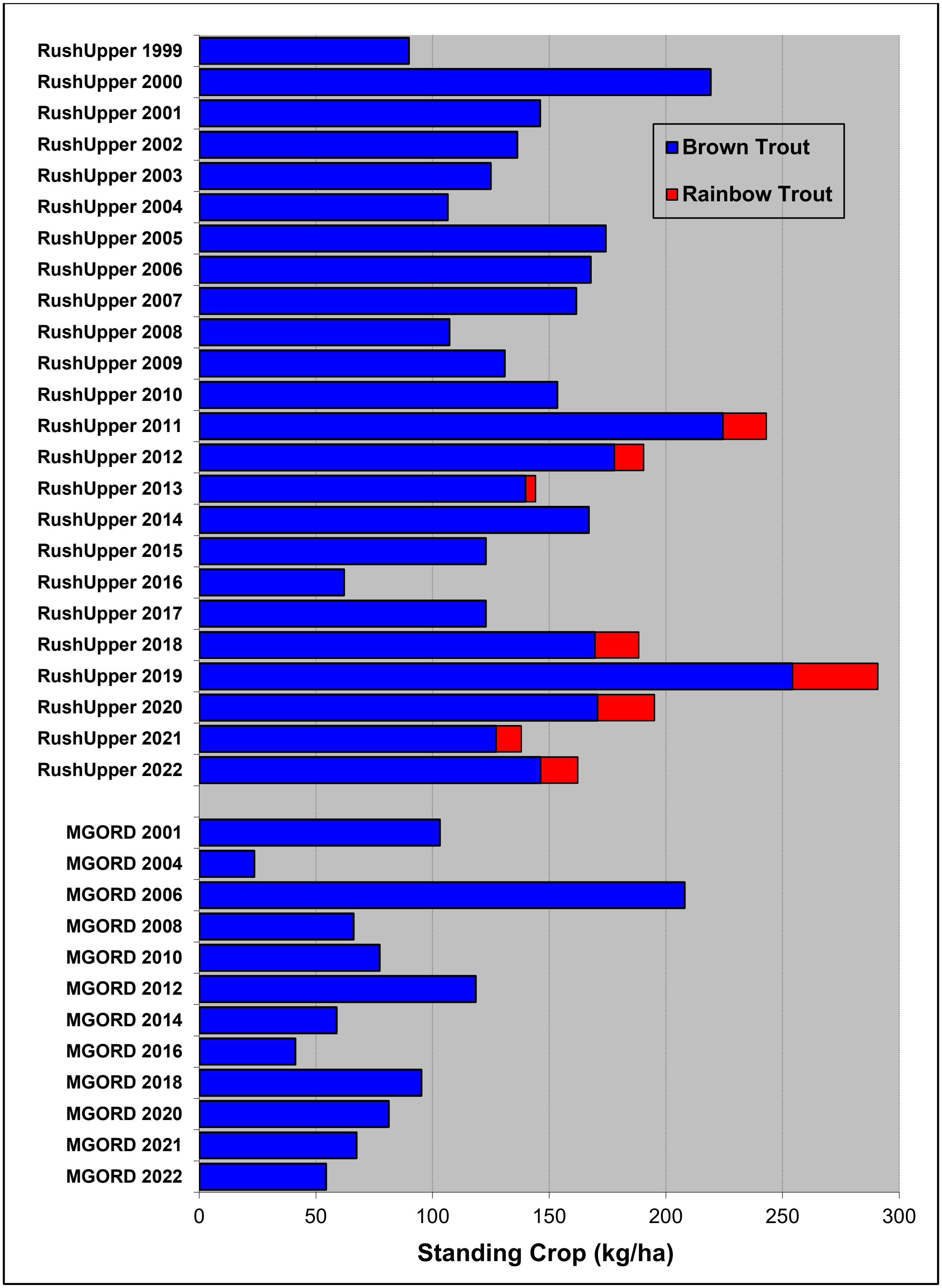


Figure 18. Estimated total standing crop (kilograms per hectare) of Brown Trout and Rainbow Trout in the Upper and MGORD sample sections of Rush Creek from 1999 to 2022.

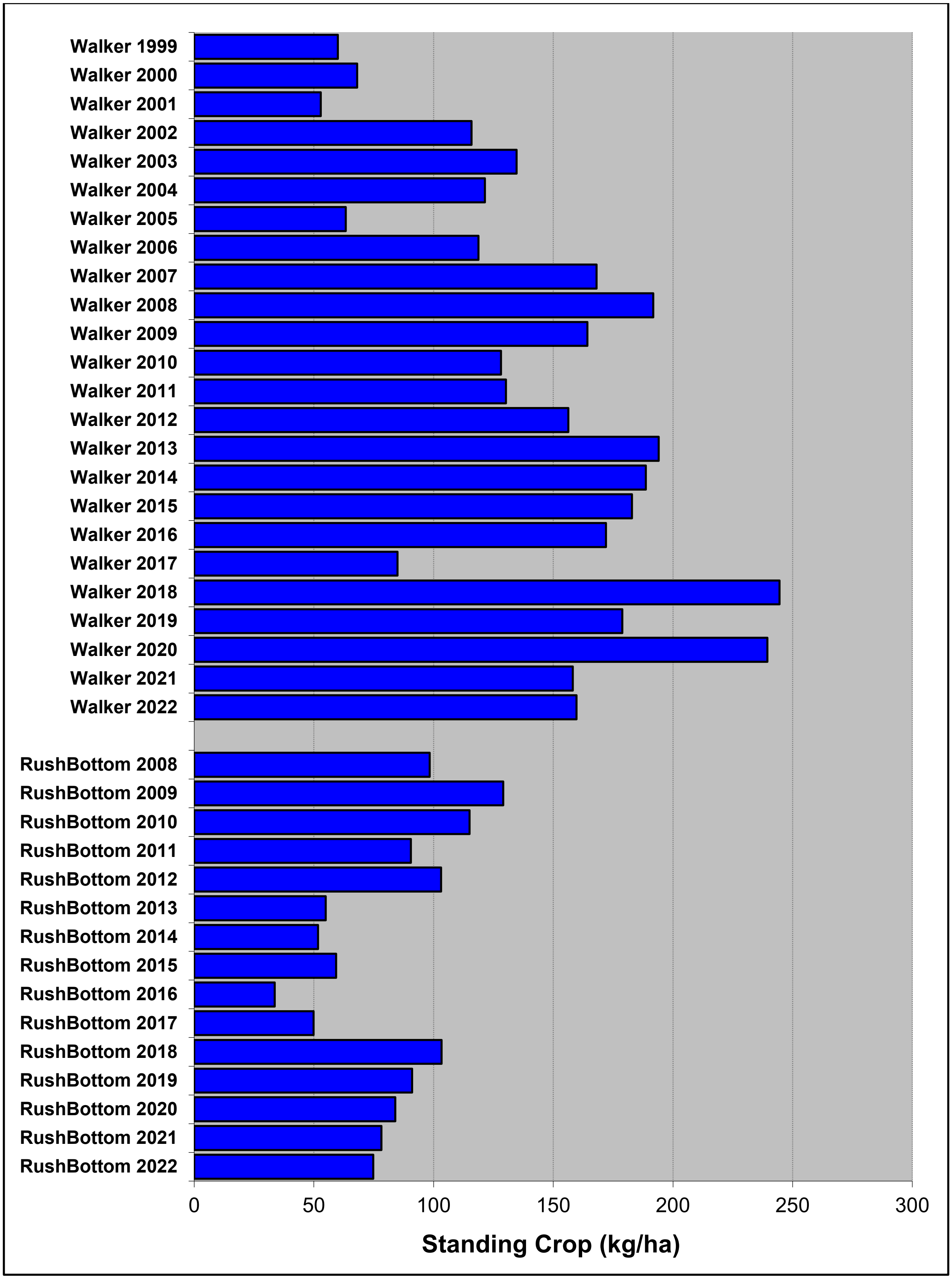


Figure 19. Estimated total standing crop (kilograms per hectare) of Brown Trout in Walker Creek and the Bottomlands sections of Rush Creek from 1999 to 2022.

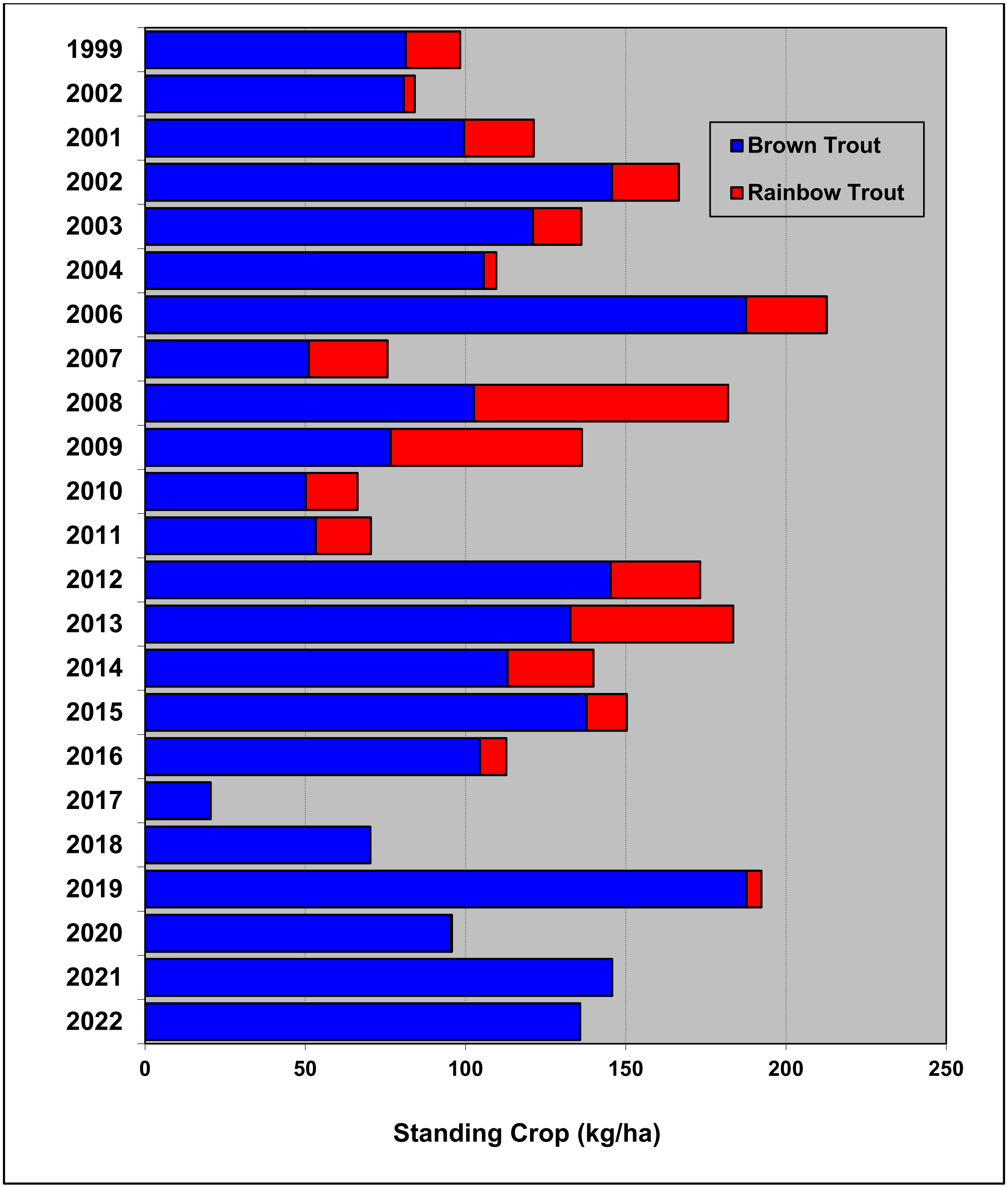


Figure 20. Estimated total standing crop (kilograms per hectare) of Brown Trout and Rainbow Trout in the Lee Vining Creek main channel sample section from 1999 to 2022.

Relative Stock Density (RSD) for Rush and Lee Vining Creeks

In the Upper Rush section, the RSD-225 equaled 17 for 2022, the first increase in RSD-225 after four straight years of large drops from the record RSD-225 value of 78 for 2017 (Table 8). The 2022 RSD-225 value was influenced by greater numbers of fish between 225-299 mm than the previous two years (Table 8). The RSD-300 value was 1 in 2022, the same as in 2021 and 2020 (Table 8). This low RSD-300 value was influenced by continued low numbers of Brown Trout >300 mm captured in 2022 (Table 8). Over 23 sampling years, a total of 155 Brown Trout ≥300 mm were captured in the Upper Rush Creek section, an average of 6.7 fish ≥300 mm per year (Table 8).

In the Bottomlands section of Rush Creek, the RSD-225 for 2022 equaled 15, the first substaintal increase since 2018 (Table 8). As in the Upper Rush section, the Bottomlands 2022 RSD-225 value was most likely influenced by an increase in the numbers of fish between 225-299 mm. The RSD-300 value was 0 in 2022, because no Brown Trout ≥300 mm were captured in the Bottomlands section (Table 8). Over the 15 sampling years, a total of 27 Brown Trout ≥300 mm were captured in the Bottomlands section, an average of 1.8 fish ≥300 mm per year (Table 8).

In the MGORD, the RSD-225 value has increased from 47 in 2019 to 48 in 2020 to 53 in 2021 and to 72 in 2022 (Table 8). In 2022, the RSD-300 value was 14, the highest RSD-300 value since 2018 (Table 8). The RSD-375 value equaled 3 in 2022, the same as in 2021 (Table 8). The two-pass catch of Brown Trout ≥150 mm in the MGORD during the 2022 season was 198 fish, which included 28 fish ≥300 mm in length and six of these fish were ≥375 mm in length (Table 8). For sampling conducted between 2001 and 2012, the annual average catch of Brown Trout ≥300 mm equaled 180 fish/year; then for the past 10 sampling years the annual average catch of Brown Trout ≥300 mm equaled 39 fish/year (Table 8). This 78% decline in larger Brown Trout coincided with the five years of drier RY's and poor summer thermal regimes within the MGORD in 2012-2016. However, in the six seasons following the five-year drought, the recruitment of larger, older fish appears to be a relatively slow process, possibly because for the past three years (2020-2022) summer water temperatures have generally been unfavorable for Brown Trout growth and survival in the MGORD (Table 8).

RSD values in Lee Vining Creek were generated for the main channel only (Table 9). The RSD-225 value for main channel increased substantially from 3 in 2021 to 19 in 2022, the largest RSD-225 value since 2018 (Table 9). In 2022, no Brown Trout greater than 300 mm in length were captured in Lee Vining Creek main channel, thus the RSD-300 value was 0, for the third consecutive year (Table 9).

Table 8. RSD values for Brown Trout in Rush Creek sections from 2000 to 2022.

Sampling Location Rush Creek	Sample Year	Number of Trout ≥150 mm	Number of Trout 150-224 mm	Number of Trout 225-299 mm	Number of Trout 300-374 mm	Number of Trout ≥375 mm	RSD-225	RSD-300	RSD-375
Upper Rush	**2022**	**235**	**196**	**37**	**2**	**0**	**17**	**1**	**0**
Upper Rush	2021	274	257	13	4	0	6	1	0
Upper Rush	2020	148	129	18	1	0	13	1	0
Upper Rush	2019	503	406	85	11	1	19	2	0
Upper Rush	2018	254	155	75	24	0	39	9	0
Upper Rush	2017	130	28	82	19	1	78	15	1
Upper Rush	2016	103	74	26	1	2	28	3	2
Upper Rush	2015	289	246	41	0	2	15	1	1
Upper Rush	2014	366	331	31	4	0	10	1	0
Upper Rush	2013	336	288	45	3	0	14	1	0
Upper Rush	2012	354	284	66	3	1	20	1	0
Upper Rush	2011	498	381	110	6	1	23	1	0
Upper Rush	2010	308	202	97	7	2	34	3	1
Upper Rush	2009	372	322	43	5	2	13	2	1
Upper Rush	2008	227	189	31	6	1	17	3	0
Upper Rush	2007	282	210	61	9	2	26	4	1
Upper Rush	2006	233	154	69	10	0	34	4	0
Upper Rush	2005	202	139	56	5	2	31	3	1
Upper Rush	2004	179	112	64	2	1	37	2	1
Upper Rush	2003	264	216	45	2	1	18	1	0
Upper Rush	2002	220	181	35	1	2	18	2	1
Upper Rush	2001	223	190	27	6	0	15	3	0
Upper Rush	2000	182	158	22	2	0	13	1	0
Bottomlands	**2022**	**145**	**123**	**22**	**0**	**0**	**15**	**0**	**0**
Bottomlands	2021	121	110	10	1	0	9	1	0
Bottomlands	2020	128	117	11	0	0	9	0	0
Bottomlands	2019	220	202	17	1	0	8	0	0
Bottomlands	2018	140	90	41	9	0	36	6	0
Bottomlands	2017	82	29	49	4	0	65	5	0
Bottomlands	2016	66	52	11	1	2	21	5	3
Bottomlands	2015	115	88	26	0	1	23	1	1
Bottomlands	2014	154	152	1	0	1	1	1	1
Bottomlands	2013	128	123	5	0	0	4	0	0
Bottomlands	2012	325	290	34	1	0	11	0	0
Bottomlands	2011	267	218	46	3	0	18	1	0
Bottomlands	2010	307	225	81	1	0	27	0	0
Bottomlands	2009	379	321	56	1	1	15	1	0
Bottomlands	2008	160	141	19	0	0	12	0	0

Table 8 (continued).

Sampling Location Rush Creek	Sample Year	Number of Trout ≥150 mm	Number of Trout 150-224 mm	Number of Trout 225-299 mm	Number of Trout 300-374 mm	Number of Trout ≥375 mm	RSD-225	RSD-300	RSD-375
MGORD	**2022**	**198**	**56**	**114**	**22**	**6**	**72**	**14**	**3**
MGORD	2021	431	204	180	35	12	53	11	3
MGORD	2020	322	167	112	37	6	48	13	2
MGORD	2019	275	145	102	24	4	47	10	1
MGORD	2018	326	98	162	51	15	70	20	5
MGORD	2017	104	12	64	17	11	88	27	11
MGORD	2016	179	46	95	18	20	74	21	11
MGORD	2015	116	33	54	20	9	72	25	8
MGORD	2014	388	184	175	19	10	53	7	3
MGORD	2013	411	237	118	41	15	42	14	4
MGORD	2012	694	176	319	173	26	75	29	4
MGORD	2011	216	36	117	55	8	83	29	4
MGORD	2010	694	252	292	115	35	64	22	5
MGORD	2009	643	156	338	123	26	76	23	4
MGORD	2008	856	415	301	118	22	52	16	3
MGORD	2007	621	144	191	259	27	77	46	4
MGORD	2006	567	60	200	280	27	89	54	5
MGORD	2004	424	130	197	64	33	69	23	8
MGORD	2001	774	330	217	119	108	57	29	14

Table 9. RSD values for Brown Trout in the Lee Vining Creek main channel section from 2000-2022.

Sampling Location Rush Creek	Sample Year	Number of Trout ≥150 mm	Number of Trout 150-224 mm	Number of Trout 225-299 mm	Number of Trout 300-374 mm	Number of Trout ≥375 mm	RSD-225	RSD-300
LV Main	**2022**	**129**	**105**	**24**	**0**	**0**	**19**	**0**
LV Main	2021	175	169	6	0	0	3	0
LV Main	2020	80	69	11	0	0	14	0
LV Main	2019	131	107	22	2	0	18	2
LV Main	2018	51	39	10	2	0	24	4
LV Main	2017	23	17	5	1	0	26	4
LV Main	2016	169	145	24	0	0	14	0
LV Main	2015	210	192	18	0	0	9	0
LV Main	2014	200	173	27	0	0	14	0
LV Main	2013	325	308	16	1	0	5	0
LV Main	2012	111	72	37	2	0	35	2
LV Main	2011	60	31	23	5	1	48	10

Table 9 (continued).

LV Main	2010	62	28	32	2	0	55	3
LV Main	2009	137	106	30	1	0	23	1
LV Main	2008	149	138	11	0	0	7	0
LV Main	2007	29	24	5	0	0	17	0
LV Main	2006	Not sampled in 2006 due to unsafe high flows						
LV Main	2005	60	37	20	2	1	38	5
LV Main	2004	70	60	8	2	0	14	3
LV Main	2003	52	27	23	2	0	48	4
LV Main	2002	100	74	23	3	0	26	3
LV Main	2001	90	71	16	3	0	21	3
LV Main	2000	51	32	18	1	0	37	2

PIT Tag Recaptures

PIT Tags Implanted between 2009 and 2022

Between 2009 and 2022, a total of 11,378 PIT tags were implanted in Brown Trout and Rainbow Trout within the annually sampled sections of Rush, Lee Vining and Walker Creeks. All PIT tagged fish received adipose fin clips. The numbers of PIT tags implanted each year varied according to fish availability and inventory of PIT tags, with year-specific information for 2009 through 2021 tabulated in Appendix B.

In 2022, a total of 697 trout received PIT tags and adipose fin clips in Rush, Lee Vining, and Walker Creeks (Table 10). In addition, five recaptured adipose fin-clipped fish had shed their original tags and were re-tagged, thus a total of 702 PIT tags were implanted during the 2022 fisheries sampling (Table 10). Of the 702 trout tagged, 610 were age-0 Brown Trout and 54 were age-1 and older Brown Trout (Table 10). For Rainbow Trout, 37 age-0 fish and one older fish were tagged (Table 10). Forty-nine of the age-1+ Brown Trout tagged in the MGORD section were up to 225 mm in total length and were presumed to be age-1 fish (Table 10). In addition, 26 age-0 Brown Trout were tagged in the MGORD (Table 10). Tagged and recaptured fish provided empirical information to estimate annual fish growth, tag retention, fish movements, and apparent survival rates.

Table 10. Total numbers of trout implanted with PIT tags during the 2022 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	225	4*	35	0	**264 Trout**
	Bottomlands	225	0	0	0	**225 Trout**
	MGORD	26	49**	1	1	**77 Trout**
Lee Vining Creek	Main Channel	30	1*	1	0	**32 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	104	0	0	0	**104 Trout**
Age Class Sub-totals:		**610**	**54**	**37**	**1**	**Total Trout: 702**

*shed tag/new tag implanted **up to 225 mm in total length

In September of 2022, a total of 107 previously tagged trout (that retained their tags) were recaptured in the Rush Creek watershed (Appendix C). Twenty-three of the recaptures occurred in the Bottomlands section, followed by 22 recaptures in the Upper Rush section (including one Rainbow Trout), 54 recaptures in Walker Creek, and eight recaptures in the MGORD (Appendix C). In September of 2022, a total of 18 previously tagged Brown Trout (that retained their tags) were recaptured in the Lee Vining Creek main channel section (Appendix C). During the 2022 sampling, only one previously tagged Rainbow Trout was recaptured, thus very limited growth rate information was available for Rainbow Trout in Rush Creek and none was available for Lee Vining Creek.

In the following text, growth between 2021 and 2022 will be referred to as 2022 growth rates. A 2022 trout refers to a fish recaptured in September of 2022. An age of a PIT tagged trout reflects its age during the sampling year. For instance, an age-1 trout in 2022 indicates that a trout was tagged in September 2021 at age-0 and its length and weight were remeasured in September 2022 when it was recaptured.

Also note there is a separate results section for reporting growth rates of recaptures from the MGORD section of Rush Creek, primarily because most of these fish were tagged at presumed age-1 (based on lengths up to 225 mm), instead of at a known age-0. When captured in the MGORD, age-0 trout were also implanted with PIT tags.

Growth of Age-1 Brown Trout between 2021 and 2022

In 2022, a total of 77 known age-1 Brown Trout were recaptured that were tagged as age-0 fish in 2021, for an overall recapture rate of 13.7% (77/561 age-0 fish tagged in 2021). Of the 77 age-1 recaptures; 39 of these fish were from Rush Creek sections, 30 fish were from Walker Creek and eight fish were from the Lee Vining Creek main channel section. Thus, by creek, the age-1 recapture rates for 2022 were 15% in Lee Vining Creek (22% in 2021, 14% in 2020, 23% in 2019, 29% in 2018 and 2% in 2017), 11% in Rush Creek (7% in 2021, 6% in 2020, 7% in 2019, 14% in 2018, 19% in 2017 and 5% in 2016), and 25% in Walker Creek (28% in 2021, 45% in 2020 and 19% in 2019). These recapture rates suggest survival between age-0 and age-1 in Rush and Walker Creeks in 2022 remained somewhat comparable to the previous year, and that survival rates in Lee Vining Creek in 2022 decreased from the previous year.

In the Upper Rush section, 16 age-1 Brown Trout were recaptured in 2022 and the average growth rates of these trout were 81 mm and 41 g (Table 11). Compared to 2021 rates, the average growth rates of the 16 age-1 Brown Trout were higher by 15 mm and 14 g (Table 11). Growth rates of age-1 Brown Trout in the Upper Rush section had generally declined annually from 2010 to 2014, but the 2015-2017 growth rates increased each year, with the 2017 growth rates the largest recorded for this section (Table 11). After the 2017 season, growth rates of age-1 Brown Trout in Upper Rush remained relatively low, the 2020 and 2021 average growth rates for age-1 Brown Trout in Upper Rush were the two lowest rates recorded for the 12 years of available data (Table 11). However, since 2020, growth rates of age-1 Brown Trout in Upper Rush have increased for two consecutive years (Table 11).

In the Bottomlands section of Rush Creek, 21 age-1 Brown Trout were recaptured in 2022 and the average growth rates of these trout were 80 mm and 41 g (Table 11). Compared to 2021 rates, the growth rates of the 21 age-1 Brown Trout were higher by 13 mm and by 15 g (Table 11). In terms of weight, the average growth rates for age-1 Brown Trout in the Bottomlands had dropped for four consecutive years, 2018-2021, before this 15 g increase in 2022 (Table 11).

In Walker Creek, 30 age-1 Brown Trout were recaptured in 2022 and the average growth rates of these 30 trout were 53 mm and 18 g; an increase of 6 mm in length from the 2021 average growth rate (Table 11). The 2022 average growth rate in weight was the same as in 2021 and this 18 g average remains the lowest value for the 12 years of available data (Table 11). The growth rates of age-1 Brown Trout in Walker Creek have typically been lower than the age-1 growth rates documented in Rush and Lee Vining Creeks (Table 11).

In Lee Vining Creek, eight age-1 Brown Trout were recaptured in 2022 and the average growth rates of these trout were 73 mm and 33 g (Table 11). Compared to 2021 rates, the growth rates of the 22 age-1 Brown Trout were higher by 10 mm in length and by 6 g in weight (Table 11). The 2022 growth rates (in weight) of age-1 Brown Trout in Lee Vining Creek were the first increase from the previous year after four straight years of declining growth rates after the record high rates documented in 2017 (Table 11).

Growth of Age-2 Brown Trout between 2021 and 2022

In 2022, a total of 22 known age-2 Brown Trout were recaptured that were tagged as age-0 fish in 2020, for a recapture rate of 3.8% (17/581 age-0 fish tagged in 2020). Seven of these fish were recaptured in Rush Creek (five from the MGORD), 11 of these fish were recaptured in Walker Creek, and four fish were recaptured in Lee Vining Creek.

In the Upper Rush section, one age-2 fish was recaptured in 2022 that had been tagged as age-0 fish in 2020 (Table 11). Between age-1 and age-2, the average growth rates of this Brown Trout were 68 mm and 67 g (Table 11). Compared to 2021 rates, the growth rates of this one age-2 Brown Trout were higher by 14 mm and by 15 g (Table 11).

In the Bottomlands section of Rush Creek, one previously tagged age-2 Brown Trout was recaptured in 2022. Between age-1 and age-2, this fish had grown by 81 mm and gained 84 g in weight (Table 11). By weight, this was the highest growth rate recorded for an age-2 Brown Trout in the Bottomlands section for the 10 years of growth data on age-2 fish (Table 11).

In Walker Creek, 11 age-2 fish were recaptured in 2022 that had been tagged as age-0 fish in 2020 (Table 11). Between age-1 and age-2, the average growth rates of these 11 Brown Trout were 37 mm and 23 g (Table 11). The 2022 average growth rate, by weight, of age-2 Brown Trout in Walker Creek was the second lowest recorded for the 11 years of available data (Table 11).

In the Lee Vining Creek main channel section, four age-2 Brown Trout were recaptured in 2022 that had been tagged as age-0 fish in 2020. Between age-1 and age-2, the growth rates of these four Brown Trout were 50 mm and 54 g, a 4 mm and 7 g increase in growth rate from the previous year (Table 11).

Growth of Age-3 Brown Trout between 2021 and 2022

In 2022, seven known age-3 Brown Trout were recaptured in Walker Creek section that were tagged as age-0 fish in 2019; six of these fish were also recaptured each year since their intitial tagging. Between age-2 and age-3, the average growth rates of these seven Brown Trout were 22 mm and 18 g, versus average growth rates of 12 mm and 19 g of four age-3 fish in 2021 (Table 11).

Growth of Age-4 Brown Trout between 2021 and 2022

In 2022, two known age-4 Brown Trout were recaptured in Walker Creek that were tagged as age-0 fish in 2018. Between age-3 and age-4, the growth rates of these two Brown Trout were 12 mm and -5 g (one fish lost 10 g and one fish weighed the same each year). One of these fish (tag #989001028114180) has been recaptured every year since its initial tagging in 2018.

An age-4 Brown Trout was also recaptured in Lee Vining Creek in 2022 and between age-3 and age-4 this fish grew by 31 mm and 67 g (Table 11). This fish was also recaptured each year since being tagged in 2018 at age-0. The last recapture of a tagged age-4 Brown Trout in Lee Vining Creek occurred in September of 2016 (Table 11).

Growth of Age-5 Brown Trout between 2021 and 2022

In 2022, one known age-5 Brown Trout was recaptured in the Upper Rush section that was tagged as age-0 fish in 2017 (and was also captured in 2021 at age-4) (Table 11). Between 2021 and 2022, this age-5 Brown Trout grew by 15 mm in length and lost 49 g in weight (Table 11). This fish was also recaptured at age-2, age-3, and age-4. Between age-3 and age-4, this fish grew by grew by 38 mm in length and by 144 g in weight. This was the first PIT-tagged age-5 fish ever recaptured in the Upper Rush section, for 14 years of documenting annual growth (Table 11).

Growth of MGORD Brown Trout between 2020 and 2021

Starting in September of 2017, PIT tagging of Brown Trout in the MGORD section of Rush Creek was focused on known age-0 fish and presumed age-1 fish. Based on past years' length-frequency histograms and growth rates of know age-1 fish (from recaptures of previously tagged age-0 fish), a cut-off of 225 mm in total length was used to define the probable upper limit for age-1 Brown Trout in the MGORD. Thus, moving forward, most recaptures of previously tagged fish within the MGORD will allow us to compute annual growth rates of fish in which their ages are known or likely accurately presumed.

In 2022, two age-1 Brown Trout were captured in the MGORD that were tagged at age-0 in 2021; both of these fish were tagged in the MGORD. Between 2021 and 2022, the average growth rates of these two fish were 123 mm and 100 g, compared to average growth rates of 100 mm and 68 g in 2021. At age-1, both these fish had total lengths of 220 mm. In weight, the growth rate of age-1 Brown Trout in the MGORD was more than two times greater than the age-1 growth rate in the Upper Rush section in both 2021 and 2022 (Table 11).

In 2022, five Brown Trout were recaptured that were tagged as presumed age-1 fish in 2021 and these five presumed age-2 fish had average growth rates of 86 mm and 167 g. The presumed age-2 fish recaptured in 2022 ranged from 276 mm to 313 mm in total length.

In 2022, one Brown Trout was recaptured in the MGORD that had been PIT tagged in the MGORD as presumed age-1 fish in 2020 and was also recaptured as presumed age-2 fish in the MGORD in 2021. Between age-2 and age-3, the average growth rates of this fish were 60 mm and 222 g.

Movement of PIT Tagged Trout between Sections

Previous annual fisheries reports have summarized documented movements of PIT tagged fish between the sample sections, with most movements occurring from the Upper Rush section upstream into the MGORD (Taylor 2021). These movements between the Upper Rush section and MGORD were initially documented during the radio telemetry study when approximately 50% of the tagged fish left the MGORD during the fall/early winter spawning period (Taylor et al. 2009). However, in 2022 none of the recaptured PIT tagged fish had moved from another section.

PIT Tag Shed Rate of Trout Recaptured in 2022

In 2022, a total of 130 trout with adipose fin clips were recaptured and five of these fish failed to produce a PIT tag number when scanned with the tag reader (four from Upper Rush and one from Lee Vining Creek). Assuming that all these fish were previously PIT tagged, the 2022 calculated shed rate was 3.8% (5 shed tags/130 clipped fish recaptured) versus shed rates of 2.3% in 2021 and 6.8% in 2020. Retention rates tend to be higher in juvenile fish because adult salmonids are known to shed tags during spawning (Bateman et al. 2009). Also, tag retention rates have also been linked to tagger's experience and crew turnover rates, with less experienced taggers resulting in higher shed rates (Dare 2003). For the past nine years, our crew members implanting tags have remained relatively stable, however in 2022 the fisheries crew was comprised of mostly new individuals and some with little or no tagging experience.

Table 11. Average growth (length and weight) of Brown Trout recaptured from 2009 through 2022 by age. Note: *denotes only one PIT tagged fish recaptured. •denotes one fish that moved from Upper Rush to the MGORD.

Stream and Reach	Cohort	Average Annual Growth in Length and Weight (mm/g)													
		2008 - 2009	2009 - 2010	2010 - 2011	2011 - 2012	2012 - 2013	2013 - 2014	2014 - 2015	2015 - 2016	2016 - 2017	2017 - 2018	2018 - 2019	2019 - 2020	2020 - 2021	2021 - 2022
Upper Rush Creek	Age 1	89/51	81/50	83/48	72/33	67/35		90/55	105/77	132/129	83/56	77/43	55/21	66/27	81/41
	Age 2		58/70	54/73	43/42	41/42		64/69	99/176•	108/239	39/66	48/71	44/55	54/42	68/67*
	Age 3				14/29		24/41				11/40*	15/27*	41/49*		
	Age 4					12/-22								38/144*	
	Age-5														15/-49*
Rush Creek Bottom -lands	Age 1	84/43	77/40	71/35	58/25	56/24		84/41	94/62	118/96	72/42	74/38	64/29	67/26	80/41
	Age 2		50/54	35/32	30/28	27/22	32/29*	62/62			39/55	36/44*		35/33*	81/84*
	Age 3			13/14	17/16	11/9	35/31						21/20*		
	Age 4				4/-11		18/20								
	Age-5														
LV Main Channel Brown Trout	Age 1		80/42	72/37	99/52	61/27		73/33	74/40	110/92*	103/77	71/41	72/29	63/27	73/33
	Age 2		66/95		77/110	33/34	35/29	47/40	47/49	77/128*		60/91*	70/81	46/47	55/54
	Age 3			34/92		23/48*	16/20*	27/32	42/75					30/48	
	Age 4				21/41*				25/47*						31/67*
	Age-5														
LV Main Channel RB Trout	Age 1					78/47		80/35				80/43*			
	Age 2						40/48*	52/50	62/74*						
	Age 3								38/82*						
	Age 4														
	Age-5														
Walker Creek Above Old 395	Age 1	68/27	51/20	71/34	68/36	59/23		58/24	72/36	66/33		55/28	54/24	47/18	53/18
	Age 2		31/26	60/56	40/33	27/21	39/35		47/44	37/37	42/52		36/30	25/19	37/23
	Age 3			28/44	18/12	9/2	20/36	27/29		42/59*	25/37	25/37		12/19	22/18
	Age 4				7/2	2/-16*		28/45*			27/37*		8/-5		13/-5
	Age-5						0/-10*								

Comparison of Length-at Age amongst Sample Sections

During the September 2022 sampling, four age-classes of PIT tagged Brown Trout were recaptured within four fisheries monitoring sections of Rush, Walker and Lee Vining creeks (Tables 12 and 13). Along with providing age-specific length information for each section, these data allowed comparisons of length-at-age between sample sections and also between the years 2013-2022 (Tables 12 and 13).

In Upper Rush, the average length-at-age-1 in 2022 was 169 mm, 15 mm greater than the average length-at-age-1 in 2021, the first increase after four consecutive years of decreases (Table 12). In 2022, age-1 Brown Trout in Upper Rush were 3 mm longer than age-1 fish in the Bottomlands section (Table 12). In the Bottomlands section, the average length-at-age-1 in 2022 was 166 mm, 11 mm longer the 2021 average length-at-age-1, and the second lowest average value for the past eight years of available data (Table 12).

In Upper Rush, the average length-at-age-2 in 2022 was 228 mm, 30 mm longer than the average length-at-age-2 in 2021 (Table 12). For Upper Rush, this was the first increase in average length after four consecutive years of decreasing average length-at-age-2 (Table 12). In the Bottomlands section the average length-at-age-2 in 2022 was 219 mm, 33 mm longer than the average length-at-age-2 in 2021 (Table 12). In 2022, the average length-at-age-2 for Brown Trout in Upper Rush was 9 mm longer than age-2 fish within the Bottomlands section of Rush Creek (Table 12).

In 2022, no PIT-tagged age-3 Brown Trout were recaptured in either the Upper Rush section or the Bottomlands section of Rush Creek (Table 12). Also, no PIT-tagged age-4 Brown Trout were recaptured in these sections either (Table 12).

In 2022, one PIT-tagged age-5 Brown Trout was recaptured in Upper Rush section and this fish was 340 mm in total length (Table 12). The 2014 sampling season was the last time a PIT tagged age-5 Brown Trout was recaptured in Upper Rush and the 2016 season was the last time a PIT tagged age-5 Brown Trout was recaptured in the Bottomlands section (Table 12).

For Walker Creek in 2022, three age-1 Brown Trout were recaptured and the average length-at-age-1 was 140 mm, 2 mm longer than the average length-at-age-1 in 2021 (Table 12). In 2022, three of the known age-1 Brown Trout recaptured in Walker Creek were less than 125 mm in total length. In 2022, 11 PIT tagged age-2 Brown Trout were recaptured in Walker Creek and the average length-at-age-2 equaled 175 mm (same as in 2021) and also the lowest average value for the eight years of available data (Table 12). In 2022, eight PIT tagged age-3 Brown Trout were recaptured in Walker Creek and the average length-at-age-3 equaled 199 mm, 6 mm less than the 2021 average, and the lowest average value for the eight years of available data (Table 12). In 2022, two PIT tagged age-4 Brown Trout were recaptured in Walker Creek and the average length-at-age-4 was 213 mm, the lowest average value for the six years of available data (Table 12).

For the Lee Vining Creek main channel in 2022, eight age-1 Brown Trout were recaptured and the average length-at-age-1 for these Brown was 161 mm, 7 mm longer than in 2021 (Table 13). In 2022, five previously tagged age-2 Brown Trout were recaptured and the average length-at-age-2 equaled 208 mm, 13 mm longer than in 2021 (Table 13). In 2022, two age-3 Brown Trout were recaptured and the average length-at-age-3 equaled 236 mm, 10 mm less than in 2021 (Table 13). In 2022, one age-4 Brown Trout was recaptured in Lee Vining Creek, this fish was 277 mm in total length, and was the first age-4 recapture in Lee Vining since 2016 (Table 13).

These findings of average lengths by age-class continue to support the previous conclusions by the Stream Scientist that very few Brown Trout reach age-4 or older in Rush Creek or Lee Vining Creek (Taylor 2022). However, the growth rates that Brown Trout exhibited in 2017 and 2018 confirmed that some age-2 and age-3 fish were near or just above lengths of 300 mm, the size class approaching the metrics of the pre-1941 fishery. These growth rates appeared to have been a function of relatively low fish densities and mostly favorable summer water temperature conditions in 2017 and 2018. The MGORD section continues to be the only section where Brown Trout consistently approach or exceed 300 mm in total length by age-2 or age-3.

Table 12. Size range of PIT tagged recaptures in 2013-2022 by age class for Brown Trout at three electrofishing sections on Rush and Walker Creeks. NOTE: years omitted if no fish were recaptured.

Section	Cohort	Size Range (mm)	Average Length (mm)
Upper Rush	Age-1	**2022** = 151-189 **2021** = 126-185 **2020** = 124-167 **2019** = 128-202 **2018** = 158-232 **2017** = 224-264 **2016** = 192-237 **2015** = 169-203	**2022** = 169 **2021** = 154 **2020** = 145 **2019** = 173 **2018** = 193 **2017** = 243 **2016** = 208 **2015** = 187
	Age-2	**2022** = 217-237 **2021** = 174-233 **2020** = 209-235 **2019** =203-251 **2018** = 236-305 **2017** = 284-337 **2016** = 289* **2015** = 205-242	**2022** = 228 **2021** = 198 **2020** = 221 **2019** = 237 **2018** = 274 **2017** = 313 **2016** = 289* **2015** = 217
	Age-3	**2021** = 220 **2020** = 287 **2019** = 251 **2018** = 295 **2014** = 226-236 **2013** = 227-263	**2021** = 220 **2020** = 287 **2019** = 251 **2018** = 295 **2014** = 231 **2013** = 245
	Age-4	**2021** = 325 **2014** = 288 **2013** = 252-255	**2021** = 325 **2014** = 288 **2013** = 254
	Age-5	**2022** = 340 **2014** = 298	**2022** = 340 **2014** = 298
Bottomlands	Age-1	**2022** = 142-204 **2021** = 155 **2020** = 141-187 **2019** = 133-196 **2018** = 166-199 **2017** = 189-246 **2016** = 172-217 **2015** = 150-181	**2022** = 166 **2021** = 155 **2020** = 155 **2019** = 168 **2018** = 181 **2017** = 221 **2016** = 197 **2015** = 169
	Age-2	**2022** = 202-236 **2021** = 186 **2019** = 219 **2018** = 251-287 **2015** = 197-239 **2014** = 192 **2013** = 156-196	**2022** = 219 **2021** = 186 **2019** = 219 **2018** = 267 **2015** = 219 **2014** = 192 **2013** = 178
	Age-3	**2021** = 214-248 **2020** = 240 **2014** = 194 **2013** = 194-227	**2021** = 231 **2020** = 240 **2014** = 194 **2013** = 204
	Age-4	**2014** = 215-219	**2014** = 216
	Age-5	**2016** = 318	**2016** = 318

*Fish was tagged in Upper Rush, but moved to MGORD between age-1 and age-2.

Table 12 (continued).

Section	Cohort	Size Range (mm)	Average Length (mm)
Walker Creek	Age-1	**2022** = 114-169 **2021** = 121-154 **2020** = 132-170 **2019** = 141-168 **2017** = 151-179 **2016** = 145-187 **2015** = 133-177	**2022** = 140 **2021** = 138 **2020** = 151 **2019** = 159 **2017** = 166 **2016** = 167 **2015** = 154
	Age-2	**2022** = 151-205 **2021** = 155-187 **2020** = 190-196 **2018** = 191-221 **2017** = 180-224 **2016** = 180-226 **2014** = 168-200 **2013** = 181-208	**2022** = 175 **2021** = 175 **2020** = 194 **2018** = 210 **2017** = 202 **2016** = 201 **2014** = 186 **2013** = 197
	Age-3	**2022** = 180-215 **2021** = 200-212 **2019** = 215-235 **2018** = 204-245 **2017** = 238 **2015** = 211-231 **2014** = 207-222 **2013** = 219-221	**2022** = 199 **2021** = 205 **2019** = 220 **2018** = 228 **2017** = 238 **2015** = 219 **2014** = 217 **2013** = 220
	Age-4	**2022** =205-221 **2020** = 224-243 **2018** = 265 **2015** = 249 **2014** = 211 **2013** = 219	**2022** = 213 **2020** = 234 **2018** = 265 **2015** = 249 **2014** = 211 **2013** = 219
	Age-5	**2014** = 220	**2014** = 220

Table 13. Size range of PIT tagged fish recaptured in 2013-2022 by age class for Brown Trout and Rainbow Trout on Lee Vining Creek. NOTE: years omitted if no fish were recaptured.

Section	Cohort	Size Range (mm)	Average Length (mm)
Brown Trout in Lee Vining Main Channel	Age-1	**2022** = 145-169 **2021** = 126-182 **2020** = 125-185 **2019** = 142-209 **2018** = 170-194 **2017** = 210 **2016** = 147-186 **2015** = 149-190	**2022** = 161 **2021** = 154 **2020** = 155 **2019** = 174 **2018** = 183 **2017** = 210 **2016** = 171 **2015** = 166
	Age-2	**2022** = 183-230 **2021** = 163-225 **2020** = 212-270 **2019** = 222-274 **2017** = 247 **2016** = 205-217 **2015** = 176-214 **2014** = 174-195 **2013** = 206-225	**2022** = 208 **2021** = 195 **2020** = 232 **2019** = 247 **2017** = 247 **2016** = 211 **2015** = 197 **2014** = 188 **2013** = 215
	Age-3	**2022** =226-246 **2021** = 246 **2017** = 280-305 **2016** = 210-256 **2015** = 188-228 **2014** = 234-241 **2013** = 238-271	**2022** = 236 **2021** = 246 **2017** = 293 **2016** = 240 **2015** = 215 **2014** = 238 **2013** = 253
	Age-4	**2022** = 277 **2016** = 237	**2022** = 277 **2016** = 237
	Age-5	None captured in past eight years	
Rainbow Trout in Lee Vining Main Channel	Age-1	**2019** = 165 **2015** = 140-177	**2019** = 165 **2015** = 157
	Age-2	**2016** = 232 **2015** = 195-216 **2014** = 201-229	**2016** = 232 **2015** = 204 **2014** = 215
	Age-3	**2016** = 242	**2016** = 242
	Age-4	None captured in past eight years	
	Age-5	None captured in past eight years	

Apparent Survival Rates

PIT tag recaptures also allowed the computation of apparent survival rates of Brown Trout between age-0 and age-1. Apparent survival rates of age-1 Brown Trout were calculated with the following equation: [# age-1 recaps in 2022/capture probability of age-1 fish] ÷ [# age-0 tagged in 2021 - # shed tags]. For mark-recapture sections, capture probabilities were derived from the recapture run data: # of recaptures/# of captures. Compared to the 2021 survival rates; the 2022 apparent survival rates increased by 2.4% in Upper Rush Creek and increased by 15.2% in the Bottomlands section of Rush Creek (Table 14). Between 2021 and 2022, the age-1 Brown Trout apparent survival rate increased by 27.7% in the Lee Vining Creek main channel section (Table 14). Walker Creek's apparent survival rate increased by 8.2% between 2021 and 2022 (Table 14).

Table 14. Apparent survival rates of age-1 Brown Trout in Rush, Walker and Lee Vining Creeks.

Creek and Section	Capture Probability	No. Age-1 Recaps in 2022	No. Age-0 Tagged in 2021	No. Shed Tags	Apparent Survival Rate
Rush – Upper	0.19	16	148	2	2016 = 22.7% 2017 = 106% 2018 = 50.2% 2019 = 17.4% 2020 = 22.2% 2021 = 55.3% **2022 = 57.7%**
Rush - Bottomlands	0.13	2	65	0	2016 = 9.7% 2017 = 72.3% 2018 = 66.8% 2019 = 12.0% 2020 = 9.8% 2021 = 8.5% **2022 = 23.7%**
Walker Creek	0.67	26	92	1	2016 = 37.8% 2017 = 7.0% 2018 = N/A 2019 = 19.8% 2020 = 46.5% 2021 = 34.4% **2022 = 42.6%**
Lee Vining Creek	0.26	22	102	0	2016 = 46.3% 2017 = 4.8% 2018 = 70.6% 2019 = 40.0% 2020 = 27.4% 2021 = 55.3% **2022 = 83.0%**

Summer Water Temperature

During the past 11 years, the Mono Basin has experienced a five-year drought (2012-2016), a record Extreme-wet RY (2017), a Normal RY with a full GLR (2018), a Wet RY (2019), a Dry-normal-1 RY (2020), a Dry RY (2021), and a Dry RY in 2022. These RY types have resulted in a range of summer water temperatures in Rush Creek, from moderate-to-severe stressful conditions in drier RYs to thermal regimes mostly conducive to fair-to-good growth conditions in wetter RYs.

In 2022, a Dry RY with GLR storage levels either below or only 3.4 feet above the Synthesis Report recommended minimum summer storage elevation of 7,100 feet in July-September resulted in mostly unfavorable summer thermal conditions, with peak water temperatures >70°F at all six Rush Creek monitoring locations (Table 15). At all six of these monitoring locations, the numbers of days with water temperatures >70°F were either the highest or second-highest ever recorded at these stations (Table 15). In July and August, three of the temperature monitoring locations recorded peak temperatures >75°F. In 2022 daily mean temperatures and average daily maximum temperatures were either the highest or second-highest recorded at all Rush Creek temperature monitoring locations since these data were collected (Table 15).

Similar to the 2013-2021 annual reports, 2022 Rush Creek summer average daily water temperature data were classified based on its predicted influence on growth of Brown Trout as either: 1) good potential growth days, 2) fair potential growth days, 3) poor potential growth days (daily averages within one degree or less of a “bad thermal day”), or 4) bad thermal days (Table 16). Development of these thermal-based growth criteria were fully described in previous annual reports (Taylor 2013 and 2014). Using these growth prediction metrics, good potential growth days in 2022 varied from 12 to 33 days in Rush Creek out of the 92-day period from July 1 to September 30 (Table 16). For all Rush Creek monitoring locations, the number of days classified as “fair” potential growth days in 2022 ranged from 14 to 22 days (Table 16). In 2022, poor potential growth days and bad thermal days ranged from 35 days at Below Narrows to 59 days at Top of MGORD (Table 16). As in past years, the number of poor growth and bad thermal days in Rush Creek generally decreased in a downstream direction due to night-time cooling, which resulted in lower daily average temperatures (Table 16). However, these downstream temperature monitoring locations experienced more days with peak temperatures >70°F, higher peaks and much higher diurnal fluctuations, including extended periods of likely stressful diurnal fluctuations.

As was done with the 2013-2021 data, the diurnal temperature fluctuations for July, August and September 2022 were characterized by the one-day maximum fluctuation that occurred each month and by monthly averages (Table 17). Also, for each temperature monitoring location, the highest average diurnal fluctuations over consecutive 21-day durations were determined (Table 17). The diurnal fluctuations throughout the summer of 2022 were relatively low at the Top of MGORD and Bottom of MGORD temperature monitoring locations, but diurnal fluctuations increased at the downstream monitoring locations, most likely due to effects of daily warming and nightly cooling of air temperatures (Table 17). Over the 21-day durations, these larger diurnal fluctuations were above the threshold of 12.6°F considered detrimental to trout growth (Werley et al. 2007) during the summer of 2022 as recorded at the Above Parker, Below Narrows and County Road temperature monitoring locations (Table 17). These same three temperature monitoring locations also had 21-day durations with diurnal fluctuations exceeding 12.6°F during the summers of 2020 and 2021 (Table 17). July through August were when these temperature monitoring locations experienced their highest 21-day diurnal fluctuations, including levels detrimental to trout growth (Werley et al. 2007).

The thermal window bounded by 66.2-71.6°F where Brown Trout may be physiologically stressed and living at the edge of their survival tolerance as defined by Bell (2006) was quantified for each Rush Creek temperature monitoring location in 2013 through 2022. The hourly temperature data for the 92-day (or 2,208-hour) summer period were sorted from low to high and the number of hours where temperatures exceeded 66.2°F were summed by month and entire summer period (Table 18). The values from 2013 - 2021 were also included to better illustrate the variability that occurred at all the temperature monitoring locations (Table 18). The 2022 data show that all the temperature monitoring locations downstream of GLR experienced the greatest number of hours bounded by the 66.2-71.6°F thermal window, with levels exceeding those experienced during the recent five-year drought (Table 18). At the Top of MGORD, hourly water temperatures exceeded 66.2°F 60% of the time and at the five downstream monitoring locations, hourly water temperatures of 66.2°F were exceeded 28% to 47% of the 92-day period (Table 18). In 2022, for the temperature monitoring locations from the Top of MGORD to County Road, the month of August had the highest number of hours where temperatures exceeded 66.2°F (Table 18). For August, temperatures exceeding 66.2°F occurred for 98% of the month at the Top of MGORD monitoring location (Table 18).

In 2022, the water temperature monitoring locations Above Parker and Below Narrows continued to document cooler water accretions from Parker and Walker Creeks having a slight, yet positive, effect on Rush Creek's summer thermal regime, including a 12% decrease in the number of days with temperatures exceeding 70°F and 2% more good growth thermal days immediately downstream of the tributaries' accretions (Tables 15-18). However, the cooling effects of the Parker and Walker accretions were nonexistent at the County Road temperature monitoring location, where unfavorable summer water temperature metrics of the number of days >70°F increased and the largest diurnal fluctuations were documented.

Summer water temperatures in Lee Vining Creek were all within the range of fair-to-good growth potential during 2022. Regardless of water-year type, excessively warm water has not been an issue in Lee Vining Creek, thus detailed analyses were not performed with the 2022 data.

Table 15. Summary of water temperature data during the summer of RY 2022 (July to September). Averages were calculated for daily mean, daily minimum, and daily maximum temperatures between July 1st and September 30th. All temperature data are presented in °F. When available, values for 2013-2021 are provided for comparison.

Temperature Monitoring Location	Daily Mean (°F)	Ave Daily Minimum (°F)	Ave Daily Maximum (°F)	No. Days > 70°F	Max Diurnal Fluctuation (°F)	Date of Max. Fluctuation
Rush Ck. – Top of MGORD	2013 = 63.1	2013 = 62.7	2013 = 63.7	2013 = 0	2013 = 3.4	7/09/13
	2014 = 64.8	2014 = 64.6	2014 = 65.0	2014 = 0	2014 = 3.9	8/13/14
	2015 = 64.4	2015 = 64.1	2015 = 64.8	2015 = 0	2015 = 2.1	7/03/15
	2016 = 63.8	2016 = 63.0	2016 = 64.7	2016 = 0	2016 = 6.5	7/07/16
	2017 = 57.0	2017 = 56.5	2017 = 58.1	2017 = 0	2017 = 5.4	9/07/17
	2018 = 60.7	2018 = 59.6	2018 = 61.9	2018 = 0	2018 = 6.7	8/20/18
	2019 = 58.5	2019 = 57.2	2019 = 59.9	2019 = 0	2019 = 8.2	8/10/19
	2020 = 63.2	2020 = 62.1	2020 = 64.4	2020 = 0	2020 = 6.4	7/02/20
	2021 = 65.9	2021 = 65.2	2021 = 66.8	2021 = 5	2021 = 6.5	7/13/21
	2022 = 65.9	**2022 = 65.0**	**2022 = 67.0**	**2022 = 3**	**2022 = 6.6**	**7/12/22**
Rush Ck. – Bottom MGORD	2013 = 63.2	2013 = 60.9	2013 = 67.1	2013 = 1	2013 = 9.0	7/09/13
	2014 = 64.8	2014 = 62.9	2014 = 68.5	2014 = 20	2014 = 8.3	7/13/14
	2015 = 64.4	2015 = 62.3	2015 = 68.0	2015 = 20	2015 = 8.4	7/06/15
	2016 = 63.8	2016 = 61.8	2016 = 66.9	2016 = 1	2016 = 8.0	7/04/16
	2017 = 57.1	2017 = 56.5	2017 = 58.5	2017 = 0	2017 = 6.4	9/07/17
	2018 = 61.0	2018 =58.9	2018 = 63.9	2018 = 0	2018 = 8.7	7/05/18
	2019 = 58.7	2019 = 56.6	2019 = 61.3	2019 = 0	2019 = 8.1	8/10/19
	2020 = 63.2	2020 = 60.5	2020 = 67.5	2020 = 17	2020 = 10.0	8/03/20
	2021 = 65.8	2021 = 63.4	2021 = 69.8	2021 = 44	2021 = 8.5	7/24/21
	2022 = 65.9	**2022 = 63.7**	**2022 = 69.5**	**2022 = 50**	**2022 = 8.7**	**7/29/22**
Rush Ck. – Old Highway 395 Bridge/Upper Rush section	2013 = 62.6	2013 = 58.8	2013 = 68.7	2013 = 40	2013 = 13.5	7/09/13
	2014 = 64.0	2014 = 60.5	2014 = 69.8	2014 = 51	2014 = 13.3	7/13/14
	2015 = N/A	2015 = N/A	2015 = N/A	2015 = N/A	2015 = N/A	N/A
	2016 = 63.5	2016 = 60.1	2016 = 68.8	2016 = 47	2016 = 12.5	7/11/16
	2017 = 59.0	2017 = 57.5	2017 = 61.0	2017 = 0	2017 = 7.6	9/07/17
	2018 = 60.9	2018 = 58.0	2018 = 65.3	2018 = 0	2018 = 10.9	7/10/18
	2019 = 58.7	2019 = 56.1	2019 = 62.3	2019 = 0	2019 = 10.7	9/14/19
	2020 = 62.6	2020 = 58.5	2020 = 68.4	2020 = 30	2020 = 14.0	8/03/20
	2021 = 65.0	2021 = 61.2	2021 = 70.8	2021 = 63	2021 = 12.8	8/02/21
	2022 = 64.7	**2022 = 60.4**	**2022 = 70.6**	**2022 = 55**	**2022 = 15.2**	**8/07/22**
Rush Ck. – Above Parker	2016 = 63.2	2016 = 58.8	2016 = 69.4	2016 = 55	2016 = 13.7	7/11/16
	2017 = 59.0	2017 = 57.2	2017 = 61.9	2017 = 0	2017 = 8.6	9/08/17
	2018 = 60.9	2018 = 57.2	2018 = 66.3	2018 = 0	2018 = 13.4	7/10/18
	2019 = 58.4	2019 = 55.5	2019 = 62.3	2019 = 0	2019 = 11.8	9/14/19
	2020 = 62.2	2020 = 57.1	2020 = 68.6	2020 = 40	2020 = 16.1	8/03/20
	2021 = 64.4	2021 = 59.6	2021 = 70.8	2021 = 61	2021 = 14.4	8/02/21
	2022 = 64.5	**2022 = 59.4**	**2022 = 70.7**	**2022 = 59**	**2022 = 15.7**	**8/07/22**

Table 15 (continued).

Temperature Monitoring Location	Daily Mean (°F)	Ave Daily Minimum (°F)	Ave Daily Maximum (°F)	No. Days > 70°F	Max Diurnal Fluctuation (°F)	Date of Max. Fluctuation
Rush Ck. – below Narrows	2013 = 61.2 2014 = 63.2 2015 = 62.3 2016 = 61.7 2017 = 58.4 2018 = 60.0 2019 = 57.8 2020 = 61.0 2021 = 63.2 **2022 = 63.2**	2013 = 56.2 2014 = 57.1 2015 = 58.8 2016 = 56.9 2017 = 56.3 2018 = 56.0 2019 = 54.4 2020 = 55.5 2021 = 58.0 **2022 = 58.0**	2013 = 67.6 2014 = 69.4 2015 = 66.1 2016 = 68.3 2017 = 61.3 2018 = 65.4 2019 = 62.2 2020 = 67.5 2021 = 69.7 **2022 = 69.5**	2013 = 24 2014 = 46 2015 = 0 2016 = 34 2017 = 0 2018 =0 2019 = 0 2020 = 16 2021 = 49 **2022 = 52**	2013 = 16.3 2014 = 17.3 2015 = 11.5 2016 = 14.3 2017 = 8.2 2018 = 12.4 2019 = 12.7 2020 = 15.7 2021 = 14.9 **2022 = 14.7**	7/19/13 7/26/14 9/23/15 7/13/16 9/07/17 7/10/18 9/22/19 8/03/20 8/12/21 **7/29/22**
Rush Ck. – County Road	2013 = 61.4 2014 = 62.0 2015 = 62.1 2016 = 61.6 2017 = N/A 2018 = N/A 2019 = 58.2 2020 = 61.0 2021 = 63.1 **2022 = 63.3**	2013 = 56.5 2014 = 56.7 2015 = 59.1 2016 = 56.0 2017 = N/A 2018 = N/A 2019 = 54.0 2020 = 54.5 2021 = 56.6 **2022 = 57.5**	2013 = 66.6 2014 = 67.8 2015 = 65.5 2016 = 68.3 2017 = N/A 2018 = N/A 2019 = 63.6 2020 = 68.5 2021 = 70.7 **2022 = 70.2**	2013 = 7 2014 = 24 2015 = 2 2016 = 32 2017 = N/A 2018 = N/A 2019 = 0 2020 = 42 2021 = 57 **2022 = 55**	2013 = 14.7 2014 = 17.6 2015 = 9.2 2016 = 16.1 2017 = N/A 2018 = N/A 2019 = 13.5 2020 = 18.2 2021 = 17.4 **2022 = 16.8**	8/02/13 7/26/14 7/28/15 7/11/16 N/A N/A 9/13/19 8/03/20 9/02/21 **7/29/22**

Table 16. Classification of 2013-2022 summer water temperature data into good growth days, fair growth days, poor growth days and bad thermal days based on daily average temperatures (92-day period from July 1 to September 30). The percent (%) designates each thermal day-type's occurrence for the 92-day summer period.

Temperature Monitoring Location	No. of Days for Good Growth Potential – Daily Ave. ≤60.5°F	No. of Days for Fair Growth Potential – Daily Ave. 60.6° – 63.9°F	No. of Days of Poor Growth Potential – Daily Ave. 64.0° - 64.9°F	No. of Bad Thermal Days - Daily Ave. ≥65°F
Rush Ck. – Top of MGORD	2013 = 14 (15%) 2014 = 5 (6%) 2015 = 7 (8%) 2016 = 10 (11%) 2017 = 66 (71%) 2018 = 47 (51%) 2019 = 65 (71%) 2020 = 6 (6%) 2021 = 0 **2022 = 12 (13%)**	2013 = 43 (47%) 2014 = 14 (15%) 2015 = 20 (22%) 2016 = 32 (35%) 2017 = 26 (29%) 2018 = 42 (46%) 2019 = 23 (25%) 2020 = 50 (54%) 2021 = 30 (33%) **2022 = 21 (23%)**	2013 = 17 (18%) 2014 = 25 (27%) 2015 = 5 (5%) 2016 = 17 (18%) 2017 = 0 2018 = 3 (3%) 2019 = 4 (4%) 2020 = 12 (13%) 2021 = 8 (9%) **2022 = 5 (6%)**	2013 = 18 (20%) 2014 = 48 (52%) 2015 = 60 (65%) 2016 = 33 (36%) 2017 = 0 2018 = 0 2019 = 0 2020 = 24 (26%) 2021 = 54 (59%) **2022 = 54 (59%)**

Table 16 (continued).

Temperature Monitoring Location	No. of Days for Good Growth Potential – Daily Ave. ≤60.5°F	No. of Days for Fair Growth Potential – Daily Ave. 60.6° – 63.9°F	No. of Days of Poor Growth Potential – Daily Ave. 64.0° - 64.9°F	No. of Bad Thermal Days - Daily Ave. ≥65°F
Rush Ck. – Bottom MGORD	2013 = 11 (12%) 2014 = 6 (6%) 2015 = 8 (9%) 2016 = 9 (10%) 2017 = 67 (73%) 2018 = 48 (52%) 2019 = 62 (68%) 2020 = 4 (4%) 2021 = 14 (15%) **2022 = 23 (25%)**	2013 = 38 (41%) 2014 = 11 (12%) 2015 = 20 (22%) 2016 = 31 (34%) 2017 = 25 (27%) 2018 = 42 (46%) 2019 = 28 (30%) 2020 = 50 (54%) 2021 = 30 (33%) **2022 = 14 (15%)**	2013 = 20 (22%) 2014 = 21 (23%) 2015 = 5 (6%) 2016 = 16 (17%) 2017 = 0 2018 = 2 (2%) 2019 = 2 (2%) 2020 = 18 (20%) 2021 = 13 (14%) **2022 = 7 (8%)**	2013 = 23 (25%) 2014 = 54 (59%) 2015 = 59 (64%) 2016 = 36 (39%) 2017 = 0 2018 = 0 2019 = 0 2020 = 20 (22%) 2021 = 35 (38%) **2022 = 48 (52%)**
Rush Ck. – Old Highway 395 Bridge/Upper Rush section	2013 = 14 (15%) 2014 = 7 (8%) 2015 = N/A 2016 = 16 (17%) 2017 = 75 (82%) 2018 = 36 (39%) 2019 = 64 (70%) 2020 = 17 (18%) 2021 = 24 (26%) **2022 = 29 (32%)**	2013 = 41 (45%) 2014 = 25 (27%) 2015 = N/A 2016 = 24 (26%) 2017 = 17 (18%) 2018 = 56 (61%) 2019 = 28 (30%) 2020 = 48 (52%) 2021 = 30 (33%) **2022 = 17 (18%)**	2013 = 33 (36%) 2014 = 27 (29%) 2015 = N/A 2016 = 19 (21%) 2017 = 0 2018 = 0 2019 = 0 2020 = 17 (18%) 2021 = 11 (12%) **2022 = 7 (8%)**	2013 = 4 (4%) 2014 = 33 (36%) 2015 = N/A 2016 = 33 (36%) 2017 = 0 2018 = 0 2019 = 0 2020 = 10 (11%) 2021 = 27 (29%) **2022 = 39 (42%)**
Rush Ck. – Above Parker Ck.	2016 = 17 (18%) 2017 = 65 (71%) 2018 = 28 (30%) 2019 = 67 (73%) 2020 = 24 (26%) 2021 = 30 (33%) **2022 = 31 (34%)**	2016 = 26 (28%) 2017 = 27 (29%) 2018 = 64 (70%) 2019 = 25 (27%) 2020 = 41 (45%) 2021 = 34 (37%) **2022 = 16 (17%)**	2016 = 24 (26%) 2017 = 0 2018 = 0 2019 = 0 2020 = 21 (23%) 2021 = 10 (11%) **2022 = 7 (8%)**	2016 = 25 (27%) 2017 = 0 2018 = 0 2019 = 0 2020 = 10 (11%) 2021 = 18 (20%) **2022 = 38 (41%)**
Rush Ck. – Below Narrows	2013 = 17 (18%) 2014 = 13 (14%) 2015 = 24 (26%) 2016 = 22 (24%) 2017 = 75 (82%) 2018 = 46 (50%) 2019 = 74 (80%) 2020 = 36 (39%) 2021 = 26 (28%) **2022 = 33 (36%)**	2013 = 69 (75%) 2014 = 58 (63%) 2015 = 44 (48%) 2016 = 52 (57%) 2017 = 17 (18%) 2018 = 46 (50%) 2019 = 18 (20%) 2020 = 53 (58%) 2021 = 39 (42%) **2022 = 22 (24%)**	2013 = 6 (7%) 2014 = 18 (20%) 2015 = 22 (24%) 2016 = 16 (17%) 2017 = 0 2018 = 0 2019 = 0 2020 = 2 (2%) 2021 = 10 (11%) **2022 = 27 (29%)**	2013 = 0 2014 = 3 (3%) 2015 =2 (2%) 2016 = 2 (2%) 2017 = 0 2018 = 0 2019 = 0 2020 = 1 (1%) 2021 = 17 (18%) **2022 = 8 (9%)**
Rush Ck. – County Road	2013 = 17 (18%) 2014 = 17 (18%) 2015 = 25 (27%) 2016 = 24 (26%) 2017 = N/A 2018 = N/A 2019 = 71 (77%) 2020 = 31 (34%) 2021 = 26 (28%) **2022 = 33 (36%)**	2013 = 64 (70%) 2014 = 59 (65%) 2015 = 39 (42%) 2016 = 50 (54%) 2017 = N/A 2018 = N/A 2019 = 21 (23%) 2020 = 50 (54%) 2021 = 31 (34%) **2022 = 15 (16%)**	2013 = 8 (9%) 2014 = 14 (15%) 2015 =23 (25%) 2016 = 13 (14%) 2017 = N/A 2018 = N/A 2019 = 0 2020 = 10 (11%) 2021 = 9 (10%) **2022 = 8 (9%)**	2013 = 3 (3%) 2014 = 2 (2%) 2015 = 5 (6%) 2016 = 5 (6%) 2017 = N/A 2018 = N/A 2019 = 0 2020 = 1 (1%) 2021 = 26 (28%) **2022 = 36 (39%)**

Table 17. Diurnal temperature fluctuations in Rush Creek for 2022: maximum daily for month, daily average for month, and highest average for consecutive 21-day duration (92-day period from July 1 to September 30). NOTE: 2021 values in ( ) for comparison.

Temperature Monitoring Location	Maximum and Average Daily Diurnal Fluctuation for July	Maximum and Average Daily Diurnal Fluctuation for August	Maximum and Average Daily Diurnal Fluctuation for September	Highest Average Diurnal Fluctuation for a Consecutive 21-Day Duration
Rush Ck. – Top of MGORD	Max = 6.6°F (6.5) Ave = 3.0°F (2.3)	Max = 3.3°F (3.4) Ave = 1.7°F (1.5)	Max = 2.9°F (2.2) Ave = 1.1°F (1.0)	3.4°F (1.6) July 11 – 31
Rush Ck. – Bottom MGORD	Max = 8.7°F (8.5) Ave = 5.6°F (5.4)	Max = 7.8°F (8.4) Ave = 6.0°F (7.3)	Max = 7.1°F (7.4) Ave = 5.9°F (6.5)	6.5°F (7.5) Aug 18 – Sept 7
Rush Ck. – Old Hwy 395 Bridge	Max = 13.2°F (11.4) Ave = 9.5°F (8.1)	Max = 15.2°F (12.8) Ave = 10.4°F (10.9)	Max = 12.8°F (12.0) Ave = 10.6°F (9.8)	11.8°F (10.6) Aug 20 – Sept 9
Rush Ck. – Above Parker Ck.	Max = 14.6°F (13.5) Ave = 11.0°F (9.6)	Max = 15.7°F (14.4) Ave = 11.5°F (12.7)	Max = 14.1°F (13.7) Ave = 11.3°F (11.4)	12.8°F (12.8) Aug 19 – Sept 8
Rush Ck. – below Narrows	Max = 14.7°F (13.9) Ave = 11.4°F (10.0)	Max = 14.2°F (14.9) Ave = 11.5°F (12.9)	Max = 14.7°F (14.9) Ave = 11.7°F (12.3)	13.2°F (12.7) Aug 19 – Sept 8
Rush Ck. – County Road	Max = 16.8°F (15.6) Ave = 14.0°F (12.8)	Max = 15.6°F (16.7) Ave = 12.5°F (15.5)	Max = 14.7°F (17.4) Ave = 12.1°F (13.9)	14.5°F (15.9) Jul 7 - 29

Table 18. Number of hours (percent of hours in parentheses) that temperature exceeded 66.2°F in Rush Creek: by month and for 92-day period from July 1 to September 30, 2013 - 2022. The total number of hours within each month is in parentheses in the column headings.

Temperature Monitoring Location	Number of Hours Temperature exceeded 66.2°F in July (744 hours)	Number of Hours Temperature exceeded 66.2°F in August (744 hours)	Number of Hours Temperature exceeded 66.2°F in Sept. (720 hours)	Number of Hours Temperature exceeded 66.2°F in 92-day period
Rush Ck. – Top of MGORD	2013 = 4 hrs (0.5%) 2014 = 315 hrs (42%) 2015 = 140 hrs (19%) 2016 = 42 hrs (6%) 2017 = 0 hrs 2018 = 0 hrs 2019 = 0 hrs 2020 = 0 hrs 2021 = 488 hrs (66%) **2022 = 246 hrs (33%)**	2013 = 4 hrs (0.5%) 2014 = 96 hrs (13%) 2015 = 205 hrs (28%) 2016 = 127 hrs (17%) 2017 = 0 hrs 2018 = 6 hrs 2019 = 0 hrs 2020 = 71 hours (10%) 2021 = 588 hrs (79%) **2022 = 728 hrs (98%)**	2013 = 0 hrs 2014 = 0 hrs 2015 = 0 hrs 2016 = 0 hrs 2017 = 0 hrs 2018 = 0 hrs 2019 = 13 hrs 2020 = 47 hrs (7%) 2021 = 35 hrs (5%) **2022 = 343 hrs (48%)**	2013 = 8 hrs (0.4%) 2014 = 411 hrs (19%) 2015 = 345 hrs (16%) 2016 = 169 hrs (8%) 2017 = 0 hrs 2018 = 6 hrs (0.3%) 2019 = 13 hrs (0.6%) 2020 = 118 hrs (5%) 2021 = 1,111 hrs (50%) **2022 = 1,317 hrs (60%)**

Table 18 (continued).

Temperature Monitoring Location	Number of Hours Temperature exceeded 66.2°F in July (744 hours)	Number of Hours Temperature exceeded 66.2°F in August (744 hours)	Number of Hours Temperature exceeded 66.2°F in Sept. (720 hours)	Number of Hours Temperature exceeded 66.2°F in 92-day period
Rush Ck. – Bottom MGORD	2013 = 121 hrs (16%) 2014 = 282 hrs (38%) 2015 = 305 hrs (41%) 2016 = 142 hrs (19%) 2017 = 0 hrs 2018 = 0 hrs 2019 = 0 hrs 2020 = 49 hrs (6%) 2021 = 444 hrs (60%) **2022 = 257 hrs (35%)**	2013 = 229 hrs (31%) 2014 = 248 hrs (33%) 2015 =282 hrs (38%) 2016 = 268 hrs (36%) 2017 = 0 hrs 2018 = 1 hr (0.01%) 2019 = 0 hrs 2020 = 234 hrs (31%) 2021 = 376 hrs (51%) **2022 = 535 hrs (72%)**	2013 = 61 hrs (9%) 2014 = 115 hrs (16%) 2015 = 17 hrs (2%) 2016 = 38 hrs (5%) 2017 = 2 hrs (0.3%) 2018 = 1 hr (0.01%) 2019 = 46 hrs (6%) 2020 = 101 hrs (14%) 2021 = 125 hrs (17%) **2022 = 247 hrs (34%)**	2013 = 411 hrs (19%) 2014 = 645 hrs (29%) 2015 = 604 hrs (27%) 2016 = 448 hrs (20%) 2017 = 2 hrs (0.09%) 2018 = 2 hrs (0.09%) 2019 = 46 hrs (2%) 2020 = 335 hrs (15%) 2021 = 945 hrs (43%) **2022 = 1,039 hrs (47%)**
Rush Ck. – Old 395 Bridge/Upper Rush	2013 = 181 hrs (24%) 2014 = 287 hrs (39%) 2016 = 216 hrs (29%) 2017 = 0 hrs 2018 = 17 hrs (2%) 2019 = 0 hrs 2020 = 113 hrs (15%) 2021 = 351 hrs (47%) **2022 = 252 hrs (34%)**	2013 = 228 hrs (31%) 2014 = 248 hrs (33%) 2016 = 263 hrs (35%) 2017 = 0 hrs 2018 = 32 hrs (4%) 2019 = 4 hrs (0.5%) 2020 = 241 hrs (32%) 2021 = 328 hrs (44%) **2022 = 350 hrs (47%)**	2013 = 73 hrs (10%) 2014 = 117 hrs (16%) 2016 = 53 hrs (7%) 2017 = 3 hrs (0.4%) 2018 = 33 hrs (5%) 2019 = 41 hrs (6%) 2020 = 87 hrs (12%) 2021 = 127 hrs (18%) **2022 = 162 hrs (23%)**	2013 = 482 hrs (22%) 2014 = 639 hrs (29%) 2016 = 532 hrs (24%) 2017 = 3 hrs = (0.1%) 2018 = 82 hrs (4%) 2019 = 45 hrs (2%) 2020 = 441 hrs (20%) 2021 = 806 hrs (37%) **2022 = 764 hrs (35%)**
Rush Ck. – Above Parker Creek	2016 = 240 hrs (32%) 2017 = 0 hrs 2018 = 70 hrs (9%) 2019 = 0 hrs 2020 = 146 hrs (20%) 2021 = 342 hrs (46%) **2022 = 276 hrs (37%)**	2016 = 269 hrs (36%) 2017 = 0 hrs 2018 = 68 hrs (9%) 2019 = 11 hrs (2%) 2020 = 257 hrs (35%) 2021 = 316 hrs (42%) **2022 = 348 hrs (47%)**	2016 = 65 hrs (9%) 2017 = 14 hrs (2%) 2018 = 44 hrs (6%) 2019 = 27 hrs (4%) 2020 = 73 hrs (10%) 2021 = 122 hrs (17%) **2022 = 157 hrs (22%)**	2016 = 574 hrs (26%) 2017 = 14 hrs (0.6%) 2018 = 182 hrs (8%) 2019 = 38 hrs (2%) 2020 = 476 hrs (22%) 2021 = 780 hrs (35%) **2022 = 781 hrs (35%)**
Rush Ck. – below Narrows	2013 = 158 hrs (21%) 2014 = 244 hrs (33%) 2015 = 129 hrs (17%) 2016 = 167 hrs (22%) 2017 = 0 hrs 2018 = 36 hrs (5%) 2019 = 0 hrs 2020 = 109 (15%) 2021 = 273 hrs (37%) **2022 = 243 hrs (33%)**	2013 = 192 hrs (26%) 2014 = 193 hrs (26%) 2015 = 189 hrs (25%) 2016 = 222 hrs (30%) 2017 = 0 hrs 2018 = 42 hrs (6%) 2019 = 13 hrs (2%) 2020 = 204 hrs (27%) 2021 = 267 hrs (36%) **2022 = 265 hrs (36%)**	2013 = 55 hrs (7%) 2014 = 105 hrs (15%) 2015 = 0 hrs (0%) 2016 = 49 hrs (7%) 2017 = 0 hrs 2018 = 36 hrs (5%) 2019 = 8 hrs (1%) 2020 = 43 hrs (6%) 2021 = 104 hrs (14%) **2022 = 109 hrs (15%)**	2013 = 405 hrs (18%) 2014 = 542 hrs (25%) 2015 = 318 hrs (14%) 2016 = 438 hrs (20%) 2017 = 0 hrs 2018 = 114 hrs (5%) 2019 = 21 hrs (1%) 2020 = 356 hrs (16%) 2021 = 644 hrs (29%) **2022 = 617 hrs (28%)**
Rush Ck. – County Road	2013 = 197 hrs (27%) 2014 = 222 hrs (30%) 2015 = 174 hrs (23%) 2016 = 212 hrs (28%) 2017 = N/A 2018 = N/A 2019 = 0 hrs 2020 = 195 hrs (26%) 2021 = 301 hrs (40%) **2022 = 290 hrs (39%)**	2013 = 172 hrs (23%) 2014 = 195 hrs (26%) 2015 = 119 hrs (16%) 2016 = 233 hrs (31%) 2017 = N/A 2018 = N/A 2019 = 76 hrs (10%) 2020 = 241 hrs (32%) 2021 = 278 hrs (37%) **2022 = 282 hrs (38%)**	2013 = 42 hrs (6%) 2014 = 79 hrs (11%) 2015 = 0 hrs (0%) 2016 = 42 hrs (6%) 2017 = N/A 2018 = N/A 2019 = 10 hrs (1%) 2020 = 41 hrs (6%) 2021 = 99 hrs (14%) **2022 = 107 hrs (15%)**	2013 = 411 hrs (19%) 2014 = 496 hrs (23%) 2015 = 293 hrs (13%) 2016 = 487 hrs (22%) 2017 = N/A 2018 = N/A 2019 = 86 hrs (4%) 2020 = 477 hrs (22%) 2021 = 678 hrs (31%) **2022 = 679 hrs (31%)**

Discussion

The 2022 sampling was marked by being the first year of fisheries monitoring under the newly issued WR-2021-0086, which amended LADWP's license and signaled the start of the 10-year post-settlement monitoring period. During this 10-year period, all monitoring activities (fisheries, geomorphic/riparian, Mono Lake limnology and waterfowl) will be conducted by consultants, with oversight from the MAT. The purpose of the post-settlement monitoring is to evaluate the effectiveness of the SEF flow regimes and, as needed, make recommended changes to these flows (timing and magnitude), as long as the overall quantity of water released by LADWP is not increased from those defined in WR-2021-0086.

The 2022 sampling year was also highlighted by a second consecutive Dry RY and thermally challenging water temperature conditions in Rush Creek during the summer months. Thus, this report's Discussion is focused on the long-term trout population metrics and summer water temperatures as related to trout health. An examination of Lee Vining air temperature is also made, in context of how air temperatures influence water temperatures. The Discussion section concludes with a methods evaluation and proposed fisheries monitoring activities for 2023.

Trout Population Metrics

Annual fisheries sampling in Rush and Lee Vining Creeks since 1999 has provided an unusually long-term data set of trout population metrics. The overarching theme of these data is that trout populations respond better to wetter runoff years than to average-to-drier runoff years. The best example of this trend was the recent five-year drought of 2012-2016 in which the recruitment of age-0 Brown Trout decreased by 95% in the Upper Rush section and by 89% in the Bottomlands section. Numbers and condition factors of older trout also decreased during this five-year drought. Then, two good runoff years in 2017 and 2018 with a full GLR, saw trout populations rebound quickly with age-0 recruitment increasing nearly two-fold (200%) in Upper Rush and more than 12-fold (1,200%) in the Bottomlands section. Growth rates and condition factors also improved in the two years post-drought. In fact, growth rates measured in September of 2017 were the highest ever recorded, with elevated streamflows all summer long. Thus, trout numbers, growth rates and conditions factors in Rush and Lee Vining Creeks can oscillate widely depending on runoff year type.

During the 2/15/23 Mono Lake level workshop hosted by the SWRCB, LADWP's presentation included a slide stating the "*fish populations are thriving*" and "*the four Mono Basin creeks have been restored"* (Figure 21). Regarding the trout populations in Rush and Lee Vining Creeks, the long-term data strongly suggest that the populations only thrive in wetter runoff years but decline during drier runoff years. The lack of sustained wet years impedes the trout populations from truly thriving, prospering, or growing vigorously. There is little reason to expect this boom-bust population metrics cycle to improve in coming years, especially with a warming climate. If anything, less reliable snowpacks and warmer summer conditions should be expected in the future. However, at this point in time, it is unknown if a more mature and restored woody riparian canopy would provide adequate shading to reduce summer thermal impacts. The long-term condition factor analyses of Brown Trout in Rush Creek also refute the statement that the fish populations are thriving. For example, in the Bottomlands section, in 14 of the 15 sample years, condition factors were <1.00, that is, fish were in poor condition in more than 90% of the years that this section was sampled. In the Upper Rush section, in 12 of 23 sample years, condition factors were <1.00. Also, of the 11 years where condition factors were barely above 1.00, seven of these years occurred in 2000-2006, and a condition factor of 1.00-1.19 is considered fish in average condition, not fish in good condition (Barnham and Baxter 1998). Good condition factors have not yet been recorded for fish in Rush Creek.

The statement that the streams have been restored implies that the restoration process is complete. In 2006-2007, the Stream Scientists summarized the status of the Termination Criteria and the feasibility of, if and when, these criteria would be met in two technical memorandums submitted to the SWRCB. The geomorphic/riparian memorandum discussed feasibility of the current stream corridors in growing and sustaining woody riparian vegetation similar to 1929 conditions by the year 2100 (Trush 2006). This statement alone implies restoration of the woody riparian vegetation is a long way from *"have been restored."* In addition, Section 5.3.2 of Order WR98-05 summarized the testimony of several experts in supporting the monitoring of restoration efforts, but also highlighted the difficulties in attempting to specify criteria for establishing when restoration should be considered complete. Dr. Kauffman testified that ecological restoration is an ongoing process, which is *"not completed at any one point in time"*, but that the restoration plan proposed by LADWP *"sets the ecosystem in the right trajectory for a goal of naturally functioning ecosystems."* Dr. Trush testified that instead of trying to define an end product, that idea should be replaced with establishing processes to allow the channels to react and function alluvially.

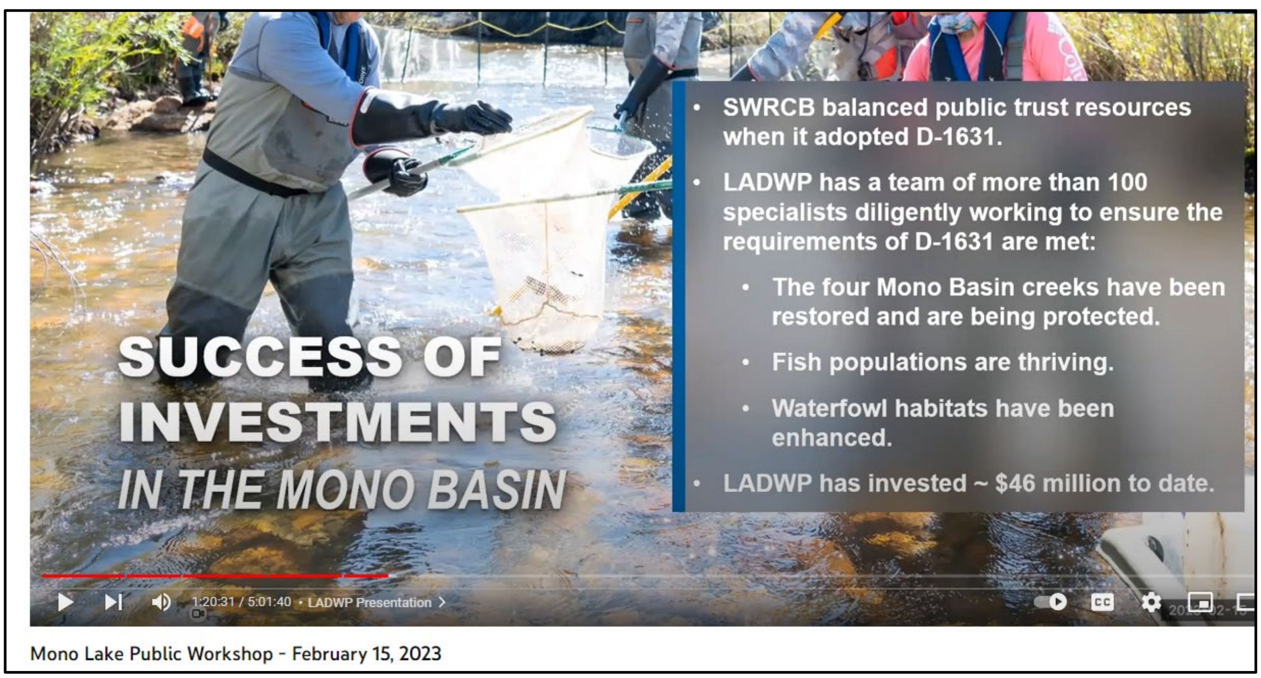


Figure 21. Screen shot of LADWP's presentation during the Mono Lake public workshop.

The second major theme of the long-term fisheries data set is that in terms of a recreational fishery for naturally produced wild trout, present-day Rush Creek (below the MGORD) is unlikely to recover to the presumed pre-1941 conditions of Brown Trout averaging 13-14 inches (330-355 mm) in length with fish of "3/4 to 2 pounds fairly consistently produced". These descriptions of the pre-1941 fishery in Rush Creek were included in SWRCB's Decision 1631 and were also the basis of setting the initial fisheries termination criteria and the fisheries monitoring objectives in WR-98-05 and WR-98-07. Our long-term PIT tag data and length-at-age data show that Brown Trout in most Rush Creek sections attain lengths ≥300 mm at age-4 or age-5 and that very few trout survive to these ages. In 2022, the Upper Rush section had an RSD-225 value of 17, meaning of the sub-population of catchable trout (>150 mm) 17% were ≥225 mm (≈9 inches) and the RSD-300 value was 1 or only 1% of the catchable trout were ≥300 mm (≈12 inches). The only two years out of the past 23 years where the RSD-300 was >5 in Upper Rush were 2017 and 2018, the two good runoff years following the five-year drought. The MGORD, a trapezoidal diversion canal, is the only section of Rush Creek downstream of GLR that consistently produces Brown Trout of memorable (RSD-300) or trophy (RSD-375) sizes and is the only reach of Rush Creek downstream of GLR that receives much fishing pressure. However, the long-term RSD-300 data for the MGORD documents a decline in the proportion of Brown Trout ≥300 mm. For the years 2001-2012, the RSD-300 ranged from 16 to 54 with an average of 30; whereas between 2013-2022, the RSD-300 ranged from 7 to 27 with an average of 16. A drive around the June Lake Loop reveals that nearly all of the fishing pressure in the Rush Creek watershed is in the lakes or creek sections where hatchery trout are planted and sport anglers may use bait and retain a daily bag limit of five trout.

Summer Water Temperatures

Water temperature metrics are varied and papers exist that summarize studies performed to evaluate thermal effects on Brown Trout (Armour 1997; Bell 2006). Diurnal fluctuations previously cited as detrimental and/or stressful to trout (Werley et al. 2007) are supported by additional research. For example, Rainbow Trout physiological changes such as increased ventilatory rates and stroke rates in response to increases in water temperature have been reported (Henry 1978). This research also documented trout acclimated to 64.5°F water and subjected to 7.2°F diurnal fluctuations exhibited signs of ventilatory and cardiovascular distress, problems commonly associated with low circulating levels of oxygen in the blood (Henry 1978). It appears these trout were unable to fully meet their oxygen requirements associated with cycling temperatures above 64.5°F. When trout are unable to fully meet their oxygen requirements, stress levels elevate and fish may become more susceptible to parasites and other disease vectors. For example, studies of riverine wild Brown Trout populations in Switzerland and proliferative kidney disease (PKD) caused by a myxozoan parasite reported that parasite prevalence and intensity on trout were most strongly correlated to daily mean water temperature during summer months (Ruben et al. 2019). This study concluded that parasite infection prevalence increased by nearly 6% for every one degree (Celsius) increase of daily mean summer water temperature above 15°C (Ruben et al. 2019). The authors speculated that the prevalence and intensity of PKD in Brown Trout will increase with ongoing climate change and continued warming of Switzerland's trout-bearing rivers.

As climate change continues to alter the thermal regimes of coldwater fish habitat, studies continue to investigate how sub-lethal water temperatures affect trout growth, and how growth limitations may influence fish distribution patterns. Chadwick and McCormick (2017) subjected Brook Trout to chronically elevated and daily oscillating temperatures and evaluated growth and physiological stress responses. This study confirmed that growth rates were reduced and that numerous physiological changes occurred to their study subjects, including cellular and endocrine stress. Growth by length and weight decreased by 43% and 35%, respectively, when trout were subjected to four days of 14.4°F temperature fluctuations (Chadwick and McCormick 2017). Similar decreases in growth rates were reported for Lahontan Cutthroat Trout, where growth declined with increasing magnitude of daily oscillations around a mean of 64.4°F (Meeuwig et al. 2004).

For the Synthesis Report, the Stream Scientists recommended that in GLR a minimum storage level of 7,100-foot elevation was maintained during the summer to avoid the release of warmer water to Rush Creek below GLR (McB&T and RTA 2010). This minimum recommended summer storage level was derived from previous GLR temperature modeling conducted in 1991 and 1992, where at reservoir storage levels below 7,100 feet an inflection point occurred where water temperatures released to the MGORD increased (Cullen and Railsback 1993).

The fact that GLR's summer storage levels for the past three summers were mostly three to 20+ feet higher than 7,100 feet, yet all three years resulted in unfavorable thermal conditions for Brown Trout in Rush Creek, begs asking the following questions: Why isn't this storage level recommendation producing adequate summer thermal conditions for good trout growth rates and condition factors? Has GLR continued to fill with sediment and is its actual storage volume significantly less than 47,000 acre-feet, thus rendering the 1993 modeled predictions of storage level versus water temperature inaccurate or no longer valid? Is changing climate leading to hotter summer air temperatures in the Mono Basin, and if so, do these air temperatures exert more thermal loading to streamflow in Rush Creek?

In regards to the question of changing climate; yes, summer air temperatures in the Mono Basin have steadily increased over the past 33 years (Table 19). Broken down by decades (1990's, 2000's, 2010's and 2020's), the metrics of average maximum and number of days with peak temperatures ≥90°F have all increased (Table 19). The average maximum air temperature in the 1990's equaled 80.4°F and in the first three years of the 2020's, the average maximum air temperature was 86.0°F. The number of days with peak air temperatures ≥90°F has recently experienced the biggest increase. In the first 25 years there were four years (1994, 2002, 2007 and 2012) where at least 10 days had maximum air temperatures ≥90°F versus in the most recent seven years (2016-2022), six of the years experienced at least 10 days with maximum air temperatures ≥90°F (Table 19). The past three years have had the highest totals of days with maximum air temperatures ≥90°F (Table 19).

Studies have shown that a combination of air temperature, direct solar radiation, basin-specific hydrology, channel bed morphology, shade, and anthropogenic disturbances all exert an influence on the water temperature regimes of streams and rivers. Harvey et al. (2011) developed nonlinear logistic models to represent the relationship between water temperature in Newfoundland streams and air temperature data, with better accuracy at weekly to monthly scales. The Pacific Northwest Research Station conducted stream-shading experiments which indicated that direct solar radiation was the primary contributor to daily fluctuations in stream temperature (Lewis 2005). Although water temperatures typically increase as air temperatures increase, Lewis (2005) stressed that just because there is a correlation between air and water temperatures, this does not imply causation. This study concluded that shading's biggest effect was on the reduction of maximum daily water temperatures (Lewis 2005). Ficklin et al. (2013) focused their modeling studies on changes to stream temperatures of Sierra Nevada watersheds as related to air temperature and basin-specific hydrology (especially changes in snowmelt hydrology in the face of climate change). As the climate heats up, for streams with a snowmelt component, increases in water temperature were shown to exhibit a connection to seasonal shifts and decreases in snowpack, earlier timing of snowmelt and changes in local hydrology, in addition to the influences of increased air temperatures (Ficklin et al. 2013). The authors concluded that substantial changes in water quality can be expected in Sierra Nevada watersheds under future climates and that these changes would be most significant during spring and summer months, and may include water temperature increases of up to 6°C or 10.8°F (Ficklin et al. 2013).

Periods of drought will most likely continue to negatively impact the Rush Creek Brown Trout fishery in terms of population size, growth rates and condition factors. However, after the recent five-year drought, the fishery exhibited resiliency and experienced quick improvements in the numbers of fish, growth rates and condition factors. Thus, changing climate and variable snowpack conditions in the eastern Sierra will most likely dictate the long-term fate and viability of Rush Creek's Brown Trout fishery. However, continued SEF releases, maturation of the riparian canopy, nine more years of annual monitoring, and adaptive management may push the restoration of the creeks and the trout fisheries in a positive direction.

Table 19. Thirty-three years of summer (July-September) air temperature data for Lee Vining, CA. Data are from Western Regional Climate Center and National Weather Service/Reno.

YEAR	Ave Max Temp (ºF)	Ave Min Temp (ºF)	Ave of Daily Ave Temp (ºF)	Number of Days ≥90ºF
1990	80.2	49.8	65.0	1
1991	81.3	51.3	66.3	4
1992	79.9	49.7	64.9	0
1993	N/A	N/A	N/A	N/A
1994	82.7	51.3	67.0	12
1995	80.8	49.9	65.3	0
1996	80.7	50.3	65.4	3
1997	79.1	49.1	64.2	0
1998	79.4	51.2	65.4	7
1999	79.4	49.6	64.5	4
1990's Averages	**80.4**	**50.2**	**65.3**	**3.4**
2000	80.6	49.4	65.0	2
2001	81.9	51.8	66.9	4
2002	81.9	51.1	66.5	14
2003	82.3	51.6	66.9	5
2004	80.6	48.3	64.5	1
2005	79.8	50.3	65.0	6
2006	80.6	50.3	65.4	7
2007	81.7	52.0	66.8	12
2008	83.3	51.5	67.4	6
2009	82.1	50.7	66.4	5
2000's Averages	**81.5**	**50.7**	**66.1**	**6.2**
2010	81.9	49.7	65.8	4
2011	81.7	51.8	66.8	1
2012	84.4	52.6	68.5	12
2013	81.3	50.4	65.9	8
2014	81.5	51.6	66.6	6
2015	80.9	50.9	65.9	5
2016	83.3	49.1	66.2	16
2017	81.4	51.3	66.4	10
2018	83.6	51.8	67.7	13
2019	81.4	50.2	65.8	2
2010's Averages	**82.1**	**50.9**	**66.6**	**7.7**
2020	83.5	50.1	67.2	17
2021	90.7	45.1	63.7	24
2022	83.8	53.5	68.6	22
2020's Averages	**86.0**	**49.6**	**66.5**	**21.0**

Methods Evaluation

As in previous years, small variations in wetted channel widths were measured, which resulted in changes to sample section areas. Thus, it is recommended that channel lengths and widths are re-measured annually.

In 2022, no fisheries sampling occurred in the Lee Vining Creek side channel. The decision to drop this section was based on reducing sampling effort and also that this side channel has carried very little flow (1-2 cfs visual estimate) since 2006 and typically supports low numbers of mostly age-0 Brown Trout. We intend not to sample this section unless a channel-forming flow event redistributes more (>25%) streamflow back into this side channel.

Starting in 2022, the water temperature monitoring was conducted solely by MLC personnel. In the past, water temperature data collected by LADWP were used in the annual fisheries report. Historically, all but one water temperature monitoring sites in Rush Creek were located downstream of GLR. The one upstream site was named At Damsite, a misleading label since this site was not at the dam and was acutally located upstream of GLR. At Damsite is associated with an LADWP flow measurement weir, located in Rush Creek between Silver Lake and GLR. LADWP staff placed the HOBO recording device within the concrete encasement of the weir to prevent tampering and/or theft. This site recorded cooler water temperatures with relatively small diurnal fluctuations (Taylor 2022). After obtaining the 2022 data from MLC, RTA noticed large diurnal fluctuations recorded at the At Damsite location and then delved back into MLC's data from 2021 and 2020. It appears that LADWP's placement of the HOBO data logger inside the flow weir's encasement was collecting inaccurate data, failing to accurately capture daily peaks and breadth of diurnal fluctutations. Thus, in 2023, RTA will work closely with MLC in site selection for collecting summer water temperature data in Rush Creek upstream of GLR.

The PIT tagging program was continued during the September 2022 sampling; tags were implanted primarily in age-0 fish in all sections and also in presumed age-1 fish in the MGORD. The PIT tagging program allowed us to continue to document annual growth rates of trout, calculate apparent survival rates, and assess the ability of fish to reach or exceed lengths of 300 mm. Continuation of the PIT tagging program is recommended during the post-settlement Fisheries Monitoring Program.

Trout size classes (<125, 125-199, and ≥200 mm) developed and discussed during the 2008 annual report should continue to be used for calculations of population estimates (Hunter et al. 2008). Using these size classes provides for long-term consistency as well as year to year consistency with the annual fisheries data sets. However, we acknowledge that in Walker Creek, some age-1 Brown Trout are occasionally less than 125 mm in total length and in MGORD are bigger than 199 mm.

To ensure that electrofishing sampling can be conducted safely and efficiently, flow in Rush Creek should not exceed **35 cfs** and flow in Lee Vining Creek should not exceed **30 cfs** during the annual sampling period. Allowances for flow variances to allow for safe wading conditions and effective sampling were included in the new SWRCB WR-2021-0086.

As of mid-April 2023, the Mono Basin is experiencing record snowfall with multiple storms. The February 1, 2023 snowpack in the Mono Basin was measured at 264% of normal and the April 1st forecast was 229% of normal and the 2023 RY was classified as Extremely-Wet, which will translate into an extended snowmelt runoff, a substaintal spill from GLR, and extended peak flows in both Rush and Lee Vining Creeks well into the summer months. These flows should translate into cooler water temperatures in Rush Creek, thus higher growth rates and condition factors of Brown Trout. Extended high flows will likely cause a delay in the fall fisheries sampling to early October, similar to what occurred during the Extremely-Wet RY in 2017.

Proposed Fisheries Sampling for 2023 Season

During the development of the post-settlement monitoring scope and budget, RTA proposed that the annual fisheries sampling was reduced to conducting population estimate sampling every other year. In the other years, single-pass electrofishing sampling would occur to collect data to evaluate population age-class structure, compute condition factors, generate growth data from recaptures of previously tagged fish, and implant PIT tags in new cohorts of fish. We intend to conduct single-pass sampling in the fall of 2023. In addition to conducting single-pass sampling at the annually sampled locations, RTA proposes sampling several locations in Rush Creek in conjunction with field observations made by MLC staff of side-channel and off-channel habitats. Summer water temperatures would also be collected within these specific habitats.

In 2009, the fisheries crew collected benthic macroinvertebrates (BMI) from Rush and Lee Vining Creeks following methods developed for determining a BMI index of biological integrity (IBI) for stream assessments in the eastern Sierra Nevada of Califorina (Herbst and Silldorff 2009). Our 2009 collections from Rush Creek showed improved IBI scores over samples collected in 2000 and reported by Herbst and Silldorff (2009). The Fisheries Stream Scientist proposes that a discussion occur with the MAT to repeat these BMI collections and IBI analyses in 2023 or 2024, 14-15 years since the past effort was made to determine if the creeks are still trending towards recovery of their BMI populations. If implemented, the responsibilities of this task could be shared with the Geomorphic/Riparian Stream Scientist (Bill Trush) and MLC staff.

The Fisheries Stream Scientist continues to recommend that a bathymetric survey of GLR is conducted to assess the amount of sediment infill and to determine the actual storage capacity of GLR. Reduced storage may negate assumptions about minimum storage levels for suitable summer water temperatures in Rush Creek, as well as affect water availability for future exports. Bathymetric surveys are commonly used to assess rates of sedimentation in reservoirs and the loss of storage capacity can depend on a number of factors (Iradukunda and Bwanbale 2021). Globally the overall loss of reservoir storage capacity is estimated at 1 to 2% of total storage capacity per year (Iradukunda and Bwanbale 2021). In 1991, the GLR thermal characteristics study by Cullen and Railsback (1993) determined that their reservoir thermal model underpredicted water surface elevations, which corresponded to a storage volume error of approximately 1,000 acre-feet and that sedimentation may likely have caused this discrepancy in their depth-to-volume regression.

Recommended changes to SEF’s based on 2022 Fisheries Monitoring

At this point, the Fisheries Stream Scientist has no recommended changes to the SEFs based on results of the 2022 fisheries monitoring. The primary stressor to Rush Creek’s trout population in 2022 was unfavorable summer water temperatures. At this point, there is no mechanical or operational “fix” to reduce or relieve stressfully warm water temperatures in Rush Creek via manipulation of water releases from GLR, other than heading into each summer season with as full of a reservoir as feasibily possible.

The Fisheries Stream Scientist will work with the Geomorphic/Riparian Stream Scientist in reviewing LADWP’s draft AOP for 2023 and provide comments, as needed. This year’s AOP includes addressing flow releases during an Extremely-wet RY, LADWP’s export allowance based on Mono Lake’s level on 4/1/23, and if GLR will require a late-season drawdown for LADWP’s implementation of the GLR’s spillway modification project in 2024 (if the project is green-lighted for construction in 2024).

References Cited

Armour, C.L. 1997. Evaluating temperature regimes for the protection of Brown Trout. U.S. Department of the Interior, National Biological Survey, Resource Publication #201. 26 p.

Barnham, C. and A. Baxter. 1998. Condition factor, K, in salmonid fish. Fisheries Notes #FN0005 State of Victoria, Australia. 3 p.

Bateman, D.S., R.E. Gresswell and A.M. Berger. 2009. Passive integrated tag retention rates in headwater populations of coastal cutthroat trout. North American Journal of Fisheries Management 29: 653-657.

Bell, J.M. 2006. The assessment of thermal impacts on habitat selection, growth, reproduction, and mortality in Brown Trout (*Salmo trutta*): a review of the literature. Vermillion River EPA Grant #WS 97512701-1. 23p.

Blackwell, B.G., M.L. Brown and D.W. Willis. 2000. Relative weight (W_r) status and current use in fisheries assessment and management. Reviews in Fisheries Science, 8(1): 1-44.

Chadwick, J.G and S.D. McCormick. 2017. Upper thermal limits of growth in brook trout and their relationship to stress physiology. Journal of Experimental Biology, 220: 3976-3987.

Cone, R.S. 1989. The need to reconsider the use of condition indices in fishery science. Transactions of the American Fisheries Society 118: 510-514.

Cullen, R.T. and S.F. Railsback. 1993. Summer thermal characteristics of Grant Lake, Mono County, California. Feasibility Study #2, Trihey and Associates, Concord, CA. 118 p.

Dare, M.R. 2003. Mortality and long-term retention of passive integrated tags by spring Chinook salmon. North American Journal of Fisheries Management 23: 1015-1019.

Ficklin, D.L., I.T. Stewart and E.P. Maurer. 2013. Effects of climate change on stream temperature, dissolved oxygen and sediment concentration in the Sierra Nevada in California. Water Resources Research, 49: 2765-2782.

Gabelhouse, D. W., Jr. 1984. A length-categorization system to assess fish stocks. North American Journal of Fisheries Management 4:273-285.

Harvey, R., L. Lye, A. Khan, and R. Paterson. 2011. The influence of air temperature on water temperature and the concentration of dissolved oxygen in Newfoundland rivers. Canadian Water Resources Journal, 36:171-192.

Henry, J.A.C. 1978. Effect of diurnal cycling temperatures on cardiovascular-ventilatory function in statically-acclimated rainbow trout, *Salmo gairdneri*. Department of Biological Sciences, Brock University, Ontario, Canada. 246 p.

Herbst, D.B. and E.L. Silldorff. 2009. Development of a benthic macroinvertebrate index of biological integrity (IBI) for stream assessments in the eastern Sierra Nevada of California. Final technical report for the Surface Water Ambient monitoring Program (SWAMP). 90 p.

Hunter, C., B. Shepard, K. Knudson, R. Taylor and M. Sloat. 2004. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2003. Los Angeles Department of Water and Power. 45 p.

Hunter, C., R. Taylor, K. Knudson, B. Shepard, and M. Sloat. 2005. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2004. Los Angeles Department of Water and Power. 51 p.

Hunter, C., R. Taylor, K. Knudson and B. Shepard. 2006. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2005. Los Angeles Department of Water and Power. 61 p.

Hunter, C., R. Taylor, K. Knudson and B. Shepard. 2007. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2006. Los Angeles Department of Water and Power. 74 p.

Hunter, C., R. Taylor, K. Knudson and B. Shepard. 2008. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2007. Los Angeles Department of Water and Power. 49 p.

Iradukunda, P. and E. Bwambale. 2021. Reservoir sedimentation and its effect on storage capacity – a case study of Murera Reservoir, Kenya. Cogent Engineering 8: 14 p.

Lewis, T. 2005. Keeping it cool: unraveling the influences on stream temperature. Pacific Northwest Research Station, Science Findings, Issue #73: 6p.

McB&T and RTA. 2010. Synthesis of instream flow recommendations to the State Water Resources Control Board and the Los Angeles Department of Water and Power. 159 p.

Meeuwig, M., H. Dunham, J.B. Hayes and G.L. Vineyard. 2004. Effects of constant and cyclical thermal regimes on growth and feeding of juvenile cutthroat trout of variable sizes. Ecology of Freshwater Fishes, 13: 208-216.

Reimers, N. 1963. Body condition and over-winter survival of hatchery-reared trout in Convict Creek, California. Transactions of the American Fisheries Society 92 (1): 39-46.

Ruben, A., P. de Coulon, C. Bailey, H. Segner, T. Wahli and J. Rubin. 2019. Keeping an eye on wild Brown Trout (*Salmo trutta*) populations: correlation between temperature, environmental parameters and proliferative kidney disease. Frontiers in Veterinary Sciences, Volume VI: 281-302.

Shepard, B., R. Taylor, K. Knudson and C. Hunter. 2009. Effects of flow, reservoir storage and water temperature on trout in Rush and Lee Vining Creeks, Mono County, California. Prepared for LADWP. 117 p.

Taylor, R. and K. Knudson. 2012. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2011. Los Angeles Department of Water and Power. 90 p.

Taylor, R. 2013. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2012. Los Angeles Department of Water and Power. 100 p.

Taylor, R. 2014. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2013. Los Angeles Department of Water and Power. 89 p.

Taylor, R. 2019. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2018. Los Angeles Department of Water and Power. 89 p.

Taylor, R. 2020. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2019. Los Angeles Department of Water and Power. 96 p.

Taylor, R. 2021. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2020. Los Angeles Department of Water and Power. 97 p.

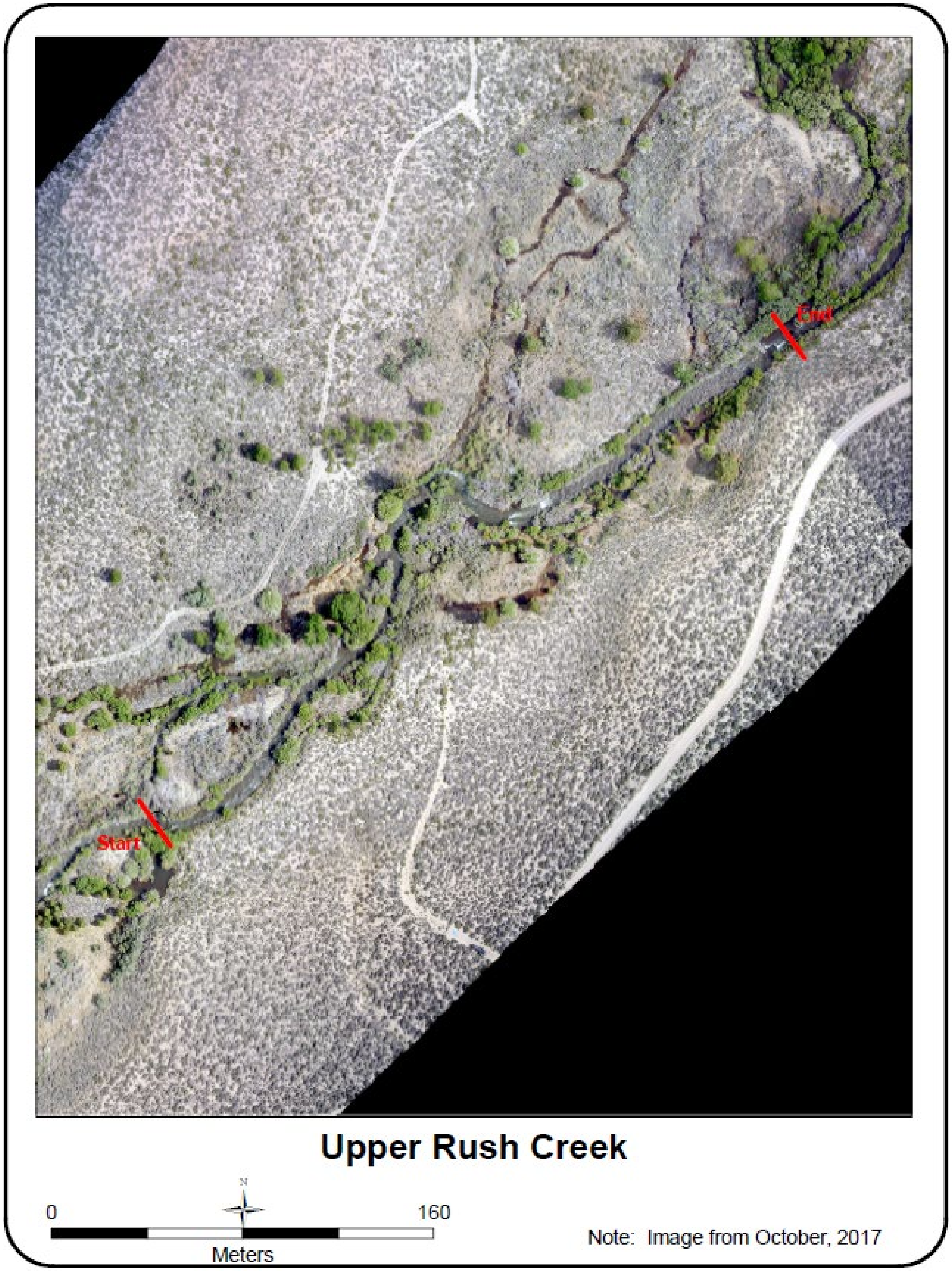
Taylor, R. 2022. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2021. Los Angeles Department of Water and Power. 97 p.

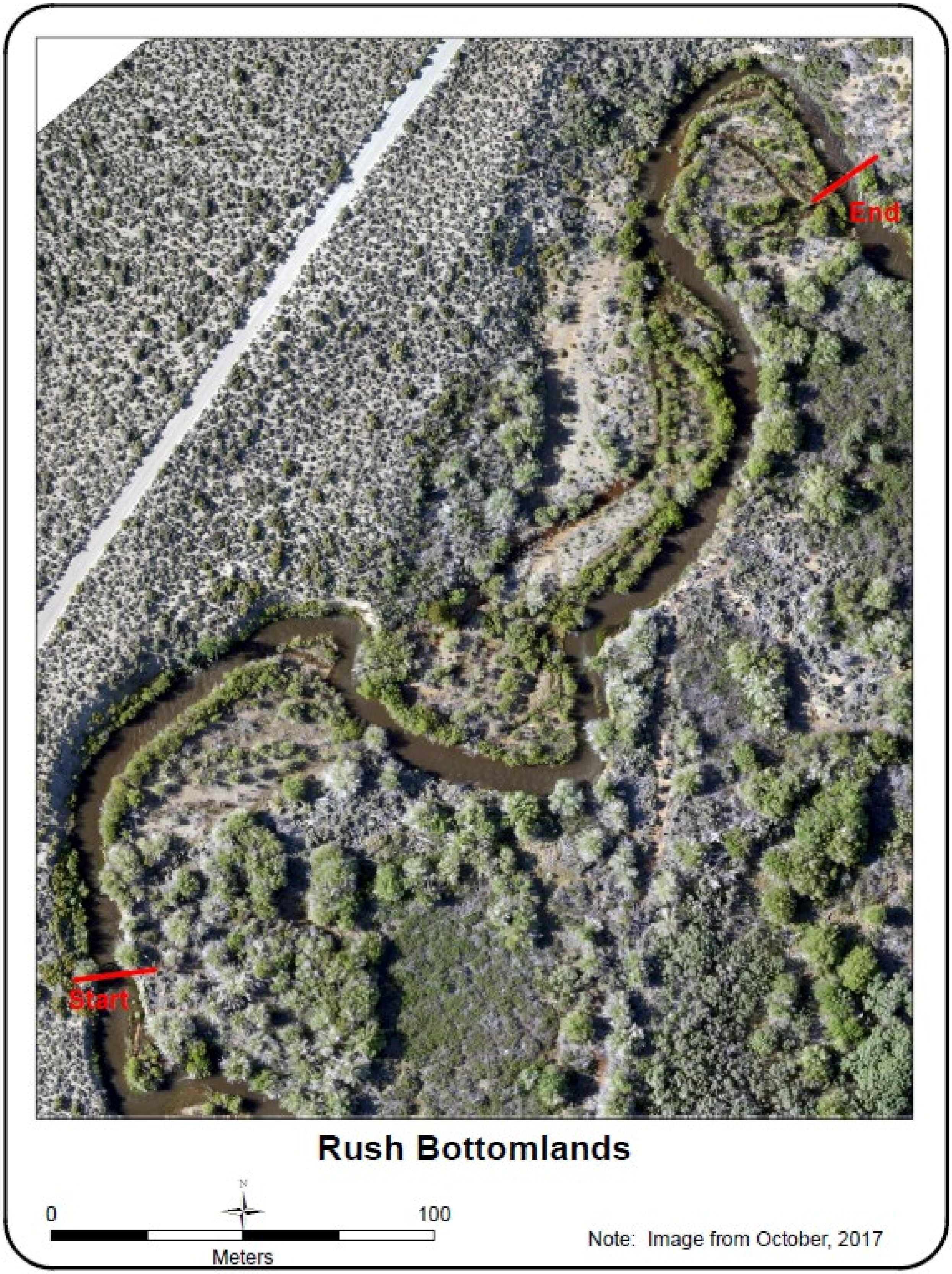
Trush, W.J. 2006. Technical Memorandum: status and recommended revisions to the SWRCB Orders 98-05 and 98-07 riparian vegetation and geomorphic termination criteria. Prepared for the SWRCB and LADWP.

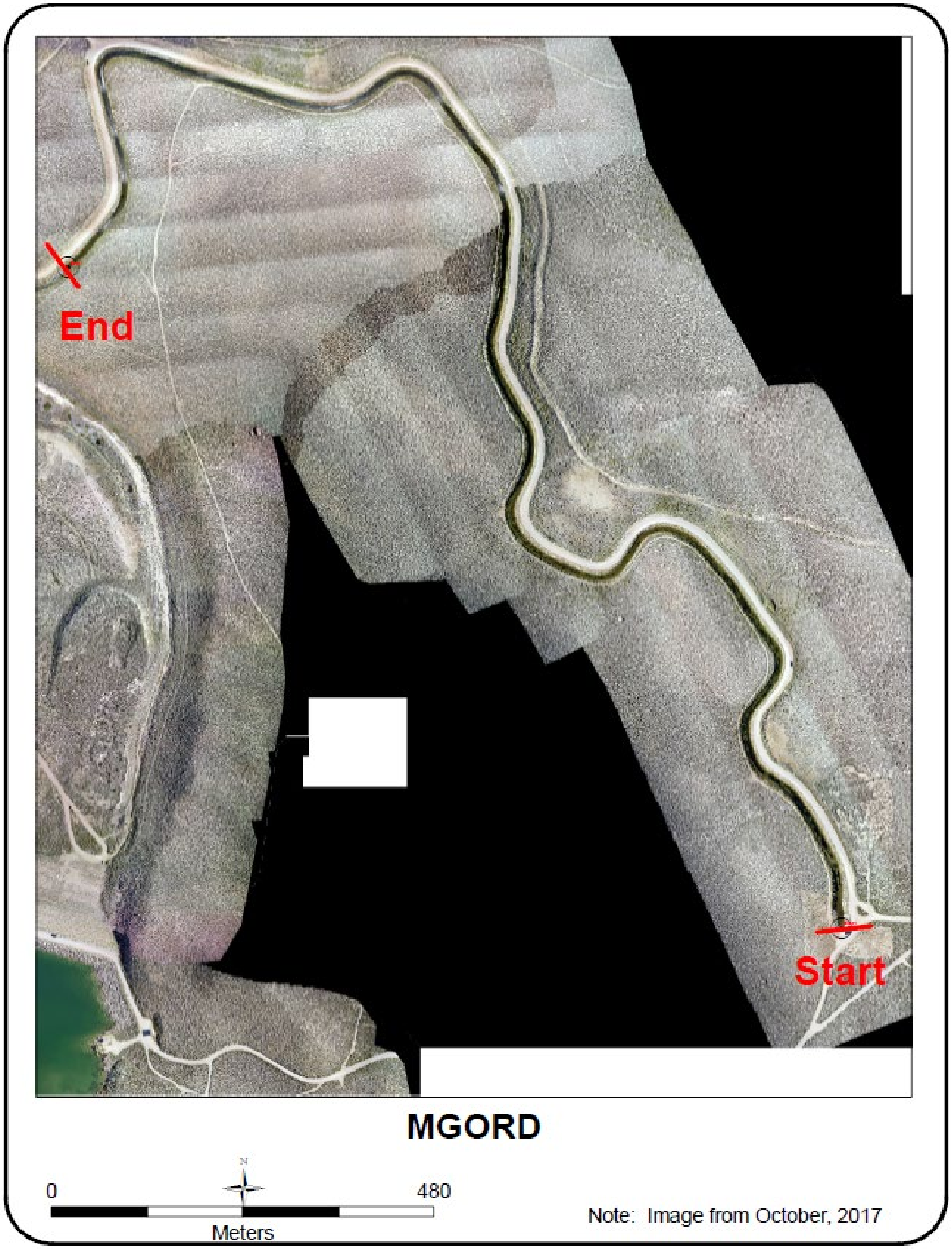
Werley, K.E., L. Wang and M. Mitro. 2007. Field-based estimates of thermal tolerance limits for trout: incorporating exposure time and temperature fluctuation. Transactions of the American Fisheries Society 136: 365-374.

Appendices for the 2022 Mono Basin Annual Fisheries Report

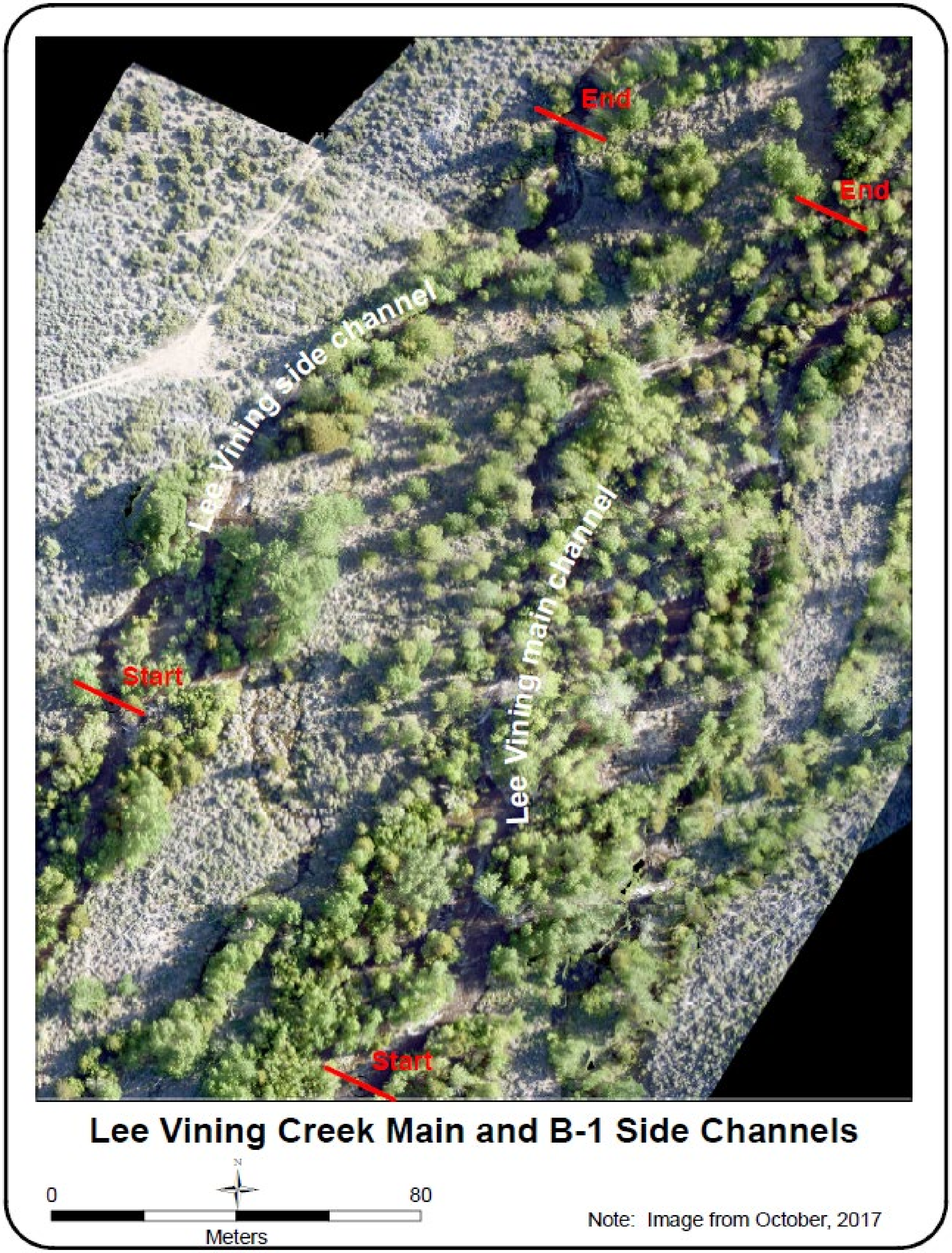
Appendix A: Aerial Photographs of Annual Sample Sites on Rush, Walker and Lee Vining Creeks


End
Start
Upper Rush Creek
N
0
160
Meters
Note: Image from October, 2017


End
Start
Rush Bottomlands
N
0
100
Meters
Note: Image from October, 2017


End
Start
MGORD
0
480
Meters
Note: Image from October, 2017


End
Start
Walker Creek
N
0
100
Meters
Note: Image from October, 2017


End
End
Lee Vining side channel
Lee Vining main channel
Start
Start
Lee Vining Creek Main and B-1 Side Channels
N
0
80
Meters
Note: Image from October, 2017

Appendix B: Tables of Numbers of Brown Trout and Rainbow Trout Implanted with PIT Tags (by sampling section) between 2009 and 2021
(Note: no tags implanted in 2013)

Table B-1. Total numbers of trout implanted with PIT tags during the 2009 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout	Number of Age-1 Brown Trout	Number of Age-0 Rainbow Trout	Number of Age-1 Rainbow Trout	Reach Totals
Rush Creek	Upper Rush	256	26	15	1	**298 Trout**
	Bottomlands	164	68	0	0	**232 Trout**
	County Road	108	29	0	0	**137 Trout**
	MGORD	54	642*	0	0	**696 Trout**
Lee Vining Creek	Main Channel	10	45	4	3	**62 Trout**
	Side Channel	5	0	0	1	**6 Trout**
Walker Creek	Above old 395	114	51	0	0	**165 Trout**
Totals:		**711**	**861**	**19**	**5**	**Total Trout: 1,596**

*Many of these MGORD trout were >age-1.

Table B-2. Total numbers of trout implanted with PIT tags during the 2010 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Reach Totals
Rush Creek	Upper Rush	242	11	4	0	**257 Trout**
	Bottomlands	284	3	0	0	**287 Trout**
	County Road	210	7	0	0	**217 Trout**
	MGORD	1	359*	0	12	**372 Trout**
Lee Vining Creek	Main Channel	24	8	0	1	**33 Trout**
	Side Channel	13	0	0	0	**13 Trout**
Walker Creek	Above old 395	81	14	0	0	**95 Trout**
Totals:		**855**	**402**	**4**	**13**	**Total Trout: 1,274**

*Many of these MGORD trout were >age-1.

Table B-3. Total numbers of trout implanted with PIT tags during the 2011 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Reach Totals
Rush Creek	Upper Rush	393	3	30	0	**426 Trout**
	Bottomlands	178	1	11	0	**190 Trout**
	County Road	196	1	6	0	**203 Trout**
	MGORD	8	142*	3	3	**156 Trout**
Lee Vining Creek	Main Channel	24	0	0	0	**24 Trout**
	Side Channel	11	14	0	0	**25 Trout**
Walker Creek	Above old 395	41	0	0	0	**41 Trout**
Totals:		**851**	**161**	**50**	**3**	**Total Trout: 1,065**

*Many of these MGORD trout were >age-1.

Table B-4. Total numbers of trout implanted with PIT tags during the 2012 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Reach Totals
Rush Creek	Upper Rush	117	1	2	0	**120 Trout**
	Bottomlands	110	1	6	0	**117 Trout**
	County Road	0	2	0	0	**2 Trout**
	MGORD	0	0	0	0	**0 Trout**
Lee Vining Creek	Main Channel	125	0	72	0	**197 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	60	0	0	0	**60 Trout**
Age Class Sub-totals:		**412**	**4**	**80**	**0**	**Total Trout: 496**

Table B-5 Total numbers of trout implanted with PIT tags during the 2014 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 Brown Trout (125-170 mm)	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 Rainbow Trout (125-170 mm)	Section Totals
Rush Creek	Upper Rush	243	86	1	0	**330 Trout**
	Bottomlands	34	43	0	0	**77 Trout**
	MGORD	13	125-199 mm = 60 Brown Trout ≥200 mm = 185 Brown Trout			**258 Trout**
Lee Vining Creek	Main Channel	127	103	5	22	**257 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	42	0	0	0	**42 Trout**
Age Class Sub-totals:		**459**	**232***	**6**	**22**	**Total Trout: 964**

*this sub-total excludes age-1 and older MGORD fish

Table B-6. Total numbers of trout implanted with PIT tags during the 2015 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	234	2*	7	0	**243 Trout**
	Bottomlands	167	3*	0	0	**170 Trout**
	MGORD	29	125-199 mm = 37 Brown Trout ≥200 mm = 83 Brown Trout (2 shed/new)			**149 Trout**
Lee Vining Creek	Main Channel	195	1*	0	0	**196 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	113	0	0	0	**113 Trout**
Age Class Sub-totals:		**738**	**6****	**7**	**0**	**Total Trout: 871**

*shed tag/new tag implanted **this sub-total excludes age-1 and older MGORD fish

Table B-7. Total numbers of trout implanted with PIT tags during the 2016 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	36	0	1	0	**37 Trout**
	Bottomlands	79	1*	0	0	**80 Trout**
	MGORD	4 BNT 1 RBT	125-199 mm = 9 BNT ≥200 mm = 154** BNT and 7 RBT			**175 Trout**
Lee Vining Creek	Main Channel	46	1*	0	0	**47 Trout**
	Side Channel	1	0	0	0	**1 Trout**
Walker Creek	Above old 395	228	1*	0	0	**229 Trout**
Age Class Sub-totals:		**394**	**166**	**2**	**7**	**Total Trout: 569**

*shed tag/new tag implanted **two of these BNT = shed tag/new tag implanted

Table B-8. Total numbers of trout implanted with PIT tags during the 2017 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	192	2*	14	0	**208 Trout**
	Bottomlands	34	0	0	0	**34 Trout**
	MGORD	38	0	2	0	**40 Trout**
Lee Vining Creek	Main Channel	31	0	0	0	**31 Trout**
	Side Channel	5	0	0	0	**5 Trout**
Walker Creek	Above old 395	0	0	0	0	**0 Trout**
Age Class Sub-totals:		**300**	**2**	**16**	**0**	**Total Trout: 318**

*shed tag/new tag implanted

Table B-9. Total numbers of trout implanted with PIT tags during the 2018 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	314	3*	72	1*	**390 Trout**
	Bottomlands	288	0	0	0	**288 Trout**
	MGORD	25	148**	1	7	**181 Trout**
Lee Vining Creek	Main Channel	87	0	8	0	**95 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	43	2*	0	0	**45 Trout**
Age Class Sub-totals:		**757**	**153**	**81**	**8**	**Total Trout: 999**

*shed tag/new tag implanted **≤250 mm in total length

Table B-10. Total numbers of trout implanted with PIT tags during the 2019 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	257	3*	28	0	**288 Trout**
	Bottomlands	152	3*	0	0	**155 Trout**
	MGORD	64	167** 8*	1	5	**245 Trout**
Lee Vining Creek	Main Channel	174	0	0	0	**174 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	137	1*	0	0	**138 Trout**
Age Class Sub-totals:		**784**	**182**	**29**	**5**	**Total Trout: 1,000**

*shed tag/new tag implanted **≤250 mm in total length

Table B-11. Total numbers of trout implanted with PIT tags during the 2020 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	242	1*	27	0	**270 Trout**
	Bottomlands	65	0	0	0	**65 Trout**
	MGORD	80	132** 1*	2	7	**222 Trout**
Lee Vining Creek	Main Channel	102	1*	0	0	**103 Trout**
	Side Channel	0	0	0	0	**0 Trout**
Walker Creek	Above old 395	92	4*	0	0	**96 Trout**
Age Class Sub-totals:		**581**	**139**	**29**	**7**	**Total Trout: 756**

*shed tag/new tag implanted **≤250 mm in total length

Table B-12. Total numbers of trout implanted with PIT tags during the 2021 sampling season, by stream, sample section, age-class and species.

Stream	Sample Section	Number of Age-0 Brown Trout (<125 mm)	Number of Age-1 and older Brown Trout	Number of Age-0 Rainbow Trout (<125 mm)	Number of Age-1 and older Rainbow Trout	Section Totals
Rush Creek	Upper Rush	148	1*	36	0	**185 Trout**
	Bottomlands	106	0	0	0	**106 Trout**
	MGORD	115	259** 1*	0	9	**384 Trout**
Lee Vining Creek	Main Channel	53	0	0	0	**53 Trout**
	Side Channel	17	0	0	0	**17 Trout**
Walker Creek	Above old 395	122	1*	0	0	**123 Trout**
Age Class Sub-totals:		**561**	**262**	**36**	**9**	**Total Trout: 868**

*shed tag/new tag implanted **≤250 mm in total length

Appendix C: Table of PIT-tagged Fish Recaptured during September 2022 Sampling

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2022 Recapture	Location of Initial Capture and Tagging	Comments
9/13/2022	BNT	186	71	989001038116907	Bottomlands	Bottomlands	
9/13/2022	BNT	156	37	989001038117067	Bottomlands	Bottomlands	
9/13/2022	BNT	166	44	989001038117099	Bottomlands	Bottomlands	
9/13/2022	BNT	142	30	989001039660965	Bottomlands	Bottomlands	
9/13/2022	BNT	164	51	989001039660981	Bottomlands	Bottomlands	
9/13/2022	BNT	165	46	989001039660989	Bottomlands	Bottomlands	
9/13/2022	BNT	156	38	989001039661018	Bottomlands	Bottomlands	
9/13/2022	BNT	164	42	989001039661143	Bottomlands	Bottomlands	
9/13/2022	BNT	163	44	989001039661278	Bottomlands	Bottomlands	
9/13/2022	BNT	161	43	989001039661349	Bottomlands	Bottomlands	
9/13/2022	BNT	161	39	989001039661390	Bottomlands	Bottomlands	
9/20/2022	BNT	236	116	989001038116856	Bottomlands	Bottomlands	
9/20/2022	BNT	202	84	989001038116947	Bottomlands	Bottomlands	
9/20/2022	BNT	169	44	989001039660966	Bottomlands	Bottomlands	
9/20/2022	BNT	152	35	989001039660967	Bottomlands	Bottomlands	
9/20/2022	BNT	153	35	989001039660971	Bottomlands	Bottomlands	
9/20/2022	BNT	160	35	989001039660974	Bottomlands	Bottomlands	
9/20/2022	BNT	204	86	989001039661035	Bottomlands	Bottomlands	
9/20/2022	BNT	165	42	989001039661264	Bottomlands	Bottomlands	
9/20/2022	BNT	197	79	989001039661272	Bottomlands	Bottomlands	
9/20/2022	BNT	157	39	989001039661321	Bottomlands	Bottomlands	
9/20/2022	BNT	168	48	989001039661334	Bottomlands	Bottomlands	
9/20/2022	BNT	172	59	989001039661357	Bottomlands	Bottomlands	
9/14/2022	BNT	237	127	989001038116541	Upper Rush	Upper Rush	
9/14/2022	BNT	157	38	989001038116661	Upper Rush	Upper Rush	
9/14/2022	BNT	169	48	989001038116886	Upper Rush	Upper Rush	
9/14/2022	BNT	176	61	989001039661000	Upper Rush	Upper Rush	
9/14/2022	BNT	165	43	989001039661042	Upper Rush	Upper Rush	
9/14/2022	BNT	161	41	989001039661270	Upper Rush	Upper Rush	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2022 Recapture	Location of Initial Capture and Tagging	Comments
9/14/2022	BNT	189	66	989001039661292	Upper Rush	Upper Rush	
9/14/2022	**RBT**	188	74	989001039661301	Upper Rush	Upper Rush	
9/14/2022	BNT	168	46	989001039661344	Upper Rush	Upper Rush	
9/14/2022	BNT	160	38	989001039661400	Upper Rush	Upper Rush	
9/14/2022	BNT	172	56	989001039661414	Upper Rush	Upper Rush	
9/14/2022	BNT	151	34	989001039661460	Upper Rush	Upper Rush	
9/14/2022	BNT	237	139	989001039661673	Upper Rush	Upper Rush	Shed tag, new tag implanted
9/14/2022	BNT	203	88	989001039661778	Upper Rush	Upper Rush	Shed tag, new tag implanted
9/21/2022	BNT	340	289	989001006111571	Upper Rush	Upper Rush	
9/21/2022	BNT	231	119	989001038116544	Upper Rush	Upper Rush	
9/21/2022	BNT	217	100	989001038116723	Upper Rush	Upper Rush	
9/21/2022	BNT	155	37	989001038116912	Upper Rush	Upper Rush	
9/21/2022	BNT	186	60	989001039660972	Upper Rush	Upper Rush	
9/21/2022	BNT	189	62	989001039661006	Upper Rush	Upper Rush	
9/21/2022	BNT	165	43	989001039661289	Upper Rush	Upper Rush	
9/21/2022	BNT	166	40	989001039661431	Upper Rush	Upper Rush	
9/21/2022	BNT	185	56	989001039661440	Upper Rush	Upper Rush	
9/21/2022	BNT	178	52	989001039661443	Upper Rush	Upper Rush	
9/21/2022	BNT	225	104	989001042091176	Upper Rush	Upper Rush	Shed tag, new tag implanted
9/21/2022	BNT	167	43	989001042091244	Upper Rush	Upper Rush	Shed tag, new tag implanted
9/15/2022	BNT	294	276	989001038117115	MGORD	MGORD	MORT, tag removed
9/22/2022	BNT	331	406	989001038117021	MGORD	MGORD	
9/22/2022	BNT	294	247	989001039661097	MGORD	MGORD	
9/22/2022	BNT	220	110	989001039661364	MGORD	MGORD	
9/22/2022	BNT	287	234	989001039661571	MGORD	MGORD	
9/22/2022	BNT	220	110	989001039661576	MGORD	MGORD	
9/22/2022	BNT	276	220	989001039661599	MGORD	MGORD	
9/22/2022	BNT	313	297	989001039661637	MGORD	MGORD	
9/19/2022	BNT	221	65	989001028114180	Walker Creek	Walker Creek	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2022 Recapture	Location of Initial Capture and Tagging	Comments
9/19/2022	BNT	205	75	989001028114224	Walker Creek	Walker Creek	
9/19/2022	BNT	186	64	989001031371657	Walker Creek	Walker Creek	
9/19/2022	BNT	192	69	989001031371667	Walker Creek	Walker Creek	
9/19/2022	BNT	196	71	989001031372378	Walker Creek	Walker Creek	
9/19/2022	BNT	198	67	989001031372382	Walker Creek	Walker Creek	
9/19/2022	BNT	196	55	989001031372383	Walker Creek	Walker Creek	
9/19/2022	BNT	180	50	989001031372417	Walker Creek	Walker Creek	
9/19/2022	BNT	206	73	989001031372427	Walker Creek	Walker Creek	
9/19/2022	BNT	207	77	989001031372431	Walker Creek	Walker Creek	
9/19/2022	BNT	195	73	989001031372435	Walker Creek	Walker Creek	
9/19/2022	BNT	215	94	989001031372450	Walker Creek	Walker Creek	
9/19/2022	BNT	205	72	989001038117262	Walker Creek	Walker Creek	
9/19/2022	BNT	165	39	989001038117264	Walker Creek	Walker Creek	
9/19/2022	BNT	151	34	989001038117266	Walker Creek	Walker Creek	
9/19/2022	BNT	176	50	989001038117279	Walker Creek	Walker Creek	
9/19/2022	BNT	162	38	989001038117280	Walker Creek	Walker Creek	
9/19/2022	BNT	164	38	989001038117290	Walker Creek	Walker Creek	
9/19/2022	BNT	177	51	989001038117301	Walker Creek	Walker Creek	
9/19/2022	BNT	179	51	989001038117309	Walker Creek	Walker Creek	
9/19/2022	BNT	172	43	989001038117314	Walker Creek	Walker Creek	
9/19/2022	BNT	183	56	989001038117328	Walker Creek	Walker Creek	
9/19/2022	BNT	189	58	989001038117355	Walker Creek	Walker Creek	
9/19/2022	BNT	170	47	989001038117357	Walker Creek	Walker Creek	
9/19/2022	BNT	133	20	989001039661169	Walker Creek	Walker Creek	
9/19/2022	BNT	143	25	989001039661174	Walker Creek	Walker Creek	
9/19/2022	BNT	143	28	989001039661179	Walker Creek	Walker Creek	
9/19/2022	BNT	119	15	989001039661189	Walker Creek	Walker Creek	
9/19/2022	BNT	140	26	989001039661192	Walker Creek	Walker Creek	
9/19/2022	BNT	165	39	989001039661193	Walker Creek	Walker Creek	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2022 Recapture	Location of Initial Capture and Tagging	Comments
9/19/2022	BNT	150	29	989001039661194	Walker Creek	Walker Creek	
9/19/2022	BNT	140	23	989001039661196	Walker Creek	Walker Creek	
9/19/2022	BNT	142	25	989001039661199	Walker Creek	Walker Creek	
9/19/2022	BNT	135	21	989001039661200	Walker Creek	Walker Creek	
9/19/2022	BNT	143	28	989001039661201	Walker Creek	Walker Creek	
9/19/2022	BNT	157	35	989001039661208	Walker Creek	Walker Creek	
9/19/2022	BNT	126	19	989001039661211	Walker Creek	Walker Creek	
9/19/2022	BNT	134	20	989001039661217	Walker Creek	Walker Creek	
9/19/2022	BNT	152	35	989001039661219	Walker Creek	Walker Creek	
9/19/2022	BNT	126	17	989001039661220	Walker Creek	Walker Creek	
9/19/2022	BNT	154	31	989001039661231	Walker Creek	Walker Creek	MORT, tag removed
9/19/2022	BNT	169	40	989001039661234	Walker Creek	Walker Creek	
9/19/2022	BNT	136	21	989001039661235	Walker Creek	Walker Creek	
9/19/2022	BNT	122	16	989001039661237	Walker Creek	Walker Creek	
9/19/2022	BNT	145	25	989001039661238	Walker Creek	Walker Creek	
9/19/2022	BNT	131	22	989001039661249	Walker Creek	Walker Creek	
9/19/2022	BNT	140	26	989001039661251	Walker Creek	Walker Creek	
9/19/2022	BNT	127	20	989001039661257	Walker Creek	Walker Creek	
9/19/2022	BNT	142	26	989001039661259	Walker Creek	Walker Creek	
9/19/2022	BNT	154	33	989001039661322	Walker Creek	Walker Creek	
9/19/2022	BNT	138	22	989001039661323	Walker Creek	Walker Creek	
9/19/2022	BNT	151	31	989001039661333	Walker Creek	Walker Creek	
9/19/2022	BNT	114	13	989001039661347	Walker Creek	Walker Creek	
9/19/2022	BNT	140	23	989001039661450	Walker Creek	Walker Creek	
9/16/2022	BNT	161	40	989001006111367	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	277	219	989001028114728	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	226	132	989001031371963	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	158	39	989001038117129	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	230	114	989001038117179	Lee Vining Ck	Lee Vining Ck	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2022 Recapture	Location of Initial Capture and Tagging	Comments
9/16/2022	BNT	183	64	989001038117193	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	145	37	989001039661062	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	149	32	989001039661715	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	166	46	989001039661730	Lee Vining Ck	Lee Vining Ck	
9/16/2022	BNT	169	45	989001039661736	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	162	39	989001006111367	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	231	130	989001031371963	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	246	147	989001031372031	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	191	66	989001038117162	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	213	98	989001038117230	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	221	118	989001038117243	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	173	48	989001039661731	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	166	40	989001039661746	Lee Vining Ck	Lee Vining Ck	
9/23/2022	BNT	182	59	989001042090998	Lee Vining Ck	Lee Vining Ck	Shed tag, new tag implanted

Section II. Mono Basin Waterfowl Habitat Restoration Program 2022 Monitoring Report

Prepared by Deborah House, Mono Basin Waterfowl Program Director

and Motoshi Honda, Watershed Resources Specialist

Mono Basin Waterfowl Habitat Restoration Program

Statement of Compliance and Summary of 2022 Monitoring

Prepared for the State Water Resources Control Board

The Los Angeles Department of Water and Power (LADWP) conducts monitoring in compliance with the 1996 Mono Basin Waterfowl Habitat Restoration Plan and the 1998 State Water Resources Control Board Order WR 98-05. LADWP completed the following monitoring tasks in 2022:

Hydrology:
- Monthly Mono Lake elevation readings
- Daily stream flows in Rush, Lee Vining, Parker and Walker Creeks

Limnology:
- Meteorological, physical/chemical, phytoplankton, and brine shrimp population monitoring

Saltcedar Eradication
- Coordinated with California State Parks to report saltcedar eradication results

Waterfowl Populations:
- Summer ground surveys and documentation of habitat use
- Fall surveys at Mono Lake, Bridgeport Reservoir and Crowley Reservoir
- Still-image photography of waterfowl habitats at Mono Lake, Bridgeport Reservoir and Crowley Reservoir

The *Mono Basin Waterfowl Habitat Restoration Program 2022 Monitoring Report* included herein provides detailed discussion of monitoring methods, results, and discussion for each component. Below are brief summaries of the results of the 2022 monitoring year.

Hydrology

Runoff during the 2021-2022 Water Year was 60,928 acre-feet, or 51% of the long-term average. Mono Lake experienced an overall decrease in lake level as compared to 2021. The peak lake level in 2022 of 6,379.6 feet occurred in February, followed by a continuous decline through the remainder of the year, with no runoff-associated increase in lake level due to the dry conditions. In December 2022, Mono Lake was at 6,378.1 feet, or 1.5 feet lower than in December 2021.

Limnology

The mean Artemia population in 2022 was 17,182 m^{-2}. This value is 28% lower than the post-meromictic population peak of 2021. The *Artemia* population centroid was 223 days. Lake transparency remained below 1 meter in all months for the third year in row. The winter of 2021-22 was slightly warmer but dry, and the summer of 2022 was warm and wet. Several strong, medium- and short-term trends in air temperature are seen in the data, however, the only long-term trend apparent is of increasing average summer minimum temperatures.

Saltcedar Eradication

The saltcedar eradication program being conducted by California State Parks has been very effective. The five sites treated in 2022 represents a small number compared to previous years.

Waterfowl Populations

Following two above-average years of waterfowl breeding activity, the continuing drop in lake level resulted in reduced waterfowl breeding at Mono Lake in 2022. The breeding waterfowl population at Mono Lake in 2022 was 241, or approximately 120 pairs, and the productivity of dabbling ducks was 44 broods. In 2022, breeding activity was concentrated along the northwest shore at DeChambeau, Mill and Wilson Creeks, and at Simons Spring where conditions were most favorable, and generally associated with nearshore water features, primarily freshwater ponds, freshwater outflow areas around the lake. Waterfowl totals and broods at the Restoration Ponds, were still below the long-term average. The County Ponds continued to be dry, thus reducing available habitat as compared to previous years.

Fall waterfowl totals at Mono Lake were comparable to the long-term mean. Fall waterfowl totals at Bridgeport were below, but were more than double the long-term average at Crowley Reservoir. Although Bridgeport and Crowley support larger and more diverse waterfowl populations, Mono Lake supports a significant proportion of the local Northern Shoveler and Ruddy Duck fall migratory populations in Mono County.

Recommendations

Based on the results of these monitoring programs, the following are recommendations for the Mono Basin Waterfowl Habitat Restoration Program. These recommendations are further described and discussed in the report:

- Continue to implement measures to support lake level recovery
- Enhance and restore the functioning of the Restoration Ponds
- Investigate algal community dynamics
- Conduct a second waterfowl time budget study
- Reinstate the vegetation monitoring program in lake-fringing wetlands and riparian areas.
- Reinstate annual restoration meetings

Deborah House

Deborah House

Mono Basin Waterfowl Monitoring Program Director

May 1, 2023

Mono Basin Waterfowl Habitat Restoration Program 2022 Monitoring Report

Prepared by:
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Los Angeles Department of Water and Power
Bishop, CA 93514

Prepared for the State Water Resources Control Board
and Los Angeles Department of Water and Power
April 2023

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EXECUTIVE SUMMARY

In 1983, National Audubon Society v. Superior Court resulted in the California State Water Resources Control Board (SWRCB) reevaluating the effect of water diversions by the City of Los Angeles (City) on the public trust values of Mono Lake. SWRCB Decision 1631, signed in 1994, amended the City's water rights, establishing instream flow requirements for the Mono Basin creeks and placing limitations on water exports from the Mono Basin. Order WR 98-05 (SWRCB 1998) directed the Los Angeles Department of Water and Power (LADWP) to implement waterfowl habitat restoration measures and monitoring to mitigate the loss of waterfowl habitat in the Mono Basin from diversions. This report summarizes the results of monitoring conducted in 2022 under the Mono Basin Waterfowl Habitat Restoration Plan (Plan) (LADWP 1996a), including hydrology, limnology and secondary producers, and waterfowl population surveys and studies.

Runoff during the 2021-2022 Water Year was 60,928 acre-feet, or 51% of the long-term average. Runoff in the Mono Basin was insufficient to maintain lake level in 2022 and Mono Lake experienced an overall decrease in lake level as compared to 2021. The peak lake level in 2022 of 6,379.6 feet was in February, followed by a continuous decline through the remainder of the year, and no runoff-associated increase in lake level was observed. The lake dropped to its lowest level in 2022 of 6,378.1 feet for November and December. At the final lake level read in December, Mono Lake had experienced a net decline of 1.5 feet in 2022, and was 1.4 feet lower than in December of 2021. Input from the two major tributaries (Rush and Lee Vining Creeks) in 2022 was 55,203 acre-feet, or 58% of the long-term mean.

The winter of 2021-22 was slightly warmer but dry, while the summer of 2022 was warm and wet. Several strong, medium- and short-term trends in air temperature are seen in the data, however, only one long-term trend is apparent. Medium-term trends (over the last 40-43 years) include increases in the minimum and maximum temperatures observed in both winter and summer. Over a shorter time period of 13-17 years, winter and summer maximum temperatures have been increasing at an even more substantial rate. When temperature data going back to 1951 is incorporated, the only clear long-term trend is for increasing average summer minimum temperatures. The medium- and short-term trends observed are consistent with conditions expected under various climate change regimes, however these trends are not yet completely reflected in the long-term patterns.

Epilimnetic water temperature in 2022 was above the long-term average in spring and summer due to warmer than normal monthly ambient temperatures, while hypolimnetic water remained above the long term-average between May and November. The September and October hypolimnetic water temperatures were the highest recorded since 1991.

In 2022, Mono Lake remained monomictic, following the end of the previous period of meromixis in 2020. The mean lakewide *Artemia* population in 2022 was 17,182 m^{-2}. This value was approximately 28% lower than 2021, when a “post-meromictic” population peak was observed. The centroid, or center of temporal distribution of adult *Artemia*, was August 20, or day 223. A significant number of adults did not appear until June. Numbers peaked in July, and remained elevated through August, thus shifting the centroid into late August.

Lake transparency remained below 1 meter in all months for the third year in row. A recent notable change is the sustained high abundance of phytoplankton. In prior years, Mono Lake transparency improved in the summer months, frequently exceeding 10 meter due to intensified *Artemia* grazing in summer. Algal food sources for *Artemia* appear to be readily available, but, as suggested by lake transparency, are not being controlled by *Artemia*.

Shoreline conditions for waterfowl are dynamic at Mono Lake, and are influenced by lake level, wave and wind action, and spring and other fresh water inflow, and now, grazing by feral horses. As is expected, the continued decline in lake level resulted in an overall increase in barren playa on shore, and fewer shoreline ponds. In 2022, waterfowl habitat conditions were still fairly good at Simons Springs, Wilson Creek, Mill Creek and DeChambeau Creeks due to factors such as the presence of spring flow to the lake shore, onshore ponds, and shallow feeding areas. Conditions were poor along the South Shore Lagoons due to vegetation encroachment and a lack of open water pond. Further drying of the deltas of Rush and Lee Vining Creeks was evident, however, transient conditions provided for excellent waterfowl foraging habitat at the delta of Rush Creek in mid-summer.

Grazing by feral horses was particularly heavy in the Warm Springs and Simons Spring areas. The intense grazing has had some interesting effects, at least in the short-term, on the conditions, and the dynamics of waterbird use, particularly in the Warm Springs area because of the reduction in dense mats of meadow and wetland vegetation previously in the area. The Warm Springs area continued to be very wet, with multiple shallow, open water ponds, attracting waterbirds to feed and shorebirds to attempt nesting in places previously unavailable because of dense cover.

In contrast to Mono Lake, the levels of Bridgeport and Crowley Reservoirs were higher in fall of 2022 as compared to 2021. Harmful algal blooms affected both reservoirs temporarily in early fall. At Bridgeport Reservoir the higher reservoir level resulted in moister conditions of the adjacent meadows and floodplain. At Crowley Reservoir, there was very heavy growth of widgeongrass (*Ruppia* sp.) likely contributing to the very high waterfowl numbers in 2022.

The saltcedar eradication program being conducted by California State Parks has been very effective. The five sites treated in 2022 represents a small number compared to previous years.

Following two above-average years of waterfowl breeding activity, the continuing drop in lake level resulted in reduced waterfowl breeding at Mono Lake in 2022. The breeding waterfowl population at Mono Lake in 2022 is estimated to have been 241, or approximately 120 pairs, which is significantly lower than the long-term mean of 308.1 +/-18.8 SE or 154 pairs. The 44 dabbling duck broods in 2022 was also below the long-term average of 48.6 +/-3.8. In 2022, breeding activity was concentrated along the northwest shore at DeChambeau, Mill and Wilson Creeks, and at Simons Spring where conditions were most favorable. Although conditions did not appear good for supporting much breeding activity, a large number of Gadwall broods were observed foraging at the mouth of Rush Creek in late July when protected conditions of low flow and shallow water created ideal waterfowl foraging conditions, at least temporarily. Most dabbling duck activity was concentrated in and around nearshore water features, primarily freshwater ponds, and freshwater outflow areas around the lake. At the Restoration Ponds, waterfowl totals and brood numbers continued to be below the long-term average. The County Ponds continued to be dry, thus reducing available habitat as compared to previous years.

Lake level has strongly influenced the size of the breeding waterfowl population at Mono Lake. Spring lake levels, particularly in April, have had the largest influence on the size of the breeding population. Habitat conditions in early spring may influence the breeding waterfowl population by influencing whether waterfowl pairs chose to settle and breed at Mono Lake. Annual brood numbers (i.e., productivity) have been strongly influenced by the June lake level, at least above a threshold of 6,382 feet. Below 6,382 feet, there has been no significant effect of lake level, and a smaller breeding population has persisted. *Artemia* biomass was not found to be correlated with either breeding waterfowl totals, or annual brood numbers.

A total of 13 waterfowl species and 18,351 individuals were detected during the five Mono Lake fall surveys and the estimated total waterfowl for 2022 is 25,261 +/-2,691

SE, which does not differ from the long-term mean. Unlike breeding waterfowl populations at Mono Lake, fall migratory waterfowl have not been directly influenced by lake level. Fall migratory populations have been positively correlated with the abundance of *Artemia*, and fall waterfowl numbers have been higher on average during years when Mono Lake is monomictic and lower under meromictic conditions. Similar to the long-term trend observed in the *Artemia* population, the fall waterfowl data suggests a weak linear downward trend in total fall waterfowl.

Despite its much larger size, Mono Lake supports fewer total waterfowl than either Bridgeport or Crowley Reservoirs. Waterfowl totals at Mono Lake have accounted for 24% of all waterfowl at the three survey areas. Although Bridgeport and Crowley support larger and more diverse waterfowl populations, Mono Lake supports a significant proportion of the local Northern Shoveler and Ruddy Duck fall migratory populations. In 2022, fall waterfowl totals at Mono Lake were comparable to the long-term mean, and Bridgeport Reservoir was significantly lower than the long-term means. The total number of waterfowl at Crowley Reservoir was more than double the long-term mean.

With the exception of the Ruddy Duck, most waterfowl use at Mono Lake occurs in lake-fringing ponds, or very near to shore. The near shore areas used by waterfowl are generally shallow, have gentle offshore gradients, and freshwater spring, creek, or brackish water input. Mono Lake is deep, highly saline, with limited shallow shoreline areas. These features limit the habitat quality for waterfowl, and may ultimately limit recovery of waterfowl populations.

Of the restoration measure outlined in Order 98-05, lake level recovery remains the single most important measure for improving and maintaining waterfowl habitat. Higher lake levels - at least within the range of levels observed – appear to improve breeding habitat conditions, and increase brood production. Increased lake levels have resulted in more shoreline ponds and greater connectivity between nesting habitats and preferred foraging areas. As shown by the fall waterfowl data, however, lake level alone may not enhance use by migrating waterfowl, as the biomass of secondary producers such as *Artemia* is a key variable, potentially swamping out any effect of lake level change. Mono Lake must continue to remain productive in order to continue to support waterfowl populations, and improving our understanding of how to support lake productivity should be considered.

In addition to continuing to implement measures to support lake level recovery, the next most viable project to improve waterfowl habitat in the Mono Basin is to enhance and restore the functioning of the Restoration Ponds. Although due to their small size, the total number of waterfowl that could be supported by the Restoration Ponds is just a fraction of that occurring on Mono Lake, there are management strategies, repairs, and improvements that would increase waterfowl use. The most basic of improvements would be to restore water delivery to the County Ponds, which have been dry for several years. The second would be to implement seasonal or rotational flooding regime to enhance forage production for waterfowl, while continuing to provide waterfowl habitat year-round at the ponds. In addition, we also recommend the Mono Basin Waterfowl Director work with partners restoring the functioning of the DeChambeau Ponds on ensuring that monitoring efforts are not being duplicated.

Because maintaining the long-term productivity of Mono Lake is important for the wildlife that depend on it, consideration should be given to conducting more detailed studies and monitoring of the algal community, particularly as it relates to the *Artemia* population. Notable changes in Mono Lake limnology include a sustained high abundance of phytoplankton starting in 2015, significant decreases in transparency during the summer months, and a weak, downward trend in the *Artemia* population. The extensive limnological dataset has limited information on specifics of the algal community that may help interpret these more recent trends in Mono Lake ecology.

In addition to projects and actions directly related to habitat improvement, there are studies that would improve our understanding of variables influencing waterfowl use of Mono Lake. Firstly, we recommend that the second year of the waterfowl time budget study, as required by Order 98-05, be completed. A time budget study allows for the determination of the relative importance of different shoreline sites for migratory waterfowl, and would provide insight into the importance of the various habitat types for feeding, resting, or drinking. Secondly, document invertebrate diversity and abundance at various shoreline locations and waterfowl habitats (e.g. spring outflow, stream outflow sites, fresh and brackish ponds) to understand the nature of these resources as they relate to waterfowl.

Due to the expanding range of feral horses in the Mono Basin, we recommended that the wetland and riparian vegetation monitoring program be reinstituted. In 2021, the wetland transects were prioritized since most of the horse activity had been seen at Warms Springs and Simons Springs. However, horses have been observed near the Rush Creek delta on occasion since the summer of 2021, thus continuing the transects in Rush and Lee Vining Creek deltas is prudent. Following the previous monitoring schedule of every 5 years, we recommend conducting the wetland and riparian transects in 2026.

Finally, I recommend that annual Mono Basin restoration meetings focused on the science and monitoring outcomes be reinstated in order to foster communication and knowledge sharing among the Mono Basin parties.

1.0 INTRODUCTION

Mono Lake is a large terminal saline lake at the western edge of the Great Basin in Mono County, California. The largest lake in Mono County, Mono Lake has an east-west dimension of 13 miles, a north-south dimension of over nine miles (Raumann et al. 2002), and a circumference of approximately 40 miles. With an average depth of over 60 feet and a maximum depth of approximately 150 feet (Russell 1889), Mono Lake is a large, moderately deep terminal saline lake (Jellison and Melack 1993, Melack 1983). The deepest portions of the lake are found south and east of Paoha Island in the Johnson and Putnam Basins, respectively (Raumann et al. 2002). Shallower water and a gently sloping shoreline are more typical of the north and east shores (Vorster 1985, Raumann et al. 2002).

Mono Lake is widely known for its value to migratory waterbirds, supporting up to 30% percent of the North American Eared Grebe (*Podiceps nigricollis*) population, the largest nesting population of California Gull (*Larus californicus*) in California (Winkler 1996), and up to 140,000 Wilson's (*Phalaropus tricolor*) and Red-necked Phalaropes (*P. lobatus*) during fall migration (Jehl 1986, Jehl 1988).

Saline lakes are highly productive ecological systems (Jellison et al. 1998), however productivity is influenced by factors such as salinity, water depth, temperature, and water influx and evaporation on a seasonal, annual, and inter-annual basis. Saline lakes often respond rapidly to environmental changes, and alterations to the hydrological budget (Jehl 1988, Williams 2002). Water demands for agriculture, human development and recreation, as well as changes in climate are impacting saline lakes globally (Wurtsbaugh et al. 2017).

In 1941, the City of Los Angeles (City) began diverting water from Lee Vining Creek, Rush Creek, Walker Creek, and Parker Creek for municipal water supply. From 1941-1970, when the City was exporting an annual average of 56,000 acre-feet, the elevation of Mono Lake dropped over 29 feet. In 1970, the completion of the second aqueduct in Owens Valley expanded the capacity of the Los Angeles Aqueduct system, resulting in increased diversions, frequent full diversion of flows from Lee Vining, Walker, Parker and Rush Creek and a drying of the creek channels (SWRCB 1994). From 1970 to 1989, Mono Lake dropped another 12.6 feet as yearly exports averaged 82,000 acre-feet, with a peak export of 140,756 acre-feet in 1979. The elevation of Mono Lake dropped to a record low of 6,372.0 feet above mean sea level in 1982. In 1979, the National Audubon Society filed suit with the Superior Court of California against the City (National Audubon Society v. Superior Court), arguing that the diversions in the Mono

Basin were resulting in environmental damage and were a violation of the Public Trust Doctrine.

After a series of lawsuits and extended court hearings, the State Water Resources Control Board (SWRCB) amended the City's water rights with the Mono Lake Basin Water Right Decision 1631 (Decision 1631) (SWRCB 1994). Decision 1631 established instream flow requirements for the Mono Basin creeks for fishery protection, and placed limitations on water exports from the basin until the surface elevation of Mono Lake reached 6,391 feet. In addition to diversion reductions, Decision 1631 required LADWP to conduct restoration and monitoring of Mono Lake ecological resources.

SWRCB Order 98-05, adopted on September 2, 1998, defined waterfowl restoration measures and elements of a waterfowl habitat monitoring program for Mono Lake. The Mono Basin Waterfowl Habitat Monitoring Plan has been implemented continuously since. In 2017, LADWP conducted a comprehensive analysis of restoration actions taken under Order 98-05 since its inception. The *Mono Basin Waterfowl Habitat Restoration Program Periodic Overview Report* (LADWP 2018) summarized the results of this analysis and included recommendations to increase effectiveness of various monitoring tasks, and to reduce the cost of the monitoring project while continuing to provide indices to track restoration progress. Some of the recommendations set forth in the 2018 report have been implemented, although changes to the waterfowl and limnology monitoring programs in place were not implemented.

The SWRCB approved Amended Water Rights Licenses 10191 and 10192 for LADWP on October 1, 2021. Conditions 21 and 22 of these amended licenses define how the Mono Basin Waterfowl Habitat Restoration Program will be managed in the future, including some revisions to previous orders. This Waterfowl Habitat Restoration report summarizes the results of monitoring conducted in 2022 pursuant to Restoration Order 98-05 and the amended licenses.

2.0 WATERFOWL HABITAT RESTORATION MEASURES

The SWRCB issued Order 98-05 in 1998, defining waterfowl habitat restoration measures and associated monitoring to be conducted in compliance with Decision 1631. The export criteria of Decision 1631 were developed to result in an eventual long-term average water elevation of Mono Lake of 6,392 feet (SWRCB 1994). In determining the most appropriate water level for the protection of public trust resources at Mono Lake, the SWRCB recognized that there was no single lake elevation that would maximize protection of, and accessibility to, all public trust resources. Decision 1631 stated that maximum restoration of waterfowl habitat would require a lake elevation of 6,405 feet. Raising the lake elevation to 6,405 feet however, would have precluded use of any water from the Mono Basin by the City for municipal needs, and inhibited public access to South Tufa, the most frequently visited tufa site. Furthermore, it was determined that a lower target lake elevation of 6,390 feet would accomplish some waterfowl habitat restoration, and that there were opportunities to restore additional habitat, mitigating the overall loss as a result the target being set below 6,405 feet. A target level of 6,392 feet was ultimately established as this level would restore some waterfowl habitat, allow continued access to South Tufa, and ensure compliance with federal air quality standards.

As noted in Order 98-05, and recognized in the restoration plans, the most important waterfowl habitat restoration measures were maintaining an average lake elevation of 6,392 feet, and restoring perennial flow to streams tributary to Mono Lake. In addition to lake level recovery, and stream restoration, Order 98-05 included the following measures to be undertaken by LADWP:

1. reopen distributaries in the Rush Creek bottomlands,
2. provide financial assistance for the restoration of waterfowl habitat at the County Ponds and Black Point or other lake-fringing wetland area,
3. participate in a prescribed burn program subject to applicable permitting and environmental review requirements;
4. participate in exotic species control efforts if an interagency program is established in the Mono Basin; and
5. develop a comprehensive waterfowl and waterfowl habitat monitoring program.

Table 2.1 describes each restoration measure required under Order 98-05, providing a brief discussion on LADWP’s progress to date and the current status. Some of these projects have been completed, some are ongoing, and others have been determined by the stakeholders to be unfeasible. More details regarding these restoration measures can be found in the *Periodic Overview Report* (LADWP 2018).

Table 2.1. Mono Basin Waterfowl Habitat Restoration Activities

Mono Basin Waterfowl Habitat Restoration Activities *(as described in SWRCB Order 98-05 and the Mono Basin Waterfowl Habitat Restoration Plan dated February 29, 1996, where relevant)*				
Activity	**Goal**	**Description**	**Progress to Date**	**Status**
Rewatering Distributary Channels to Rush Creek (below the Narrows)	To restore waterfowl and riparian habitat in the Rush Creek bottomlands.	Rewater the Channel 4bii complex	Rewatering of side channels was evaluated in 2002 by the State Appointed Stream Scientists and LADWP. At that time, rewatering of Channel 4bii was deferred because natural revegetation of riparian and wetland species was occurring. The area was reevaluated in 2007 and rewatering was completed in March 2007.	Complete
		Rewater the Channel 8 complex, unplugged lower section	In 2002, the sediment plug was removed and the Channel 8 complex widened at the upstream end. In contrast to rewatering for constant flow, the final design called for flows overtopping the bank and flowing into Channel 8 at approximately 250 cfs and above. Woody debris was spread and willows were transplanted along new banks following excavation. Further rewatering of Rush Creek Channel 8 complex was deferred by the Stream Scientists. Final review was conducted by McBain and Trush (2010). After presentation of the final review, LADWP followed the recommendations of the Stream Scientists and SWRCB approved the plan. Channel 8 was rewatered in March 2007.	Complete
		Rewater the Channel 10 complex	Rewatering of side channels was evaluated in 2002 by the State Appointed Stream Scientists and LADWP. This evaluation concluded that rewatering the Channel 10 complex would result in detrimental impacts to reestablished fishery and riparian habitats. Therefore, there have been no further actions taken to rewater this channel. Project is complete.	Complete

Mono Basin Waterfowl Habitat Restoration Activities, cont. *(as described in SWRCB Order 98-05 and the Mono Basin Waterfowl Habitat Restoration Plan dated February 29, 1996, where relevant)*				
Activity	**Goal**	**Description**	**Progress to Date**	**Status**
Rewatering Distributary Channels to Rush Creek (below the Narrows)	To restore waterfowl and riparian habitat in the Rush Creek bottomlands.	Rewater Channel 11, unplugged lower portion	Rewatering of side channels was evaluated in 2002 by the State Appointed Stream Scientists and LADWP. At that time, it was determined that there would be little benefit to unplugging Channel 11 compared to the impacts to reestablished riparian vegetation from mechanical intrusion. Further evaluation was conducted by the Stream Scientists. After presentation of the final review, LADWP followed the recommendations of the Stream Scientists not to rewater the channel. This item is now approved by SWRCB and was therefore considered complete in 2008.	Complete
		Rewater the Channel 13 complex	Rewatering of side channels was evaluated in 2002 by the State Appointed Stream Scientists and LADWP. At that time, it was determined that Channel 13 would not be stable or persist in the long term and riparian vegetation was already rapidly regenerating in this reach. Therefore, there have been no further actions taken to rewater Channel 13. Project is considered complete.	Complete

Mono Basin Waterfowl Habitat Restoration Activities, cont. *(as described in SWRCB Order 98-05 and the Mono Basin Waterfowl Habitat Restoration Plan dated February 29, 1996, where relevant)*				
Activity	**Goal**	**Description**	**Progress to Date**	**Status**
Financial Assistance to United States Forest Service (USFS) for Waterfowl Habitat Improvement Projects at County Ponds and Black Point areas	To support repairs and improvement of infrastructure on USFS land in the County Ponds area.	Upon request of the USFS, Licensee (LADWP) shall provide financial assistance in an amount up to $250,000 for repairs and improvements to surface water diversion and distribution facilities and related work to restore or improve waterfowl habitat on USFS land in the County Ponds area.	LADWP was to make available a total of $275,000 for waterfowl restoration activities in the Mono Basin per Order 98-05. This money was to be used by the USFS if they requested the funds by December 31, 2004. Afterwards, any remaining funds are to be made available to any party wishing to do waterfowl restoration in the Mono Basin after SWRCB review. This funding allocation has been included in Section 21.a of Amended Licenses 10191 and 10192 to be administered by the Mono Basin Monitoring Administration Team (MAT).	In Progress
	To support waterfowl habitat improvement projects on USFS land in the Black Point area.	Upon request of the USFS, Licensee (LADWP) shall provide financial assistance in an amount up to $25,000 for waterfowl habitat improvements on USFS land in the Black Point area.		

Mono Basin Waterfowl Habitat Restoration Activities, cont. *(as described in SWRCB Order 98-05 and the Mono Basin Waterfowl Habitat Restoration Plan dated February 29, 1996, where relevant)*				
Activity	**Goal**	**Description**	**Progress to Date**	**Status**
Prescribed Burn Program	To enhance lake-fringing marsh and seasonal wet meadow habitats for waterfowl	The licensee shall proceed with obtaining the necessary permits and approval for the prescribed burning program described in the Mono Basin Waterfowl Habitat Restoration Plan dated February 29, 1996 and provide the SWRCB a copy of any environmental documentation for the program. Following review of the environmental documentation, the SWRCB may direct Los Angeles to proceed with implementation of the prescribed burning program pursuant to D1631 and Order 98-05, or modify the program.	LADWP began a prescribed burn program with limited success. LADWP requested to remove this item from the requirements in 2002 and the SWRCB instead ruled that the prescribed burn program will be deferred until Mono Lake reaches the target elevation. Per Condition 21.b in Amended Licenses 10191 and 10192, when Mono Lake reaches an elevation of 6,391 feet, the SWRCB will consider the options and benefits of Licensee reactivating the prescribed waterfowl habitat burn program. If the program is reactivated, Licensee shall proceed with obtaining the necessary permits and approvals for the prescribed burning program.	Deferred
Saltcedar Eradication Program	To control non-native vegetation in the Mono Basin	In the event that an interagency program is established for the control or elimination of saltcedar or other non-native vegetation deemed harmful to waterfowl habitat in the Mono Basin, Licensee (LADWP) shall participate in that program and report any work it undertakes to control saltcedar or other non-native vegetation.	LADWP continues treatment of saltcedar as needed. Progress of the salt cedar eradication efforts is reported in the annual reports following the vegetation monitoring efforts. This item is carried over to Condition 21.c in Amended licenses 10191 and 10192.	Ongoing

3.0 WATERFOWL HABITAT RESTORATION MONITORING PROGRAM

The Plan and SWRCB Order WR 98-05 directed LADWP to conduct monitoring to assess the success of waterfowl habitat restoration efforts, evaluate the effects of changes in the Mono Lake area, and plan for future restoration activities. Components of the Mono Basin Waterfowl Habitat Monitoring Program (Program) include hydrology, limnology, the vegetation status of riparian and lake-fringing wetlands, and waterfowl population surveys. Table 3.1 provides a brief description of the monitoring components, their required frequency under the Plan and Order 98-05, and the dates that each monitoring task has been performed.

In 2022, monitoring conducted under the Program included lake elevation, stream flows, lake limnology and secondary producers, saltcedar eradication, waterfowl population surveys and aerial photography of waterfowl habitats. The remainder of this report provides a summary and discussion on the 2022 data collected under the Program.

Table 3.1. Mono Basin Habitat Restoration Monitoring Program

Mono Basin Habitat Restoration Monitoring Program *(as described in SWRCB Order 98-05 and the Waterfowl Habitat Restoration Plan dated February 29, 1996)*			
Monitoring Component	**Description**	**Required Frequency**	**Dates Monitoring Performed**
Hydrology	Lake Elevation	Weekly through one complete wet/dry cycle after the lake level has stabilized.	Monthly data collected 1936-present; ongoing
	Stream Flows	Daily through one complete wet/dry cycle after the lake level has stabilized.	Daily data collected 1935-present; ongoing
	Spring Surveys	Five-year intervals (August) through one complete wet/dry cycle after the lake level has stabilized.	1999, 2004, 2009, 2014, 2019; ongoing
Lake Limnology and Secondary Producers	Meteorological data, data on physical and chemical environment of the lake, phytoplankton, and brine shrimp population levels	Annually (monthly February-December) until the lake reaches a relatively stable level. LADWP will evaluate monitoring at that time and make a recommendation to the SWRCB whether or not to continue.	1987-present; ongoing
Vegetation Status in Riparian and Lake Fringing Wetland Habitats	Establishment and monitoring of vegetation transects and permanent photopoints in lake fringing wetlands	Five-year intervals or after extremely wet year events (whichever comes first) until 2014. Due to heavy feral horse in some areas, reinstate the program and monitor all sites again in 2026. Reevaluate the need to continue monitoring after 2026 data are evaluated.	2000, 2005, 2010, 2015; ongoing
	Aerial photographs of lake fringing wetlands and Mono Lake tributaries	Five-year intervals until target lake elevation of 6,392 feet is achieved.	1999, 2005, 2009, 2014; ongoing

Table 3.1 Continued

Mono Basin Habitat Restoration Monitoring Program *(as described in SWRCB Order 98-05 and the Waterfowl Habitat Restoration Plan dated February 29, 1996)*			
Monitoring Component	**Description**	**Required Frequency**	**Dates Monitoring Performed**
Waterfowl Population Surveys and Studies	Fall aerial counts	Two counts conducted every other year October 15-November 15. All waterfowl population survey work will continue until 2014, through one complete wet/dry cycle after the target lake elevation of 6,392 feet is achieved. Since 2002, six fall counts have been conducted annually at Mono Lake, Bridgeport Reservoir and Crowley Reservoir. Helicopter, boat and ground counts were conducted in 2022 due to lack of fixed wing services.	Annually; ongoing
	Aerial photography of waterfowl habitats	Conducted during or following one fall aerial count.	Annually; ongoing
	Ground counts	Total of eight ground counts annually (two in summer, six in fall). All waterfowl population survey work will continue until 2014, or through one complete wet/dry cycle after the target lake elevation of 6,392 feet is achieved. Since 2002, three summer ground counts have been conducted. Fall ground counts were replaced with six aerial counts.	Annually; ongoing
	Waterfowl time activity budget study	To be conducted during each of the first two fall migration periods after restoration plans are approved, and then again when the lake is at or near the target elevation.	Conducted one of two fall migration periods in 2000; completion of second study is recommended

3.1 Hydrology

Lake Level

Mono Lake is hydrographically closed and as such, all surface and groundwater drains towards Mono Lake. Lake elevation, salinity, and water chemistry are influenced by inputs via surface water, springs, precipitation, and subsequent evaporative losses (Vorster 1985). The Mono Basin receives drainage and runoff from several nearby mountains and ranges including the Sierra Nevada, Cowtrack Mountain, the Excelsior Mountains, and others.

Climate has influenced the Mono Lake environment over geologic and historic time. Mono Lake is the saline and alkaline remnant of the much larger Lake Russell, present in the Pleistocene. At its highest, Lake Russell stood at 7,480 feet above sea level, and was once hydrologically connected to the Lahontan and Owens-Death Valley systems (Reheis, Stine and Sarna-Wojcicki 2002). Starting in the late Pleistocene, climatic variation resulted in the contraction of Lake Russell, and hydrologic isolation of Mono Lake. These climatic variations resulted in the level of Mono Lake fluctuating from an extreme high stand of 7,200 feet, to an extreme low of an approximately 6,368-foot lake elevation (Scholl et al. 1967 in Vorster 1985). Since 1941, lake level and salinity have been influenced by water exports by the City, and more recently, climate change may be becoming more influential.

In April of 1941, the City began exporting water from the Mono Basin by diverting Lee Vining Creek, Rush Creek, Walker Creek, and Parker Creek. The pre-diversion elevation of Mono Lake in April of 1941 was 6,416.9 feet. From 1941-1970, annual exports averaged 56,000 acre-feet, and the surface elevation of Mono Lake dropped over 29 feet during this same time period. In 1970, the completion of the second aqueduct in the Owens Valley expanded the capacity of the system, resulting in an increase in diversions, frequent full diversion of flows from Lee Vining, Walker, Parker and Rush Creek and a drying of the creek channels (SWRCB 1994). From 1970 to 1989, Mono Lake dropped another 12.6 feet as yearly exports averaged 82,000 acre-feet, with a peak export of 140,756 acre-feet in 1979. The lake level dropped to a record low of 6,371.0 feet in 1982, representing a cumulative 45-foot vertical drop in lake elevation as compared to the pre-diversion level. Decision 1631 amended the City's water rights license in order to support reaching a long-term average lake elevation of 6,392 feet.

Stream Flow

There are seven perennial creeks tributary to Mono Lake, all of which originate on the east slope of the Sierra Nevada. The perennial creeks are primarily snow-melt fed systems, with peak flows typically occurring in June or July, especially in normal-to-wet years. Peak flows may occur in April or May in dry years or on the smaller creeks (Beschta 1994). Rush Creek is the largest tributary, accounting for approximately 50% of stream-flow contributions to Mono Lake. Parker and Walker Creeks are small creeks tributary to Rush Creek. Rush Creek was permanently re-watered in 1982, however Parker Creek and Walker Creek, were not re-watered until 1990. Mono Lake's second largest tributary, Lee Vining Creek, was re-watered in 1986. Along the west shore is Log Cabin Creek, a small tributary monitored as part of the spring monitoring program. Flows in DeChambeau Creek along the northwest shore are intermittent, and do not consistently reach the lakeshore. Mill and Wilson Creeks are along the northwest shore of Mono Lake. Mill Creek is the third largest tributary to Mono Lake.

3.1.1 Hydrologic Monitoring Methodologies

Mono Lake Elevation

LADWP hydrographers record the elevation of Mono Lake monthly using a staff gauge installed at the boat dock on the west shore. The staff gauge is demarcated in tenths and hundredths of a foot. The Mono Lake Committee (MLC) also measures lake level, and since 1979, lake level data reported by the MLC has averaged 0.3 feet higher than LADWP data. Lake elevation is used to evaluate progress in meeting the target lake level, and for determining the annual allowable export. Lake elevation data is also used to evaluate the response of biological indicators including secondary producers, vegetation, and waterfowl.

Stream Flow

LADWP is required to monitor stream flow in the four Mono Lake tributaries from which the City diverts water for export - Rush Creek, Lee Vining Creek, Parker Creek and Walker Creek. Instream base and channel maintenance flows, initially dictated by Decision 1631 and Order 98-05, were superseded by Amended licenses 10191 and 10192. Flows are based on "Runoff Year" type. Runoff Year is the period from April 1-March 31. Runoff year type (Table 3.2) is based on a comparison of the total acre-feet of predicted runoff to the 1941-1990 average runoff of 122,124 acre-feet. Runoff predictions are based on the results of snow course surveys conducted along drainages contributing to Mono Basin runoff. The runoff year type assigned to any one year is based on the LADWP April 1 Mono Basin runoff forecast, although adjustments may be made on May 1. Runoff year type is used to determine the required annual restoration flows for Rush and Lee Vining Creeks. Instream and channel maintenance flows for other Mono Lake tributaries were not specified by the Order.

Table 3.2. Runoff Year Types as Per Amended Licenses 10191 and 10192

Water Year- Type	Runoff	Percent Exceedance
Dry	Less than or equal to 68.5% of average runoff	80 - 100 %
Dry/Normal	Between 68.5% and 82.5% of average runoff	60 - 80%
Dry/Normal I	Greater than 68.5% and less than or equal to 75%	60 - 70%
Dry/Normal II	Greater than 75% and less than or equal to 82.5%	70 - 80%
Normal	Greater than 82.5% and less than or equal to 107% of average runoff	40 - 60%
Wet/Normal	Greater than 107% and less than or equal to 136.5% of average runoff	20 - 40%
Wet	Greater than 136.5% of average runoff	0 - 20%
Extreme Wet	Greater than 160% of average runoff	0 - 8%
The year-type classifications are based on 1941-1990 average runoff of 122,124 AF		

LADWP hydrographers collect flow data using continuous instream data recorders that measure flow at 15-minute intervals. The measuring stations used to determine Rush Creek flows are Mono Gate One Return Ditch (STAID 5007) and Grant Lake Spill (STAID5078). Lee Vining Creek flows are measured at Lee Vining Creek below Conduit (STAID5009). The stations for Parker (Parker Creek below Conduit - STAID5003) and Walker Creek (Walker Creek below Conduit -STAID5002) are located just downstream of the diversion point into the Mono Crater Tunnel. Stream flow data are used to determine compliance with the Mono Basin Stream and Stream Channel Restoration Plan (LADWP 1996b) and amended licenses, and to provide environmental data to evaluate the response of biological indicators under the Mono Basin Waterfowl Habitat Restoration Plan (LADWP 1996a).

In order to provide a more complete record of annual stream flow contributions to Mono Lake, we also report on flows for DeChambeau Creek, and the estimated inputs of Mill Creek and Wilson Creek. LADWP maintains a continuous instream data recorder station on DeChambeau Creek west of Highway 395 (Dechambeau Creek above

Diversion -STAID5049). LADWP does not maintain flow measuring stations on Mill or Wilson Creeks, however flow data was obtained from USGS National Water Information System (waterdata.usgs.gov) for Mill Creek below Lundy Lake (10287069) and Lundy Power Plant Tailrace (10287195). Mill Creek below Lundy Lake measures flow in Mill Creek downstream of the diversion to the Lundy Powerhouse. The Lundy Power Plant Tailrace measures flows downstream of the Lundy Powerhouse. Water downstream of the Lundy Powerhouse is split between return flows to Mill Creek, a diversion to Conway Ranch, and a diversion to Wilson Creek. Further downstream on Wilson Creek, water is diverted off of Wilson Creek for use in the Restoration Ponds.

3.1.2 Hydrology Data Summary and Analysis

Lake Elevation

Monthly LADWP Mono Lake elevation data were summarized for 2022, and for the time period 1990-2022. This time series represents the period during which a preliminary injunction was in place that halted exports until the lake level recovered to 6,377 feet, and the implementation of Decision 1631, beginning in September 1994. Patterns of lake elevation change were evaluated on a yearly and long-term basis.

Although “Runoff Year” type is used for determining yearly prescribed stream flows, hydrologic data were summarized by “Water Year”, or the period from October 1-September 30 of each year. This is the preferred approach for biological analysis as the “Water Year” will encompass winter precipitation contributing to ecological conditions and processes the following year.

Stream Flow

The real-time station flow data were converted into daily flow, which was used to calculate monthly and annual inflow into Mono Lake. Inflow from Rush Creek is estimated by summing Mono Gate One Return Ditch (MGORD) (STAID 5007), Grant Lake Spill (STAID5078), Parker Creek below Conduit (STAID5003) and Walker Creek below Conduit (STAID5002). Lee Vining Creek below Conduit (STAID5009) and Dechambeau Creek above Diversion (STAID5049) are used to estimate inflow from Lee Vining and Dechambeau Creeks, respectively.

The contribution of Mill and Wilson Creek into Mono Lake cannot be precisely determined due to a lack of direct measure, and therefore the input amounts we report should be considered estimates. The estimated combined contribution of Mill Creek and Wilson Creek was calculated by summing USGS Stations Mill Creek below Lundy Lake (10287069) and Lundy Power Plant Tailrace (10287195). This calculation will overestimate flows to Mono Lake as diversions to Conway Ranch and the Restoration Ponds have not been subtracted.

3.1.3 Hydrology Results

Lake Elevation

In 2022, Mono Lake experienced a period of decreasing lake level throughout the calendar year (Figure 1). Lake level was fairly constant January through April, showing only a minimal 0.1-foot rise in level during this period. The lake was at its highest level in 2022 of 6,379.6 feet in February. Due to extremely low runoff in the Mono Basin in 2022, no post-runoff boost in lake level was seen, and the lake level steadily decreased through the remainder of the year, to a low of 6,378.1 feet in November and December and a net decline in lake elevation in calendar year 2022 of 1.5 feet.

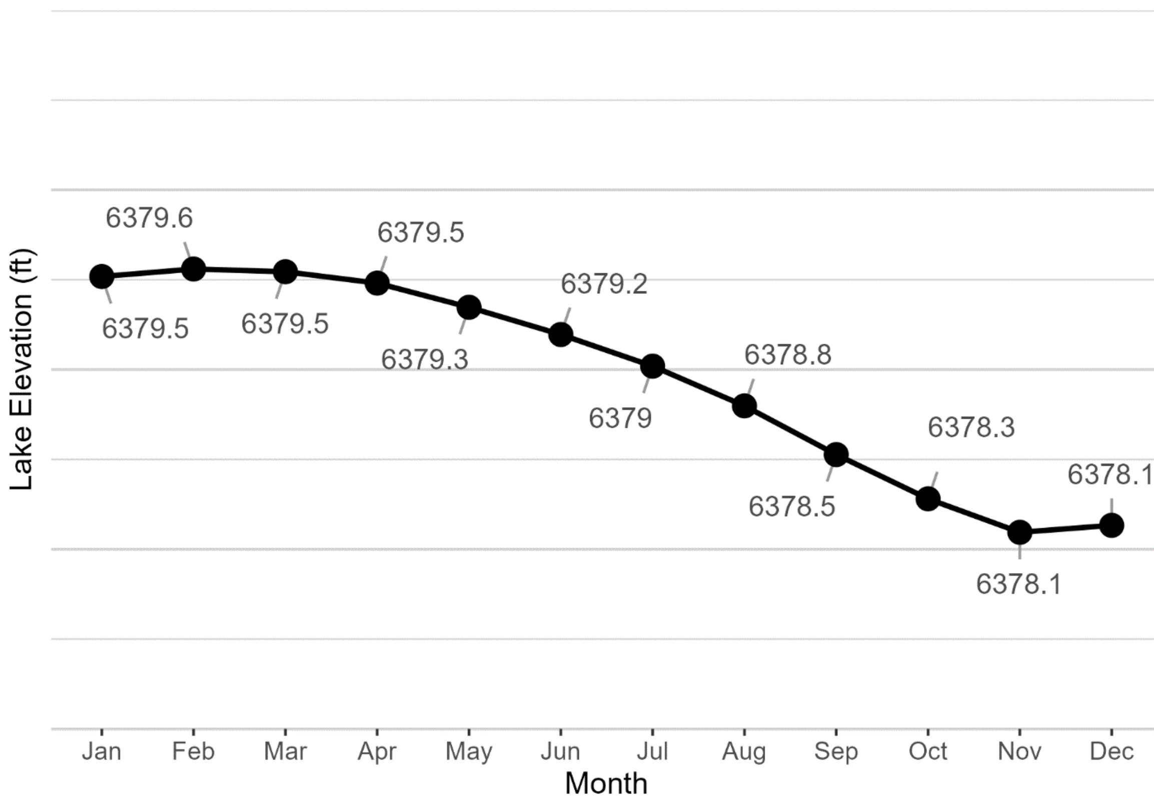


Figure 1. Mono Lake Monthly Elevation - 2022

Runoff during the 2021-2022 Water Year was 60,928 acre-feet, or 51% of the long-term average, and was a “Dry” water year-type. The 2021-22 Water Year was the sixth driest since 1935. Starting with the 2011-2012 Water Year (which marked the beginning of an extreme 5-year drought), six years have ranked within the bottom ten in terms of runoff (2011-12, 2013-14, 2014-15, 2019-20, 2020-21, and 2021-22).

Since Decision 1631, there have been three distinct wet periods, however the magnitude and duration of the wet periods has decreased progressively. The first wet period lasted from 1995 to 1998 and averaged 147% of normal based on the 50-year average between 1971 and 2020; the second wet period only lasted two years (2005 to 2006) and averaged 156% of normal; the third wet period also lasted two years (2010 to 2011) and averaged 131% of Normal. Following this third wet period was an extended drought that resulted in the driest 5-year period on record. This extended dry period year ended in 2017 with what was the second wettest on record of 207% of normal, or an “Extreme Wet year”.
From 1994 to 2019, Mono Lake has experienced four periods of increasing elevation, and four subsequent decreases, through a total elevation range of almost 8.0 feet (Figure 2). The highest elevation the lake achieved since 1994 was 6,384.7 feet, which occurred in July 1999. During a period of extended drought from 2012-2016, the lake elevation dropped almost 7 feet to a low of 6,376.8 feet in October 2016, the lowest level since implementation of the Order. Following the “Extreme Wet” runoff year of 2016-2017, followed by a “Normal” and then “Wet Normal” year, the lake level has shown some recovery from the extreme low point of 2016, but started to decline again with two consecutive very dry years (Figure 2).

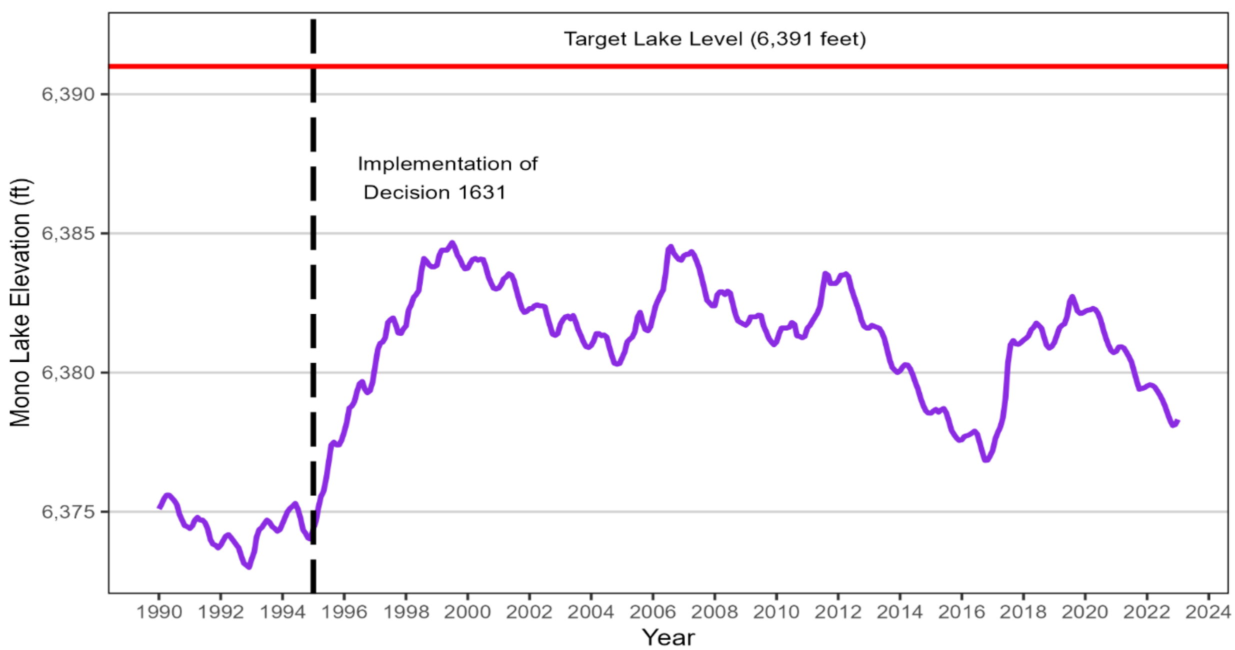


Figure 2. Mono Lake Elevation Between 1990 and 2022

Since Decision 1631, there have been four periods of lake level increase associated with above- average runoff.

Stream Flows

In 2022, the input from Rush Creek was 35,073 acre-feet or approximately 60% of the long-term average since 1990 (Table 3.3). Since 1990, Rush Creek has provided the largest inputs to Mono Lake averaging 59,102-acre-foot discharge, an average contribution of 50% since 1990, with a peak input over this time period in 2017 of 145,349 acre-feet. The input from Lee Vining Creek in 2022 was 20,130 acre-feet, or approximately 55% of the long-term average of 36,713 acre-foot. As was the case with Rush Creek, the highest input in this time period was 91,132 acre-feet in 2017. Input from the two major tributaries (Rush and Lee Vining Creeks) in 2022 was 55,203 acre-feet, or 58% of the long-term average since 1990. The input from Dechambeau Creek in 2022 was 368 acre-feet, 55% of the long-term mean. DeChambeau Creek has averaged 823 acre-feet since 1982 and has contributed less than 1% of total annual input since 1990. We were unable to get the 2022 flow data from Southern California Edison for Mill and Wilson Creek.

Table 3.3. Annual Flow Volume in Acre-Feet of Five Mono Lake Tributaries Based on Water Year

Year	Rush	Lee Vining	Dechambeau	Mill/Wilson
1990	71,046	18,643	325	9,115
1991	35,713	20,561	264	8,725
1992	44,632	20,798	178	10,590
1993	77,460	42,279	439	18,710
1994	56,776	29,376	450	11,118
1995	94,595	66,443	910	31,899
1996	91,841	56,284	1,243	25,557
1997	82,423	66,317	1,485	30,912
1998	93,177	62,335	1,325	27,113
1999	58,047	46,204	1,150	19,472
2000	50,497	40,432	749	16,370
2001	49,357	31,033	575	13,272
2002	45,900	36,599	405	12,708
2003	49,028	30,777	529	15,199
2004	47,644	31,871	549	15,115
2005	72,765	55,367	994	26,640
2006	108,899	75,860	1,459	32,149
2007	38,428	24,090	997	10,173
2008	45,159	25,631	587	13,265
2009	36,569	30,653	585	15,769
2010	57,622	34,775	671	19,343
2011	96,432	65,454	1,150	29,997
2012	46,890	19,486	926	11,272
2013	35,084	18,319	475	10,416
2014	31,893	20,047	340	8,539
2015	32,753	16,525	272	8,485
2016	44,242	28,748	275	15,232
2017	145,349	91,132	1,433	45,410
2018	63,397	33,624	1,211	21,720
2019	89,858	48,687	1,095	27,762
2020	41,437	20,149	747	11,344
2021	35,435	16,752	425	9,190
2022	35,073	20,130	368	-

3.1.4 Hydrology Discussion

Lake Elevation

Mono Lake has experienced a continuous decrease in lake level since 2019. At the final lake level read in December of 2022 (6378.1 feet), Mono Lake was 1.4 feet lower than in December 2021. As is typical of dry years (LADWP 2018), the maximum lake level occurred early in spring, and there was no runoff-associated increase in lake level observed in summer.

The implementation of Decision 1631 has resulted in a stabilization of Mono Lake elevation. Since export amounts are now regulated, and greatly reduced as compared to historic export amount prior to Decision 1631, variations in lake level are mainly driven by climate and runoff patterns. An updated lake level model will help determine the influence of various factors currently influencing Mono Lake elevation, including climate change.

Stream Flows

The 2022 runoff resulted in below-average total stream discharge into Mono Lake from the primary tributaries. The decreased stream discharge contributed to the decrease in lake level observed in 2022. Runoff in the Mono Basin has been typified by dry periods interrupted by short wet periods, except in the late 1930s to early 1940s, the late 1970s to 1980s, and the late 1990s when wet periods were found to last longer than the more recent wet periods (LADWP 2018). As mentioned previously, five of the ten lowest runoff years have occurred since 2011-12, including 2014-15 and 2020-21, the driest and the second driest on record, respectively. Recent dry years appear to be much drier. As a result, one or two wet years may not be enough to reverse the declining trend of the lake level unless a wet year is as extreme as 2016-17, or multiple successive wet years occur.

3.2 Limnology

Mono Lake supports a relatively simple yet productive aquatic ecosystem. Planktonic and benthic algae form the foundation of the food chain in the lake. The phytoplankton community is primarily composed of coccoid chlorophytes (*Picosystis* spp.), coccoid cyanobacteria, and several diatoms (primarily *Nitzschia* spp.) (Jellison and Melack 1993). Filamentous cyanobacteria (*Oscillatoria* spp.), filamentous green algae (*Ctenocladus circinnatus*), and the diatom *Nitzchia frustulum* dominant the benthic algal community (Herbst 1986).

Secondary producers in Mono Lake consist of invertebrate species. The most abundant secondary producer in the pelagic zone is the Mono Lake brine shrimp (*Artemia monica*). In the littoral zone, secondary producers including the alkali fly (*Ephydra hians*), long-legged fly (*Hydrophorus plumbeus*), biting midge (*Cuciloides occidentalis*), and deer fly (*Chrysops* spp.) graze on benthic algae (Herbst 1986).

Within the hydrographically closed basin, the water chemistry of Mono Lake is influenced by climate, water inputs, evaporative losses, and the chemical composition of the surrounding soils and rocks. The waters are saline and alkaline, and contain high levels of sulfates, chlorides, and carbonates. For the period 1938-1950, the salinity of Mono Lake was approximately 50 g/L, and by 1964 salinity had increased to 75 g/L, and up to 100 g/L by 1982 (Vorster 1985). Since implementation of Decision 1631, the salinity has varied from 72.4 to 97.8 g/L, which is approximately two to three times as salty as ocean water. The lake water is also highly alkaline, with a pH of approximately 10, due to the high levels of carbonates dissolved in the water.

The limnological monitoring program at Mono Lake is one component of the Plan and is required under SWRCB Order No. 98-05. The purpose of the limnological monitoring program as it relates to waterfowl is to assess limnological and biological factors that may influence waterfowl use of lake habitat (LADWP 1996a). The limnological monitoring program has four components: meteorology, physical/chemical analysis, chlorophyll *a*, and brine shrimp population monitoring.

An intensive limnological monitoring program at Mono Lake has been funded by LADWP since 1982. Until 2012, the Marine Science Institute (MSI), University of California, Santa Barbara, (UCSB) administered these funds, and UCSB researchers and technicians used facilities at the Sierra Nevada Aquatic Research Laboratory (SNARL) to conduct the monitoring. After receiving training in limnological sampling and laboratory analysis methods from the scientists and staff at MSI and SNARL, LADWP Watershed Resources staff assumed responsibility for the program, and have been conducting the limnological monitoring program at Mono Lake since July 2012.

This report summarizes the results of monthly limnological field sampling conducted in 2022, and discusses the results in the context of the entire period of record. In addition, past findings are summarized to evaluate long term trends in water chemistry and *Artemia* population dynamics.

3.2.1 Limnological Monitoring Methodologies

Methodologies for both the field sampling and the laboratory analysis followed those specified in *Field and Laboratory Protocols for Mono Lake Limnological Monitoring* (*Field and Laboratory Protocols*) (Jellison 2011). The methods described in *Field and Laboratory Protocols* are specific to the chemical and physical properties of Mono Lake and therefore may vary from standard limnological methods (e.g. Strickland and Parsons 1972). The methods and equipment used by LADWP to conduct limnological monitoring were consistent and follow those identified in *Field and Laboratory Protocols* except where noted.

Meteorology

One meteorological station on Paoha Island provided the majority of the weather data. The station is located approximately 30 m from shore on the southern tip of the island. The base of the station is at 1,948 m (6,391 feet) above sea level, several meters above the current surface elevation of the lake. During the visit to the island in 2021, LADWP staff found the anemometer missing and the radiation shield for the air temperature and relative humidity sensors dislodged, resulting in no wind data and erratic readings in relative humidity and air temperature. Daily precipitation and air temperature recorded at LADWP Cain Ranch weather station, which was established in May 1931, are presented in this report.

In addition to Cain Ranch data, monthly average maximum and minimum temperatures from Mono Lake (045779) between 1950 and 1988 and Lee Vining (044881) since 1988 were obtained from the Western Regional Climate Center (www.wrcc.dri.edu) and combined to gain insight into climatic trends. Winter temperature was calculated by averaging the monthly average maximum (or minimum) temperature from December of the previous year to February of the subsequent year. For example, monthly average from December 2018 was combined with monthly average from January and February 2019 to obtain the winter average for 2019. Summer temperature was calculated based on monthly temperature between June and August. All long-term averages were based on the 50-year record between 1951 and 2000.

Field Sampling and Laboratory Procedures

Sampling of the physical, chemical, and biological properties of the water including the *Artemia* community was conducted at 12 buoyed stations at Mono Lake (Figure 3) on the dates listed in Table 3.4. The water depth at each station at a lake elevation of 6384.5 feet (1,946 m) is indicated on Figure 3. Stations 1-6 are considered western sector stations, and stations 7-12 are eastern sector stations.

Due to inclement weather in April and snow in December, no sampling was conducted in April and December 2022. No DO reading was taken in September due to DO meter malfunction. Monitoring was generally conducted on two separate days: 1) the first day for dissolved oxygen, ammonium, and chlorophyll *a* sampling, and 2) the second day for *Artemia* sampling, CTD casting, and Secchi readings. Surveys were generally conducted around the 15th of each month.

Table 3.4. Mono Lake Limnology Sampling Dates for 2022

Month	DO	CTD/Secchi/*Artemia*
Feb	2/10/2022	2/17/2022
Mar	3/24/2022	3/24/2022
Apr	-	-
May	5/25/2022	5/17/2022
Jun	6/22/2022	6/22/2022
Jul	7/20/2022	7/20/2022
Aug	8/24/2022	8/25/2022
Sep	-	9/22/2022
Oct	10/19/2022	10/20/2022
Nov	11/17/2022	11/17/2022
Dec	-	-

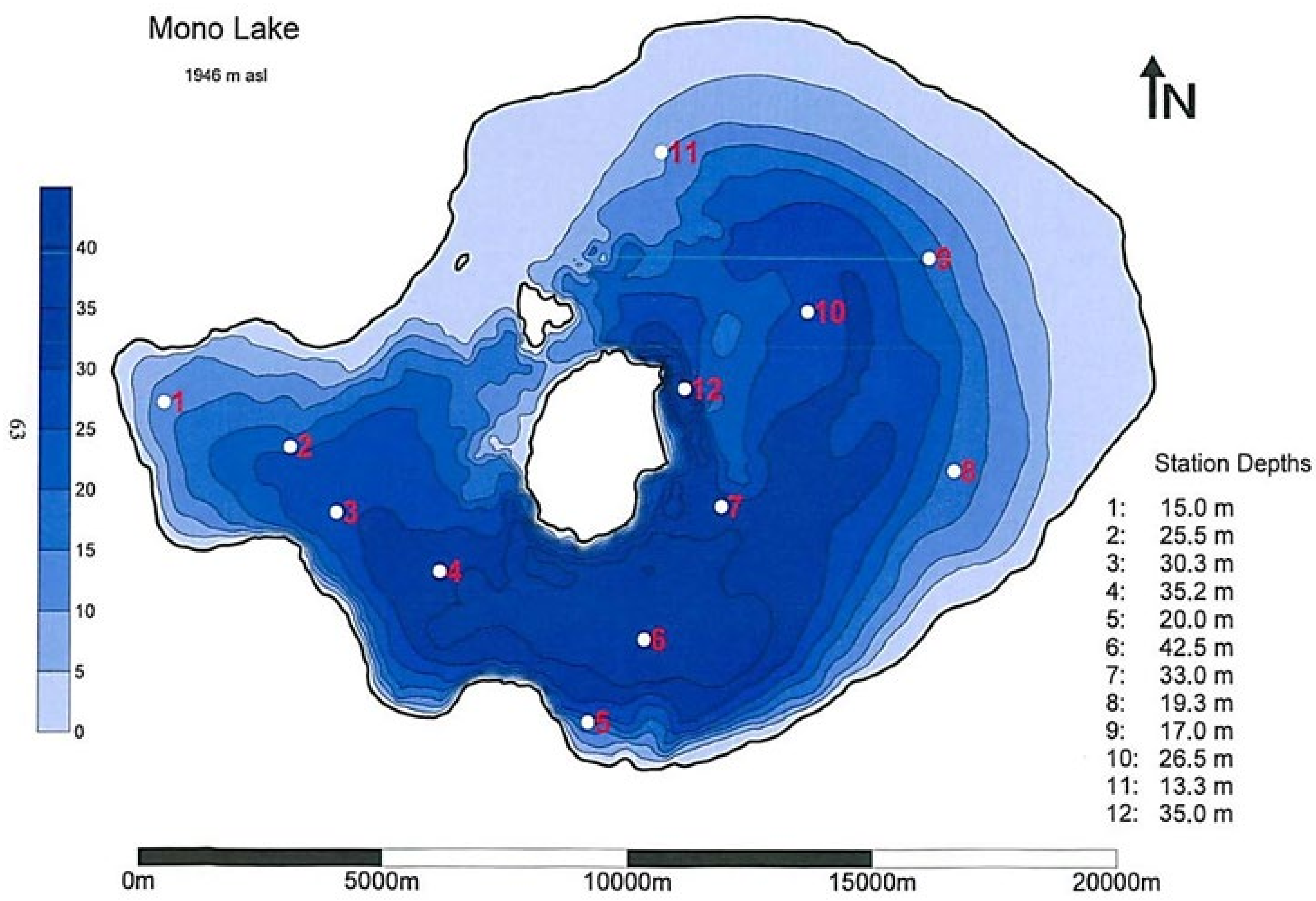


Figure 3. Sampling Stations at Mono Lake and Associated Station Depths

Physical and Chemical

Transparency

Lake transparency was measured each month at all 12 stations using a Secchi disk.

Temperature, Conductivity, and Salinity

A high-precision conductivity-temperature-depth profiler (CTD) (Sea-Bird 19Plus V2) was used to record conductivity at 9 stations (2, 3, 4, 5, 6, 7, 8, 10 and 12) on a monthly basis. The Sea-Bird CTD is programmed to collect data at 250 millisecond intervals. During sampling, the CTD was initially lowered just below the surface of the water for 40 seconds during the pump delay time. The CTD was then lowered at a rate of approximately 0.5 meter/second with data collected at approximately 12.5-centimeter depth intervals. In situ, conductivity measurements at Station 6 are corrected for temperature (25˚C). Conductivity and temperature readings at the depth closest to a whole number are assigned to that depth and reported at one-meter intervals beginning at one meter in depth down to the lake bottom. Salinity expressed in g/L was calculated based on the equation in past compliance reports (LADWP 2004).

Three adjustments were performed on all data since 2012: 1) depth adjustment based on air pressure, 2) depth adjustment based on water density of the lake, and 3) replacing conductivity with standardized conductivity to calculate water density. All values presented in this report have been corrected, and, consequently, historical values of adjusted conductivity and salinity in this report are different from those reported 2012-2021.

Dissolved Oxygen

Dissolved oxygen is measured at one centrally-located station (Station 6) with a Yellow Springs Instruments Rapid Pulse Dissolved Oxygen Sensor (YSI model 6562). Readings were taken at one-meter intervals and at 0.5-meter intervals in the vicinity of the oxycline and other regions of rapid change. Data are reported for one-meter intervals only without calibration to Mono Lake salinity.

Ammonium and Chlorophyll α Sampling

A change in the consultant and laboratory unfortunately resulted in the ammonium and chlorophyll *a* analyses not being conducted as requested by LADWP. The procedure used was not sensitive enough to detect ammonium levels of Mono Lake water, and chlorophyll *a* values did not follow the range of values reported in past; thus, the results are therefore not presented in this report.

Artemia Population Sampling

Artemia Population

The *Artemia* population was sampled by one vertical net tow at each of the 12 stations (Figure 3). Samples were taken with a plankton net (0.91 m x 0.30 m diameter, 118 µm Nitex mesh) towed vertically through the water column. Samples were preserved with 5% formalin in Mono Lake water.

An 8x to 32x stereo microscope was used for all *Artemia* counts. Depending on the density of shrimp, counts were made of the entire sample or of a subsample made with a Folsom plankton splitter. When shrimp densities in the net tows were high, samples were split so that approximately 100-200 individuals were subsampled. Shrimp were classified as nauplii (instars 1-7), juveniles (instars 8-11), or adults (instars >12), according to Heath (1924). Adults were sexed and the reproductive status of adult females determined. Non-reproductive (non-ovigerous) females were classified as empty. Ovigerous females were classified as undifferentiated (eggs in early stage of development), oviparous (carrying cysts) or ovoviviparous (naupliar eggs present). The net efficiency of 70% was applied to all samples.

An instar analysis was completed for seven of the twelve stations (Stations 1, 2, 5, 6, 7, 8, and 11). Nauplii at these seven stations were further classified as to specific instar stage (1-7). Biomass was determined from the dried weight of the shrimp tows at each station. After counting, samples were rinsed with tap water and dried in aluminum tins at 50°C for at least 48 hours. Samples were weighed on an analytical balance immediately upon removal from the oven. Calculation of long-term *Artemia* population statistics followed the method proposed by Jellison and Rose (2011). Daily values of adult *Artemia* between sampling dates were linearly interpolated using the R package *zoo*. The mean, median, peak and centroid day (calculated center of abundance of adults) was then calculated for the time period May 1 through November 30, during which adult *Artemia* population is most abundant. Long-term statistics were determined by calculating the mean, minimum, and maximum values for the entire time period of monitoring.

Artemia Fecundity

When mature females were present, an additional net tow was taken from four western sector stations (1, 2, 5 and 6) and three eastern sector stations (7, 8 and 11) to collect adult females for fecundity analysis including body length and brood size. Live females collected for fecundity analysis were kept cool and in low densities during transport to the LADWP laboratory in Bishop, CA.

Immediately upon return to the laboratory, ten females from each sampled station were randomly selected, isolated into individual vials, and preserved with 5% formalin. Female length was measured using 8x magnification from the tip of the head to the end of the caudal furca (setae not included). Egg type was noted as undifferentiated, cyst, or naupliar. Undifferentiated egg mass samples were discarded. Brood size was determined by counting the number of eggs in the ovisac and any eggs dropped in the vial. Egg shape was noted as round or indented.

Historical Comparisons of Chemical, Physical, and Biological Data

In order to visualize temporal trends of measured parameters, lake-wide averages were first calculated for Secchi and *Artemia* population for each monitoring data while water temperature and salinity readings from Station 6 were selected for each monitoring date. Monitoring date values were then converted to monthly averages as there were more than one monitoring sessions per month in past monitoring. Monthly averages were presented in tile plots with month and year as x-axis and y-axis, respectively. Filling of cells were based on the long-term average based on available years for the respective month.

3.2.2 Limnology Data Analysis: Salinity and *Artemia* Population

Salinity is a key parameter influencing the structure of aquatic algal and invertebrate communities of closed lake systems (Herbst and Blinn 1998, Verschuren et al. 2000). High salinity has been shown to negatively affect the survival, growth, reproduction, and cyst hatching of *Artemia* in Mono Lake (Starrett and Perry 1985, Dana and Lenz 1986). Negative effects are accentuated when salinity approaches 159 g/L to 179 g/L (Dana and Lenz 1986). Salinity of Mono Lake depends on the volume of the lake with minor diversions occurring during meromixis, and lake volume, in turn, shows a close relationship with lake elevation (Pelagos 1987). Vorster (1983) presented a salinity-elevation model for a wide range of lake elevation. Neither a volume-based salinity model or Vorster's model considers temporary departures of salinity from the model during meromixis. *Artemia* populations responds to the lake mixing regime as seen after breakup of each meromixis. It is important to understand salinity during the first reported meromixis between 1983 and 1987 because the post meromixis peak in 1989 produced the largest *Artemia* population seasonal mean and peak among five such peaks, and no salinity data are available prior to 1991. It is also important to understand relationships between salinity and *Artemia* seasonal population dynamics. The latter data are available beginning in 1979. First, the hypsometry was constructed for lake elevations between 6,226 ft and 6,430 ft with the first value corresponding to 0% while the second value corresponding to 100%. Salinity values prior to 1991 were, then, estimated based on the hypsometric-based, elevation-based, and Vorster's models. Salinity values in mixolimnion and monimolimnion were estimated separately for the first recorded meromixis (1983-1987). Separate regression equations were used to estimate salinity in mixolimnion during the formation phase (July 1982 to October 1983) and persistence phases (November 1983 to October 1988) based on the formation and persistence equations developed based on the second recorded meromixis (1995-2002). Mono Lake freshwater input from two major tributaries (Rush and Lee Vining Creeks) was used to estimate salinity in monimolimnion, which tended to show a monotonic decrease throughout meromixis. Finally, estimated salinity values were used to make inference into the relationship between *Artemia* population and salinity.

3.2.3 Limnology Results

Meteorology

Daily Air Temperature

Daily maximum air temperature ranged from -6.1°C to 36.8°C (Figure 4). The daily maximum of 36.8°C was recorded on September 6 during the severe heat wave engulfing much of California while the minimum daily maximum of -6.1°C was recorded on February 23. The daily minimum temperature ranged from -19.6°C to 16.3°C. The daily minimum of -19.6°C was recorded on February 24 while the maximum daily minimum of 16.3°C was recorded on July 31. Daily maximum temperature exceeded 30°C for 22 consecutive days from August 19 to September 9. Daily minimum temperature remained above average between 1991 and present throughout the month of August and all 30 days but one day for September.

Precipitation

The total precipitation between January 1 and December 31 measured at LADWP Cain Ranch was 8.9 inches. Precipitation events were most frequent (8) in December, followed by August with seven, in 2022, and the largest single day total precipitation of 2.01 inch was recorded on December 27 (Figure 5). Monthly precipitation in 2022 did not quite follow the long-term seasonal pattern between January and May with very little precipitation recorded during the period. No precipitation was recorded in January and May. A series of thunderstorms induced by an active monsoon dropped 2.1 inches of precipitation during a five-day period in the beginning of August. The initial part of an atmospheric river event dropped 2.01 and 1.34 inches on December 27 and 31, respectively, helping the December total reach 4.78 inches. The atmospheric river event dropped a total of 9.65 inches of precipitation and pushed the 2022-23 water year total to 11.8 inches, already exceeding the long-term average (113% of Normal).

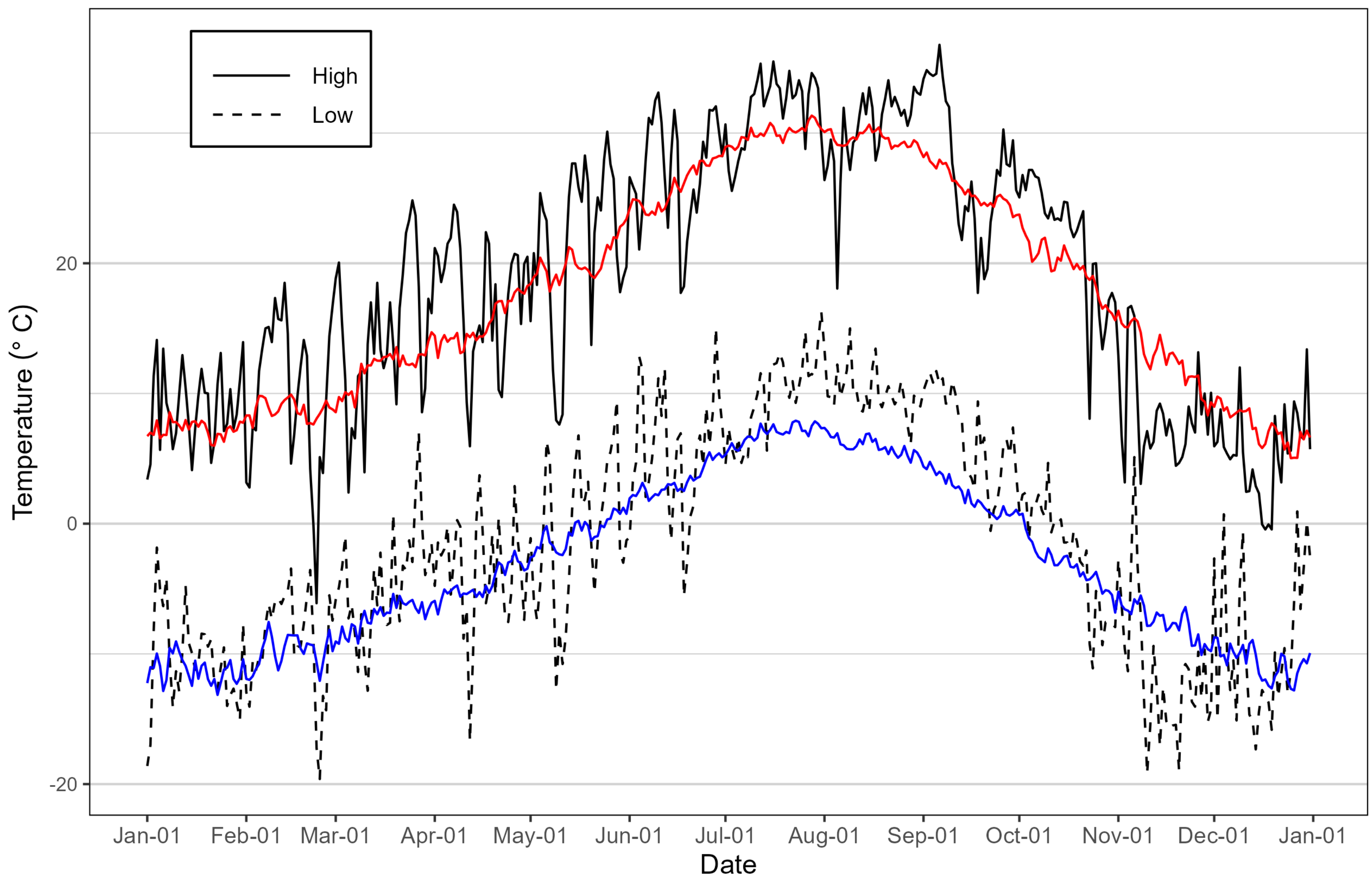
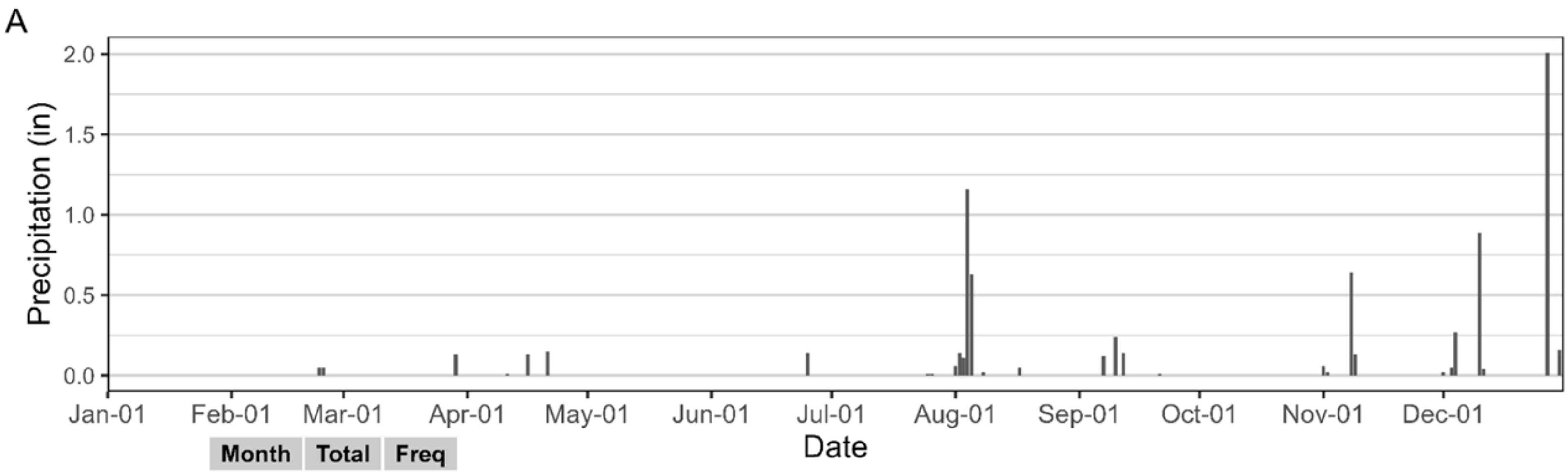


Figure 4. Daily Minimum and Maximum Air Temperature (°C)

A red line indicates the long-term average of daily maximum air temperature while a blue line indicates the long-term average of daily minimum air temperature.



B

Month	Total	Freq
Jan	0.00	0
Feb	0.10	2
Mar	0.13	1
Apr	0.29	3
May	0.00	0
Jun	0.14	1
Jul	0.02	2
Aug	2.17	7
Sep	0.51	4
Oct	0.00	0
Nov	0.85	4
Dec	4.78	8
Total	8.99	32

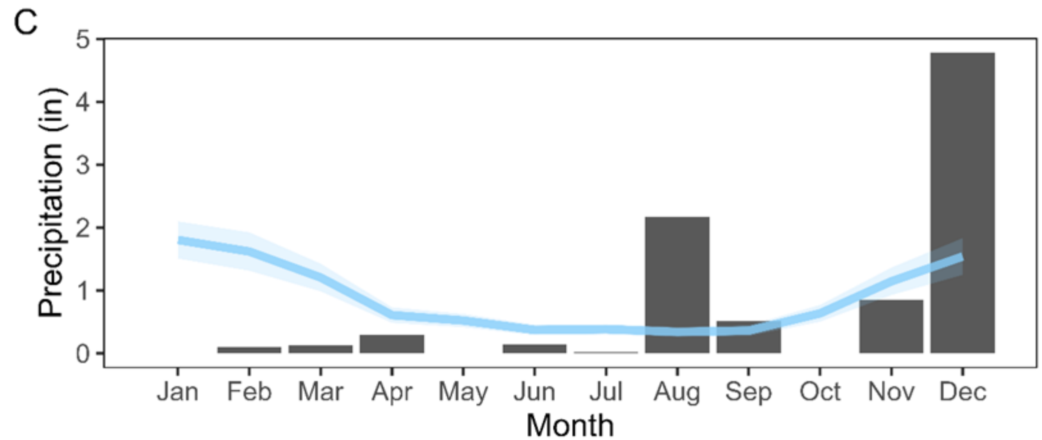


Figure 5. A) Total Daily Precipitation (in), B) Total Monthly Precipitation and Frequency of Precipitation Events, and C) Monthly Precipitation

Precipitation was recorded at LADWP Cain Ranch Weather Station. A blue line in C) indicates the 50-year average (1951-2000) of the respective months, and blue shaded area indicate 95% confidence interval.

Long Term Trends in Temperature and Precipitation

The year 2022 started with a slightly warmer January followed by an average February, and warmer March and April (
Figure 6). Between June and October, the monthly average temperature remained above the long-term average (LTA). It was particularly warm in July through September when the monthly average temperature was 2.6°C, 1.9°C, and 2.4°C higher than LTA of the respective month. November was colder (4.5°C below LTA) and it remained cold into the end of the year with December being -1.4°C below LTA.

The winter of 2021-22 diverged slightly from the recent trend of above LTA for both maximum and minimum monthly average temperature as the maximum temperature was above LTA while the minimum temperature was slightly below LTA (Figure 7). The maximum monthly average was 0.3°C above LTA and ranked 36th while the minimum monthly average was -0.06°C below LTA and ranked 37th. The summer of 2022 was warmer than LTA for both maximum and minimum monthly averages, 0.7°C and 2.1°C above the respective LTA and ranked 19rd and 3rd (Figure 8). Four of the past six years, including 2022, had minimum temperature belonging to the top ten warmest years. Winter precipitation in 2021-22 (2.08 in) remained below LTA for the third winter in row and was ranked 77th in 90 years and 42% of LTA (Figure 9). Summer precipitation exceeded 200% of LTA for the second year in row with 2.33 inches recorded because of the daily thunderstorms induced by an active monsoon in the beginning of August (Figure 5). Summer precipitation was ranked 6th and 207% of LTA. The winter of 2021-22 was slightly warmer but dry while the summer of 2022 was warm and wet.

There is no clear long-term trend for average summer and winter temperatures since 1951 except for increasing average summer minimum temperatures (Figure 10, Table 3.5). This trend is stronger since 1983 ($r=0.81$, $p<0.0001$), indicating there has been a warming trend in summer minimum temperature from the beginning of the limnology monitoring in 1979. Correlation coefficients between air temperature and years increased from 0.03 to 0.23 since 1983, and then up to 0.43 since 2005 for winter maximum; 0.05 to 0.49 since 1983 for winter minimum, and 0.26 to 0.54 since 1983, and to 0.58 since 2005 for summer maximum compared to the entire data set (>1951). A slope of the trend line indicates this increase in air temperature rise. Minimum air temperature has been rising approximately at 0.88°C and 0.53°C per a decade since 1973 and 1983 for summer and winter, respectively, while maximum temperature has been rising at a much faster rate approximately at 1.45°C and 1.78°C since 2009 and 2005 for summer and winter, respectively.

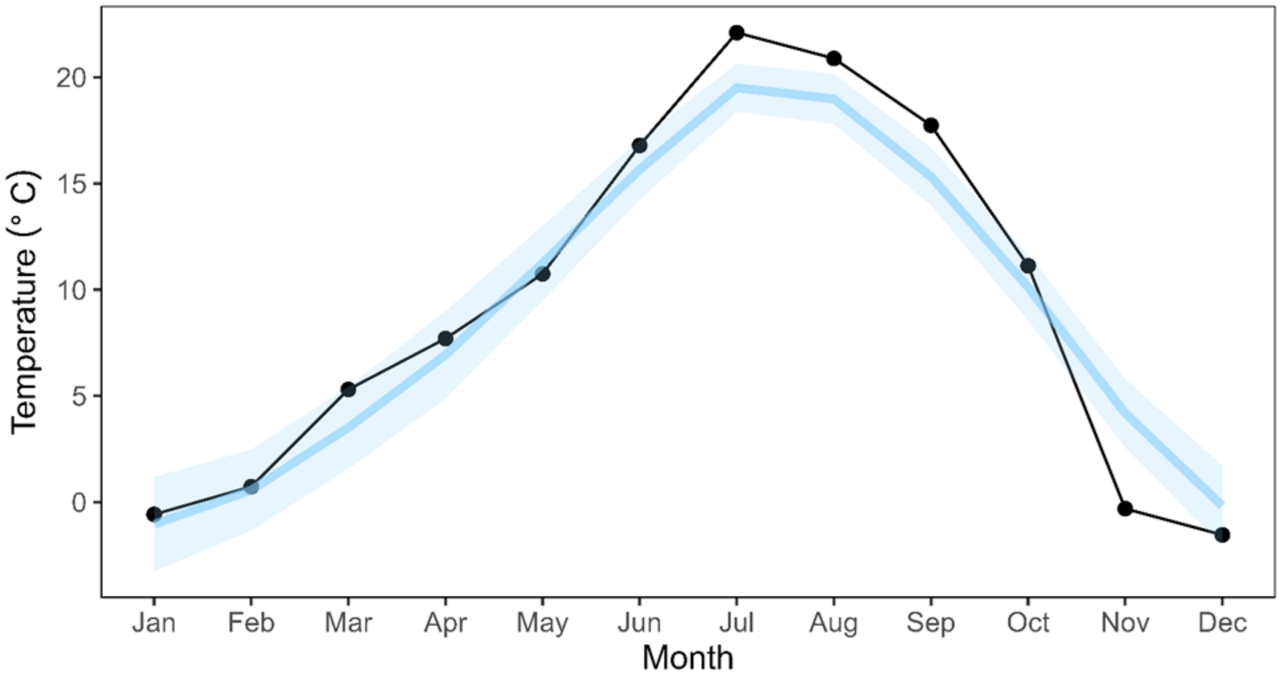


Figure 6. Monthly Temperature in 2022 Compared to the Long-term Averages

Long term average monthly temperature was calculated using records at Mono Lake (Station Number 045779-3) between 1951 and 1988, and Lee Vining (Station Number 044881) since 1989; data obtained from Western Regional Climate Center. A blue line indicates the long-term average monthly temperature and the shaded area indicates the standard errors of the respective months.

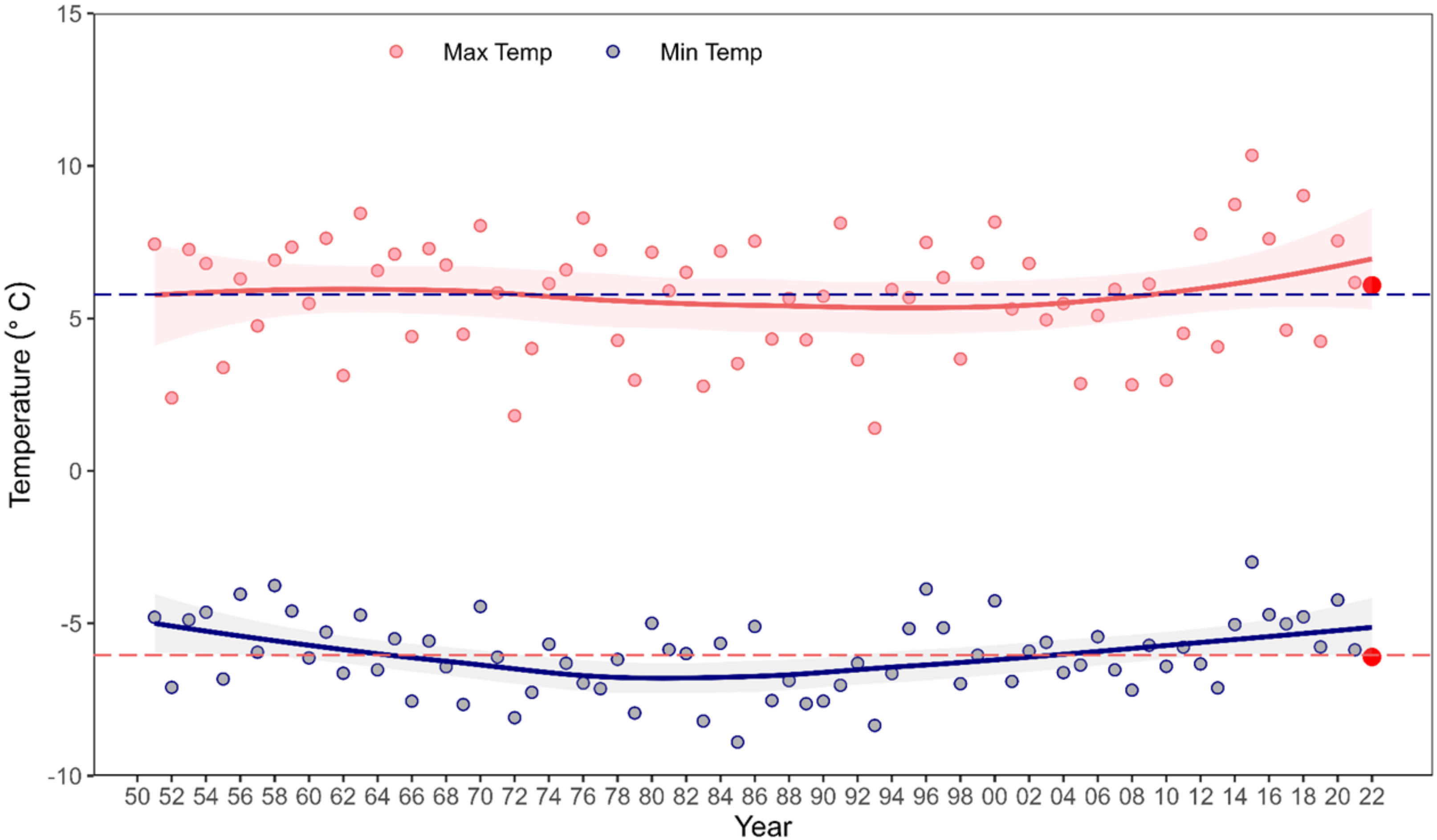


Figure 7. Average Temperature during Winter Months (December through February)

Temperature was recorded at Mono Lake (Station Number 045779) between 1951 and 1988 and at Lee Vining (Station Number 044881) since 1989; data obtained from Western Regional Climate Center. Solid lines represent trend lines based on locally estimated scatterplot smoothing (LOESS) with span of 0.5. Dashed lines indicate the 50-year average between 1951 and 2000 of monthly maximum and minimum temperatures.

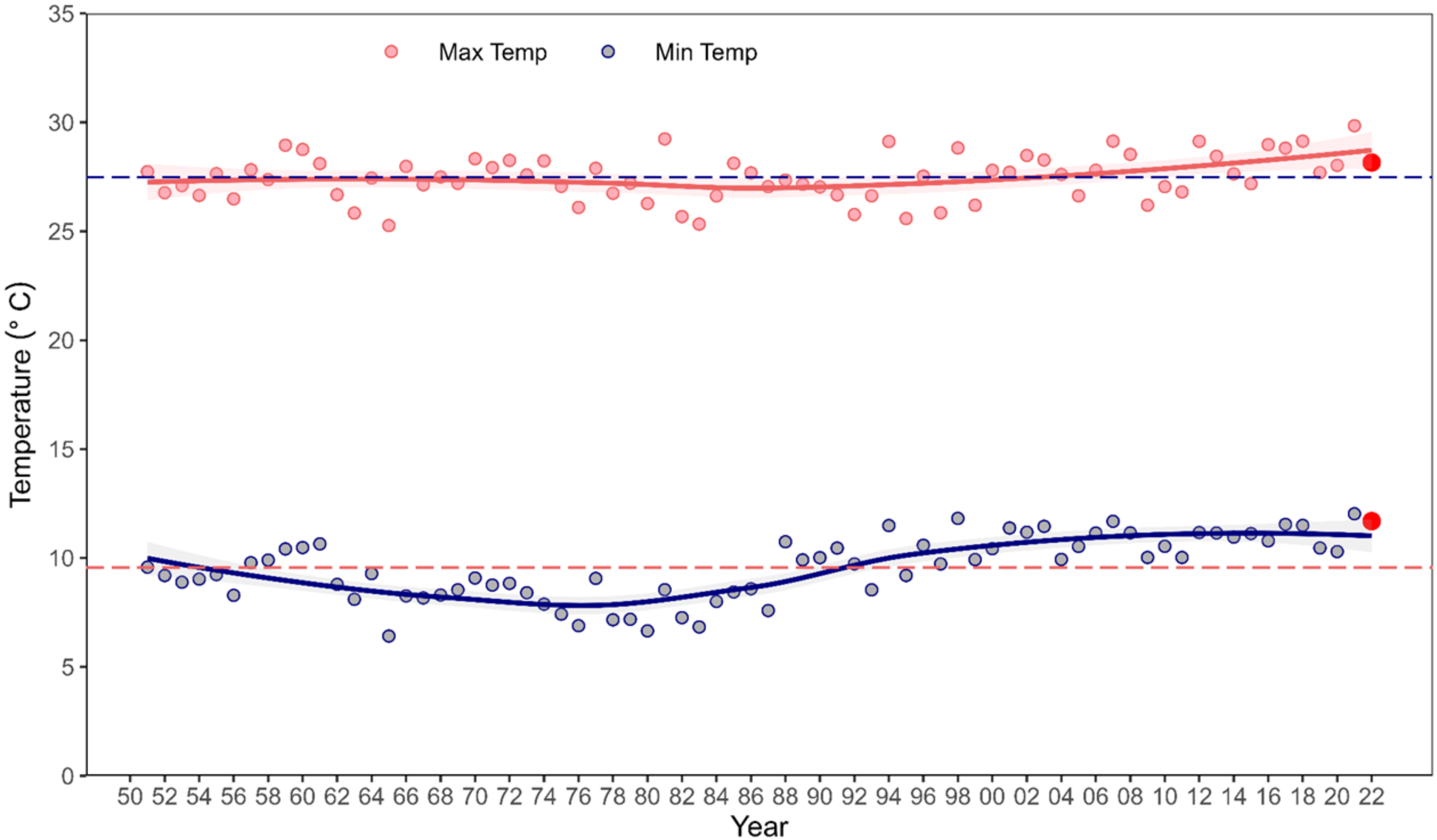


Figure 8. Average Temperature during Summer Months (June through August)

Temperature was recorded at Mono Lake (Station Number 045779-3) between 1951 and 1988 and at Lee Vining (Station Number 044881) since 1989; data obtained from Western Regional Climate Center. Solid lines represent trend lines based on locally estimated scatterplot smoothing (LOESS) with span of 0.5. Dashed lines indicate the 50-year average between 1951 and 2000 of monthly maximum and minimum temperatures.

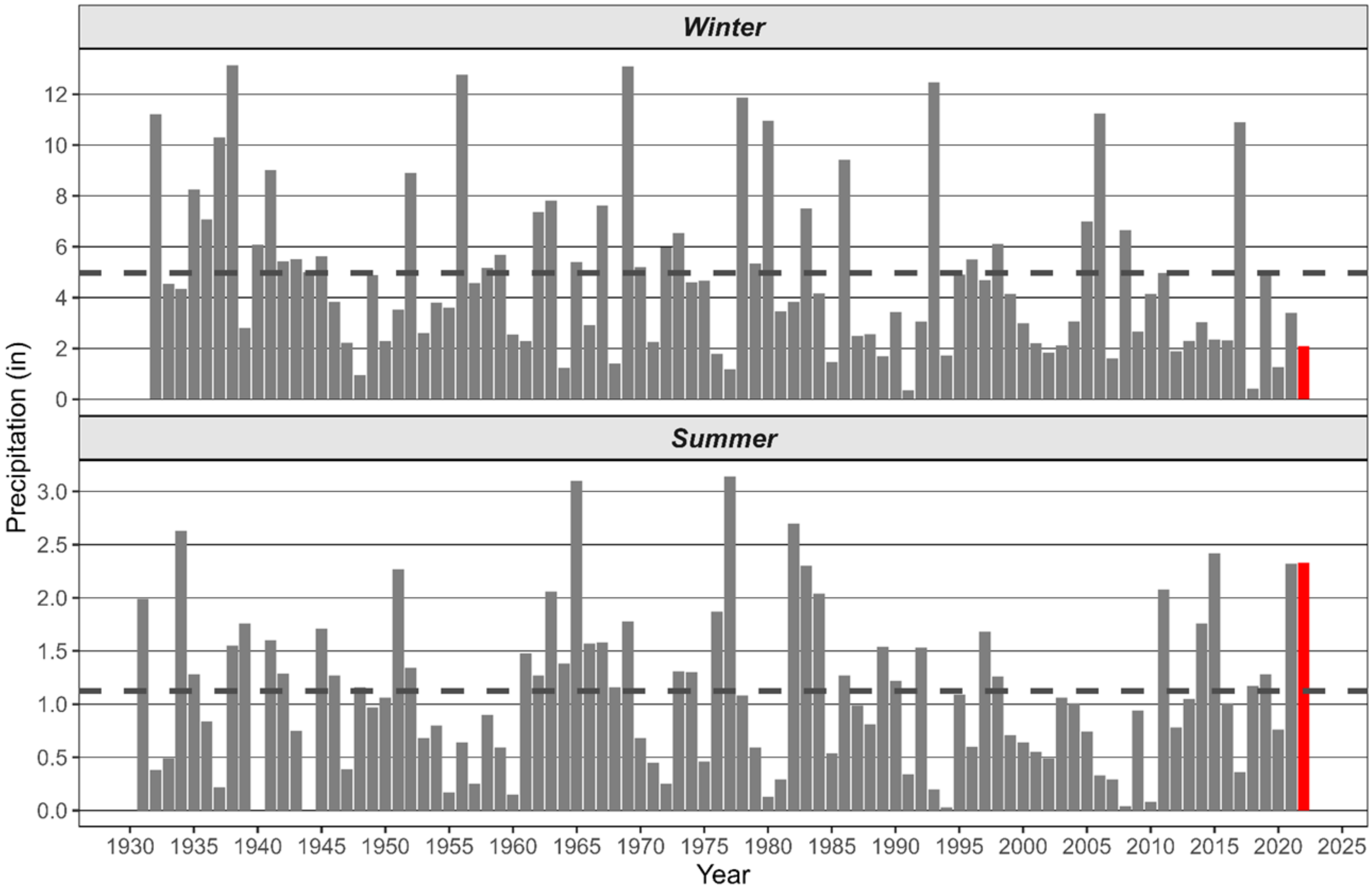


Figure 9. Total Winter (December to February) and Summer (June to August) Precipitation

Precipitation recorded at LADWP Cain Ranch since 1932. The dashed lines indicate the 50-year averages between 1951 and 2000 for the respective season.

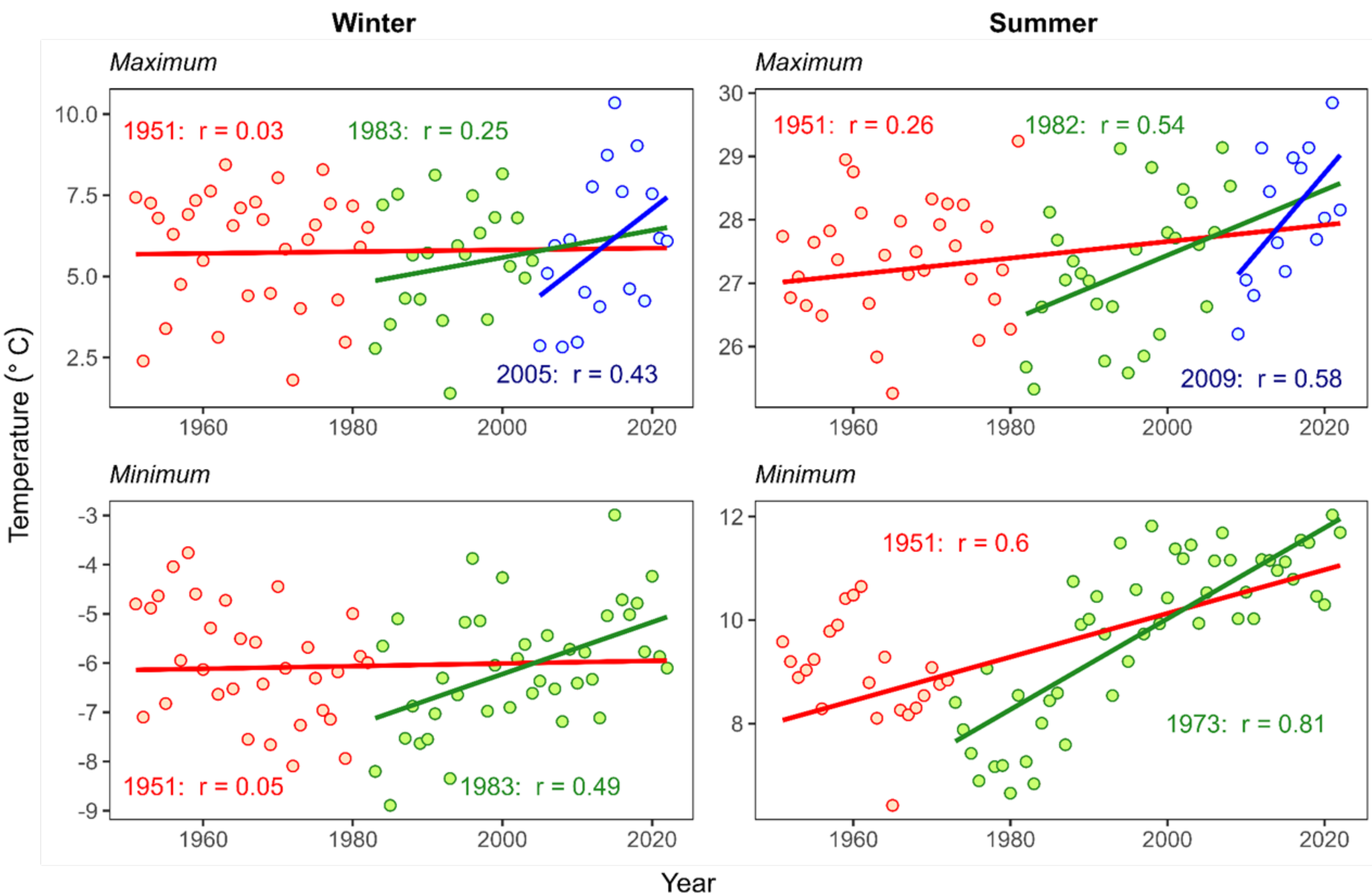


Figure 10. Correlation Coefficients between Seasonal Average Temperature and Different Sets of Years

Years on the x axis indicate a starting year of the set of years. For instance, in the case of 2004, a correlation coefficient was calculated based on years between 2004 and 2022.

Table 3.5. Rates of Air Temperature Changes (°C) during Selected Time Periods

Season	Variable	Time Period	r	Change per Decade
Summer	Maximum	1951 to present	0.26	0.13
		1982 to present	0.54	0.52
		2009 to present	0.58	1.45
	Minimum	1951 to present	0.60	0.42
		1973 to present	0.81	0.88
Winter	Maximum	1951 to present	0.03	0.03
		1983 to present	0.25	0.42
		2005 to present	0.43	1.79
	Minimum	1951 to present	0.05	0.03
		1983 to present	0.49	0.53

Physical and Chemical

Mono Lake Surface Elevation

The average monthly surface elevation of Mono Lake in January 2022 was 6379.5 feet or 1.3 feet lower than the January lake elevation in 2021 (Figure 11). Water Year 2021-22 produced 60,928 acre-feet of runoff in Mono Basin, or 51% of the long-term average, and ranked 83rd since 1935. Input from the two major tributaries (Rush and Lee Vining creeks) was 55,203 acre-feet, or 58% of the long-term average since re-watering in 1990. The lake level dropped 1.4 feet to 6378.1 feet by December 2022.

Transparency

Average lake-wide transparency remained below 1 m throughout 2022, and the maximum single station reading was 0.9 m at Station 2 in July (Table 3.6). Transparency of Mono Lake during the summer improved from 0.21 m in May to only 0.52 m in July as *Artemia* grazing reduced midsummer phytoplankton. This is the fifth time since the five-year drought (2012-16) that lake-wide transparency remained below 1 m year-round (2015, 2016, and 2020 to 2022, Figure 12). Annual peak lake-wide monthly mean transparency decreased from 5.1 m in 2013 to 1.5 m in 2014, and this low transparency has persisted except the brief increase observed between 2017 and 2019 following the near-record runoff in 2017.

Lake transparency tends to fluctuate with the lake mixing patterns with higher transparency during meromixis due to reduced nutrient availability in the euphotic zone, which, in turn, reduces primary productivity, and lower transparency during monomixis due to relatively higher nutrient availability in the euphotic zone (Figure 13). The overall trend in transparency (represented by the loess trend line) began to appear around 2008, and the trend varied among monitoring months (Figure 14). From February through May and December, the trend is more convex with higher transparency evident during earlier years of the meromictic event between 1995 and 2002. The pattern during the summer months resembles the S-shaped curve with higher transparency during earlier years of monitoring followed by the transition and lower transparency in recent years. Fall months show a negative trend but more variability especially during earlier years of monitoring.

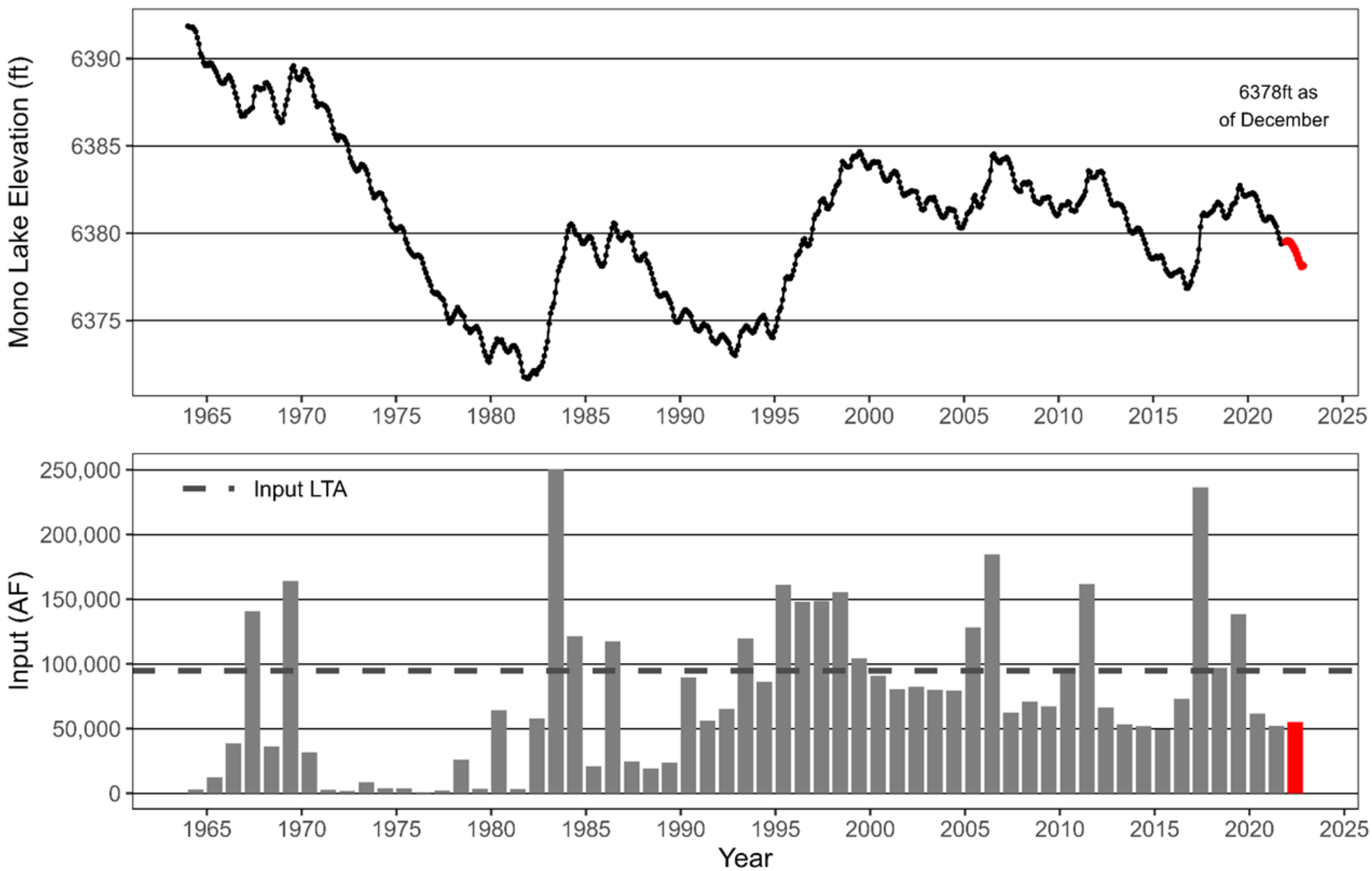


Figure 11. Mono Lake Surface Elevation (top) and Combined Inflow of Rush and Lee Vining Creeks (bottom)

Mono Lake elevation and input data are monthly average and total, respectively. Input is monthly flow volume of all tributaries to Rush Creek since 1963. The long-term average (LTA) is based values between 1982 and 2022.

Table 3.6. Secchi Depths (m) between February and December in 2022

	Feb	Mar	May	Jun	Jul	Aug	Sep	Oct	Nov
Western Sectors									
1	0.4	0.5	0.2	-	0.8	0.4	0.2	0.3	0.5
2	0.4	0.6	0.2	-	0.9	0.3	0.5	0.3	0.5
3	0.3	0.4	0.2	0.5	0.5	0.4	0.2	0.25	0.5
4	0.4	0.3	0.2	0.4	0.5	0.4	0.3	0.3	0.6
5	0.3	0.3	0.3	0.6	0.5	0.3	0.4	0.3	0.6
6	0.4	0.4	0.2	0.5	0.4	0.4	0.4	0.3	0.4
AVE	0.37	0.42	0.22	0.5	0.6	0.37	0.33	0.29	0.52
SD	0.05	0.12	0.04	0.08	0.2	0.05	0.12	0.02	0.08
Eastern Sectors									
7	0.4	0.3	0.2	0.5	0.4	0.4	0.3	0.3	0.6
8	0.4	0.4	0.2	0.5	0.4	0.3	0.4	0.3	0.5
9	0.3	0.3	0.2	0.6	0.45	0.4	0.5	0.3	0.6
10	0.4	0.3	0.2	0.5	0.5	0.3	0.4	0.3	0.5
11	0.3	0.3	0.2	-	0.45	0.5	0.4	0.3	0.5
12	0.4	0.3	0.2	-	0.4	0.3	0.4	0.3	0.5
AVE	0.37	0.32	0.2	0.52	0.43	0.37	0.4	0.3	0.53
SD	0.05	0.04	0	0.05	0.04	0.08	0.06	0	0.05
Total Lakewide									
AVE	0.37	0.37	0.21	0.51	0.52	0.37	0.37	0.3	0.52
SD	0.05	0.1	0.03	0.06	0.16	0.07	0.1	0.01	0.06

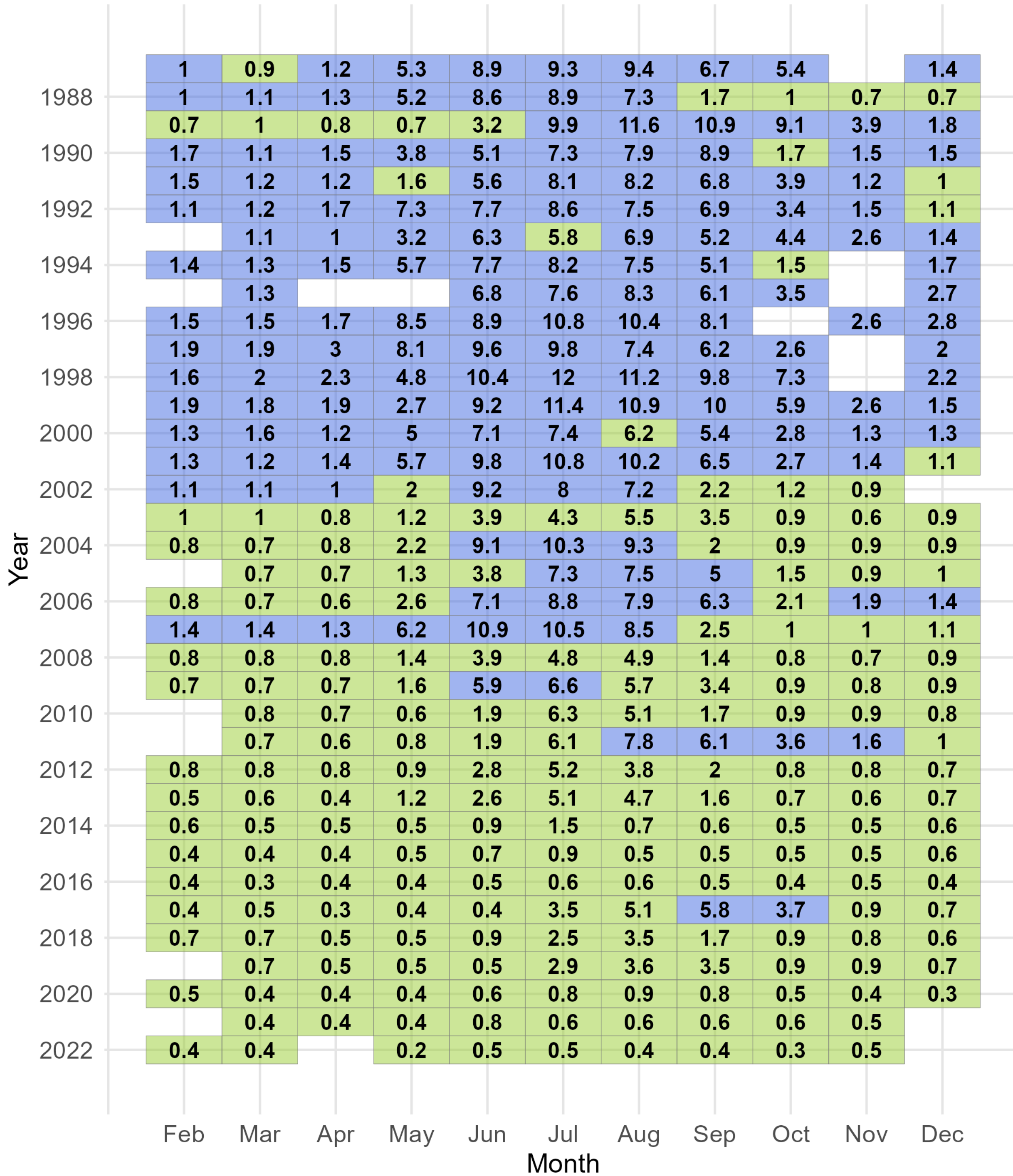


Figure 12. Long-term Lake-wide Average Secchi Depths (m)

Blue-colored cells indicate above the long-term average of the respective month while green-colored cells indicate below the long-term average of the respective month.

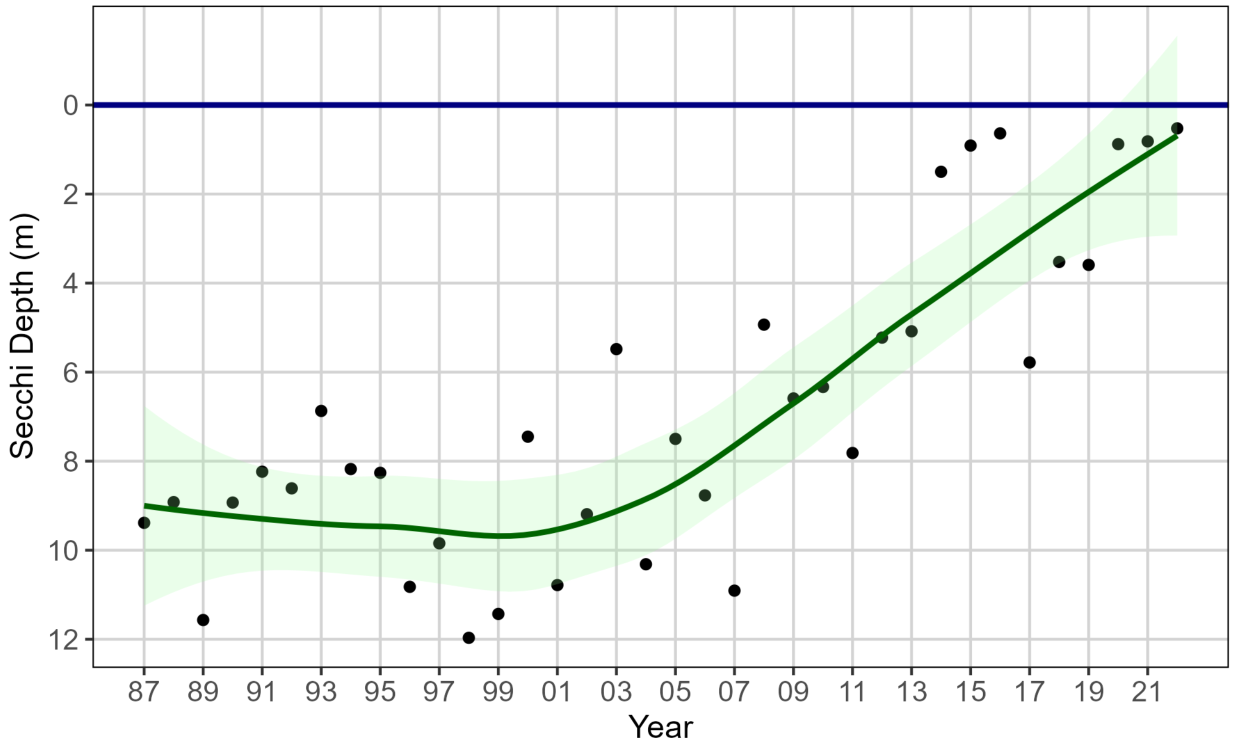


Figure 13. Trend in Annual Peak Lake-wide Secchi Depth Readings (m)

A green line is based on the LOESS fit with span of 0.75 and shaded area indicates standard error associated with the LOESS model.

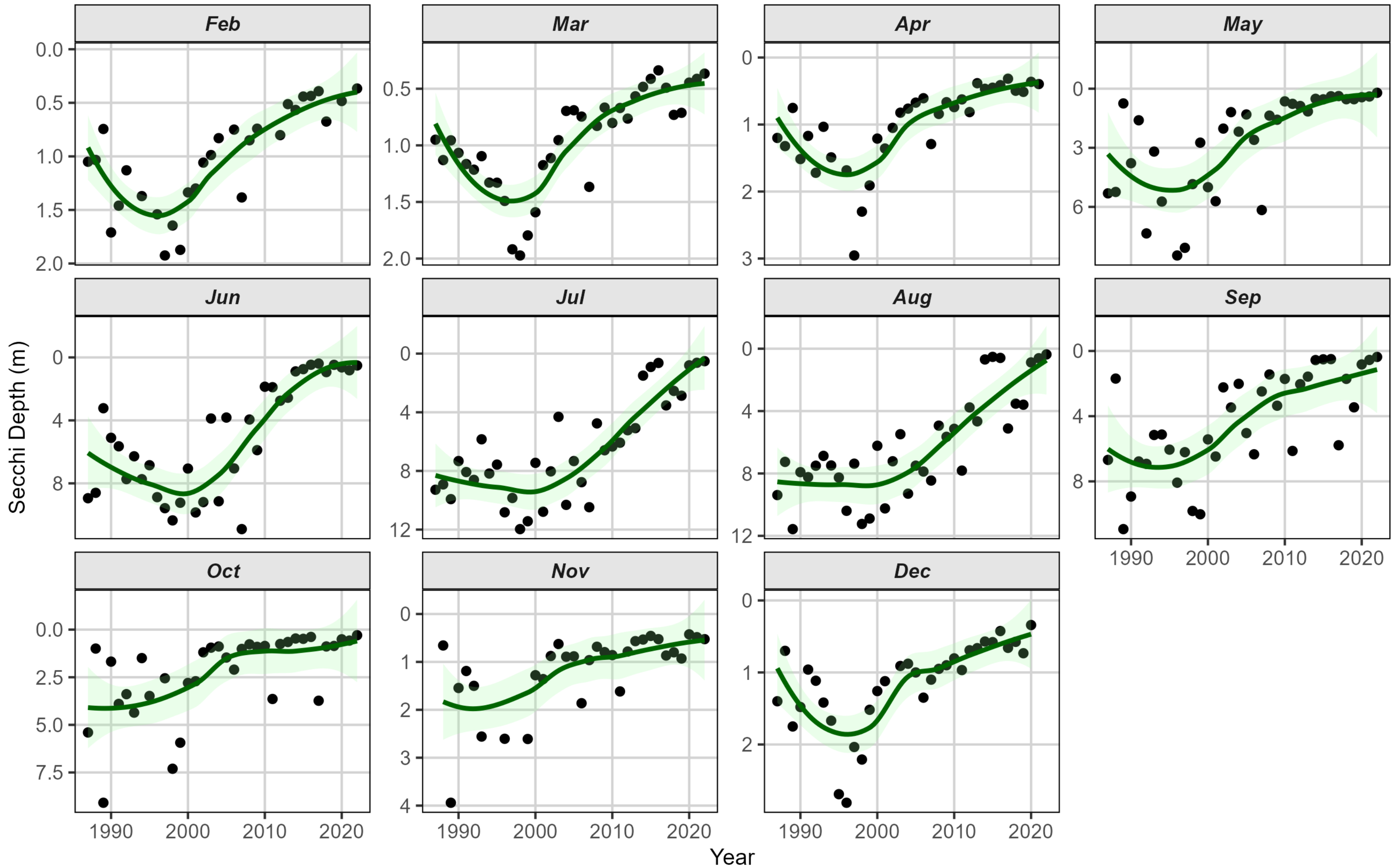


Figure 14. Monthly Trends in Annual Peak Lake-wide Secchi Depth Readings (m)

A green line is based on the LOESS fit with span of 0.75 and shaded area indicates standard error associated with the LOESS model.

Water Temperature

Mono Lake had only a 0.3°C vertical thermal gradient in February (Table 3.7, Figure 15). The thermocline started to develop at between 2-3 m in March and continued to migrate downward to 5-6 m in May, and 9-10 m in August. By October, the thermocline migrated to a depth of between 15 and 16 m. Mono Lake became nearly isothermal again in November. Average water temperature in the epilimnion ($\leq$ 10 m) fluctuated above and below the long-term average of the respective month throughout the year while average water temperature in the hypolimnion (>20 m) remained above the long-term average of the respective month after March (Figure 16, Figure 17). Water temperature in the hypolimnion in July, September, and August was highest recorded since 1991 and second highest for June and August.

Conductivity

After the latest meromictic event ended in 2020, the lake was monomictic in 2021 and 2022. In February, specific conductivity ranged between 83.5 mS to 83.9 mS (Table 3.8, Figure 18). In March, slightly higher conductivity (more than 1 mS higher than the rest of readings) was found at or around thermocline at depths of between 2-3 m, this layer of higher conductivity migrated down to 5-6 m and remained there between May and August. Dilution due to runoff was observed in July, but only between 1 and 2 m, resulting weak seasonal stratification. Declining lake level due to increased evaporation progressively increased conductivity near the surface through summer and September. The gradient decreased to 0.7 mS in November as the lake became holomictic.

Table 3.7. Water Temperature (°C) Depth Profile at Station 6 in 2022

Depth	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	3.4	9.0	-	13.1	15.9	22.0	21.8	17.2	14.6	8.5	-
2	3.2	8.0	-	12.9	16.4	22.0	21.6	16.8	14.6	8.7	-
3	3.0	6.1	-	12.7	15.7	22.1	21.5	16.7	14.5	8.7	-
4	2.9	5.3	-	12.7	15.3	22.0	21.3	16.6	14.4	8.7	-
5	2.9	5.0	-	11.3	15.3	21.3	21.4	16.7	14.5	8.7	-
6	2.9	4.9	-	10.1	15.0	20.4	21.5	16.7	14.6	8.7	-
7	2.9	4.7	-	9.6	14.5	20.2	21.5	16.7	14.6	8.7	-
8	2.9	4.6	-	9.4	13.8	18.7	21.0	16.7	14.5	8.7	-
9	2.9	4.4	-	9.1	13.2	17.4	19.7	16.7	14.5	8.7	-
10	3.0	4.3	-	8.8	12.8	15.9	16.2	16.7	14.6	8.7	-
11	3.0	4.1	-	8.7	11.4	14.6	13.7	16.6	14.6	8.7	-
12	3.0	4.1	-	8.5	10.4	13.3	11.9	16.6	14.6	8.7	-
13	3.0	4.0	-	8.1	9.7	11.2	11.0	16.2	14.6	8.7	-
14	2.9	3.8	-	7.5	9.0	10.1	10.1	15.3	14.5	8.8	-
15	2.9	3.7	-	7.0	8.7	9.4	9.2	14.0	14.0	8.8	-
16	2.9	3.6	-	6.8	8.5	8.7	8.9	12.4	10.7	8.7	-
17	2.9	3.6	-	6.6	8.0	8.4	8.7	11.1	10.1	8.7	-
18	2.8	3.6	-	6.4	7.7	8.2	8.5	10.4	10.0	8.7	-
19	2.8	3.5	-	5.9	7.3	7.9	8.3	10.0	10.0	8.7	-
20	2.8	3.4	-	5.8	7.1	7.6	8.0	9.6	10.0	8.7	-
21	2.8	3.3	-	5.6	6.9	7.5	7.8	9.4	10.0	8.7	-
22	2.8	3.3	-	5.6	6.7	7.3	7.7	9.2	10.0	8.7	-
23	2.8	3.2	-	5.5	6.6	7.2	7.6	8.8	9.9	8.7	-
24	2.8	3.1	-	5.5	6.5	7.2	7.5	8.5	9.8	8.6	-
25	2.8	3.1	-	5.5	6.4	7.1	7.4	8.3	9.7	8.6	-
26	2.8	3.0	-	5.3	6.3	7.0	7.3	8.2	9.7	8.5	-
27	2.8	3.0	-	5.2	6.3	6.9	7.2	8.1	9.6	8.5	-
28	2.8	3.0	-	5.2	6.2	6.8	7.0	8.0	9.5	8.5	-
29	2.8	2.9	-	5.0	6.1	6.7	7.0	7.8	9.5	8.4	-
30	2.8	2.9	-	5.0	6.0	6.6	6.9	7.7	9.6	8.4	-
31	2.8	2.9	-	4.9	6.0	6.5	6.9	7.7	9.6	8.3	-
32	2.8	2.9	-	4.8	5.9	6.5	6.8	7.6	9.5	8.3	-
33	2.8	2.9	-	4.7	5.9	6.5	6.8	7.5	9.4	8.3	-
34	2.8	2.9	-	4.6	5.8	6.4	6.8	7.4	9.3	8.3	-
35	2.8	2.9	-	4.6	5.7	6.4	6.7	7.4	9.0	8.3	-
36	2.8	2.9	-	4.6	5.7	6.3	6.7	7.4	9.1	8.3	-
37	2.8	2.9	-	4.6	5.7	6.3	6.6	7.3	8.9	8.3	-
38	2.8	2.8	-	4.6	5.7	6.2	6.6	7.3	8.5	8.3	-

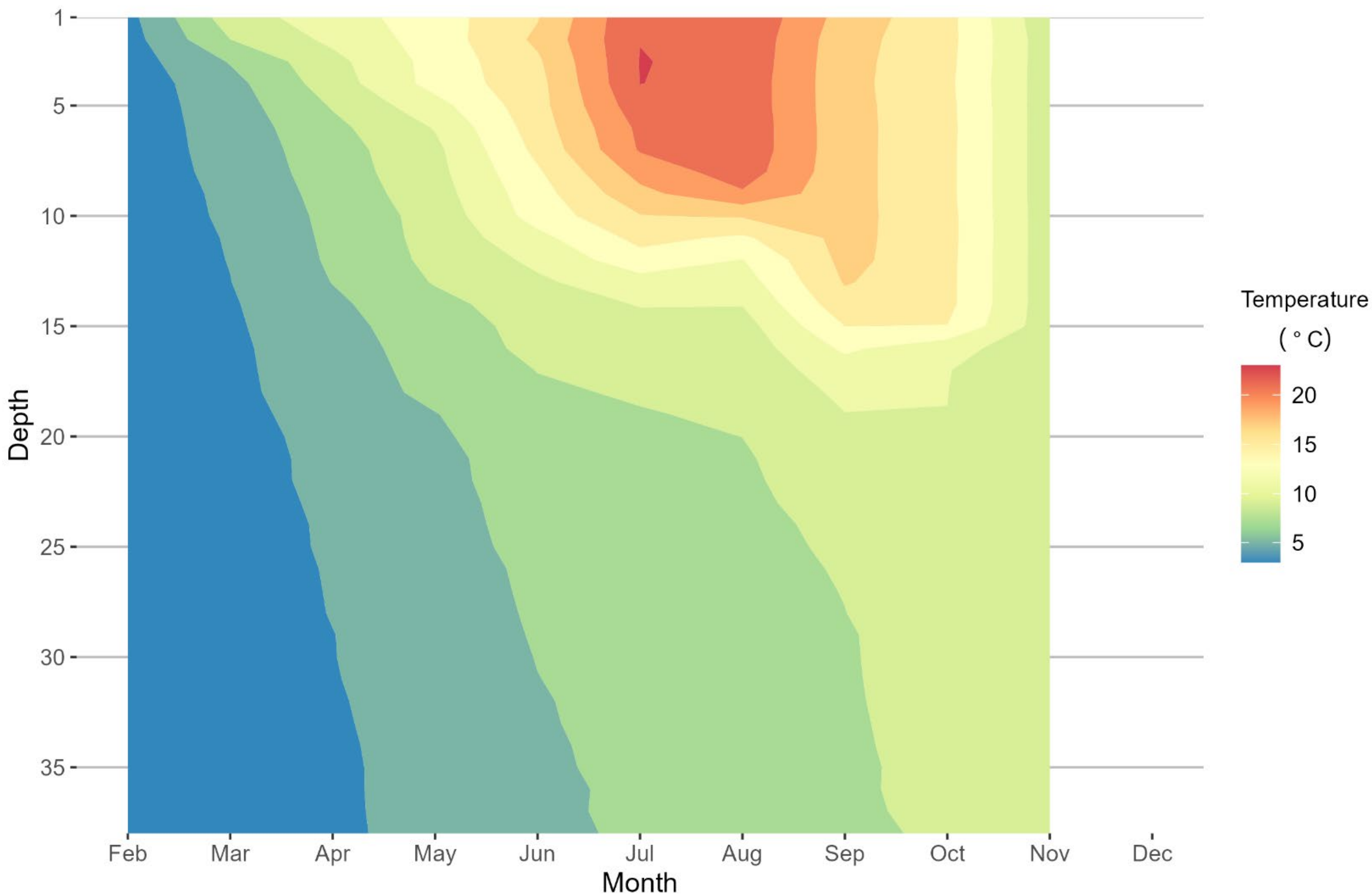


Figure 15. Water Temperature (°C) Depth Profile at Station 6 in 2022

April values were interpolated using March and May values. Missing values near the bottom were substituted with closest non-missing value above.

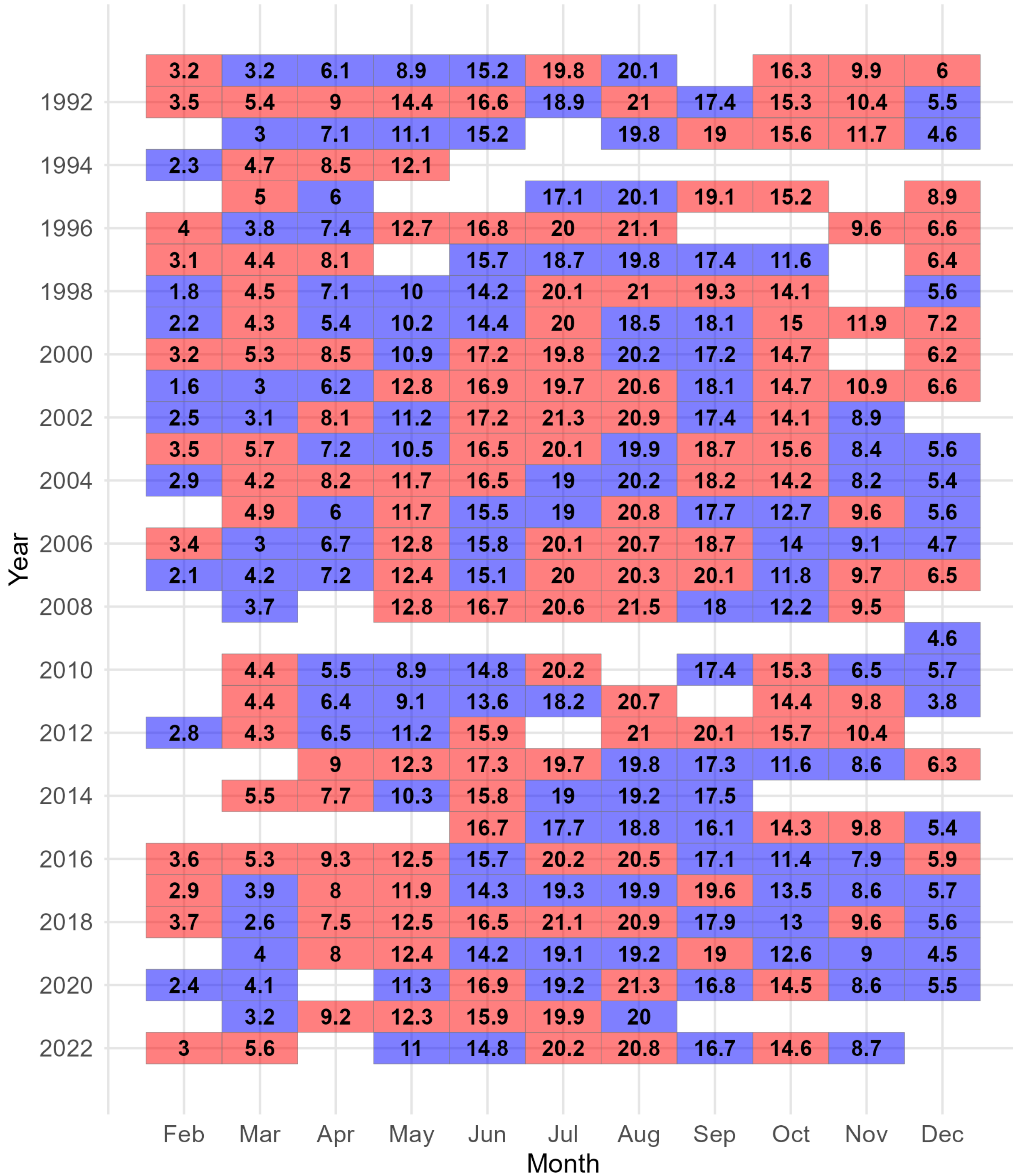


Figure 16. Average Water Temperature (°C) between 1 and 10 m at Station 6

Red-colored cells indicate above the long-term average of the respective month while blue-colored cells indicate below the long-term average of the respective month.

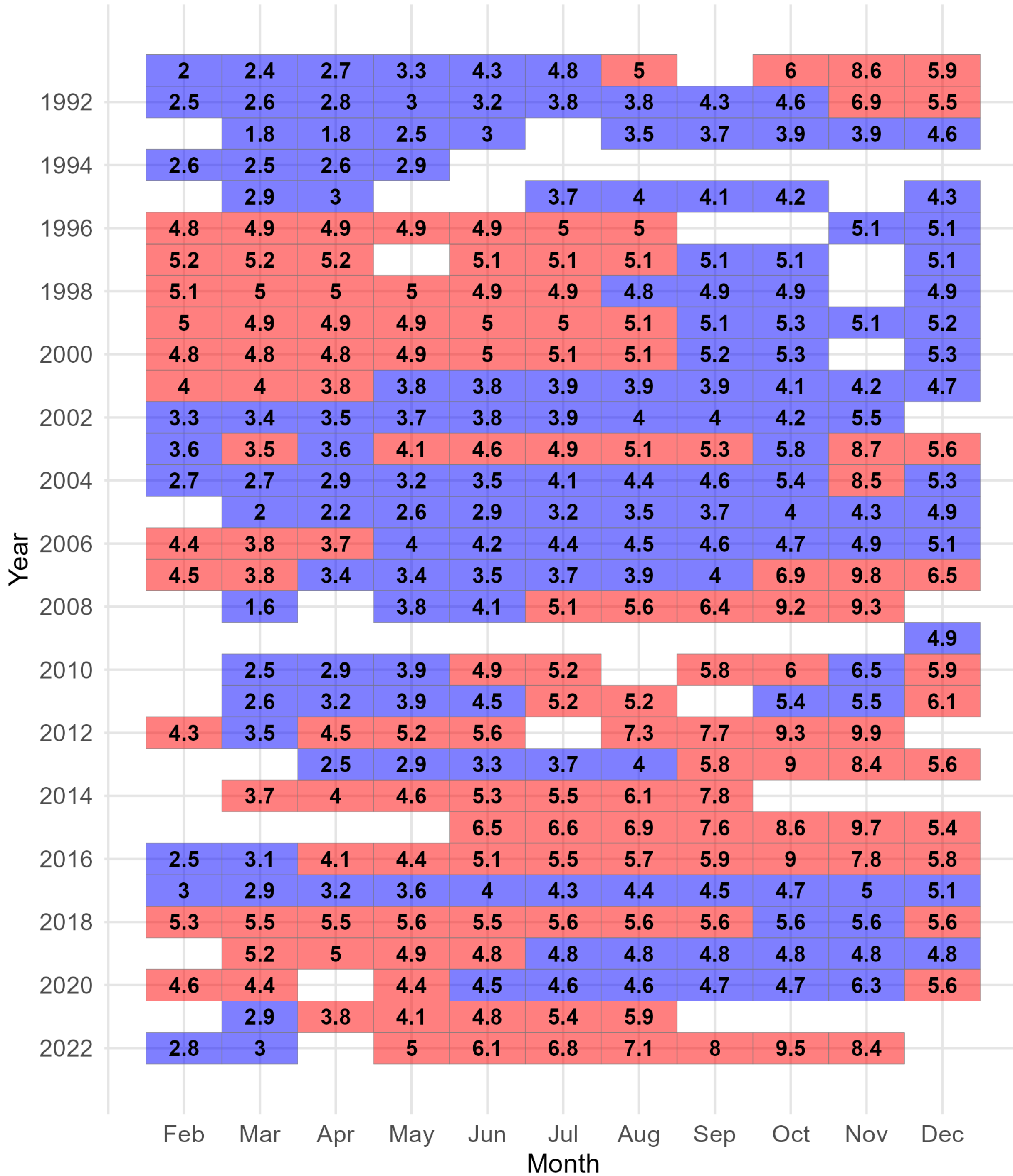


Figure 17. Average Water Temperature (°C) below 20 m at Station 6

Red-colored cells indicate above the long-term average of the respective month while blue-colored cells indicate below the long-term average of the respective month.

Table 3.8. Specific Conductivity (mS/cm at 25°C) Depth Profile at Station 6 in 2022

Depth	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	83.5	83.5	-	84.0	82.9	84.5	84.8	85.0	84.3	84.8	-
2	83.7	84.1	-	84.1	83.9	84.6	84.8	85.3	84.5	85.3	-
3	83.8	85.5	-	84.2	84.4	84.7	84.7	85.4	84.4	85.4	-
4	83.8	83.7	-	84.3	84.6	84.8	84.7	85.5	84.4	85.4	-
5	83.8	83.8	-	85.0	84.6	84.5	84.8	85.5	84.7	85.4	-
6	83.8	83.9	-	85.3	84.4	84.8	85.2	85.6	85.3	85.4	-
7	83.8	83.9	-	84.2	84.1	84.6	85.2	85.6	85.3	85.4	-
8	83.8	83.8	-	84.1	84.2	84.2	85.1	85.6	85.4	85.4	-
9	83.9	83.9	-	84.0	84.2	84.0	84.2	85.6	85.4	85.4	-
10	83.9	83.9	-	84.1	84.0	84.0	83.9	85.5	85.4	85.4	-
11	83.9	83.9	-	84.1	83.7	84.0	84.3	85.6	85.4	85.4	-
12	83.9	83.9	-	84.0	83.9	83.8	84.4	85.5	85.4	85.4	-
13	83.9	83.8	-	83.8	84.0	83.6	84.1	84.9	85.4	85.4	-
14	83.8	83.9	-	83.8	83.9	84.0	84.0	84.5	85.4	85.4	-
15	83.9	83.8	-	84.0	84.1	83.8	84.2	83.9	85.1	85.4	-
16	83.8	83.9	-	83.9	84.1	84.0	84.2	83.3	84.3	85.4	-
17	83.9	83.9	-	83.9	84.0	84.1	84.1	84.2	84.8	85.4	-
18	83.9	83.9	-	83.9	83.9	84.0	84.1	84.6	84.8	85.4	-
19	83.9	83.8	-	83.8	83.9	84.0	84.1	84.6	84.8	85.4	-
20	83.9	83.8	-	83.8	83.9	84.1	84.0	84.5	84.8	85.4	-
21	83.9	83.9	-	83.9	83.9	84.0	84.1	84.6	84.8	85.5	-
22	83.9	83.9	-	84.0	84.0	84.0	84.1	83.9	84.8	85.5	-
23	83.9	83.8	-	83.9	84.0	84.1	84.1	83.8	84.6	85.5	-
24	83.9	83.9	-	84.0	84.0	84.0	84.1	84.0	84.8	85.4	-
25	83.9	83.9	-	83.9	84.0	84.0	84.0	84.1	84.8	85.4	-
26	83.9	83.9	-	83.9	84.0	84.0	84.1	84.1	84.8	85.5	-
27	83.9	83.9	-	83.9	84.0	84.0	84.0	84.2	84.7	85.5	-
28	83.9	83.9	-	83.9	84.0	84.0	84.1	83.9	84.8	85.5	-
29	83.9	83.9	-	83.9	84.0	84.0	84.1	84.1	84.8	85.5	-
30	83.9	83.9	-	84.0	83.9	84.0	84.1	84.2	84.8	85.5	-
31	83.9	83.9	-	83.8	84.0	84.0	84.1	84.2	84.9	85.5	-
32	83.9	83.9	-	83.9	84.0	84.0	84.1	84.0	84.8	85.5	-
33	83.9	83.9	-	83.9	84.0	84.0	84.1	84.1	85.0	85.5	-
34	83.9	83.9	-	83.9	83.9	84.0	84.0	84.1	84.8	85.5	-
35	83.9	83.9	-	83.9	83.9	84.0	84.0	84.2	84.9	85.5	-
36	83.9	83.9	-	83.9	84.0	84.0	84.0	84.1	84.9	85.5	-
37	83.9	83.9	-	83.9	84.0	83.8	84.0	84.2	84.9	85.5	-
38	83.9	83.9	-	83.9	83.9	84.0	84.0	84.2	84.8	85.5	-

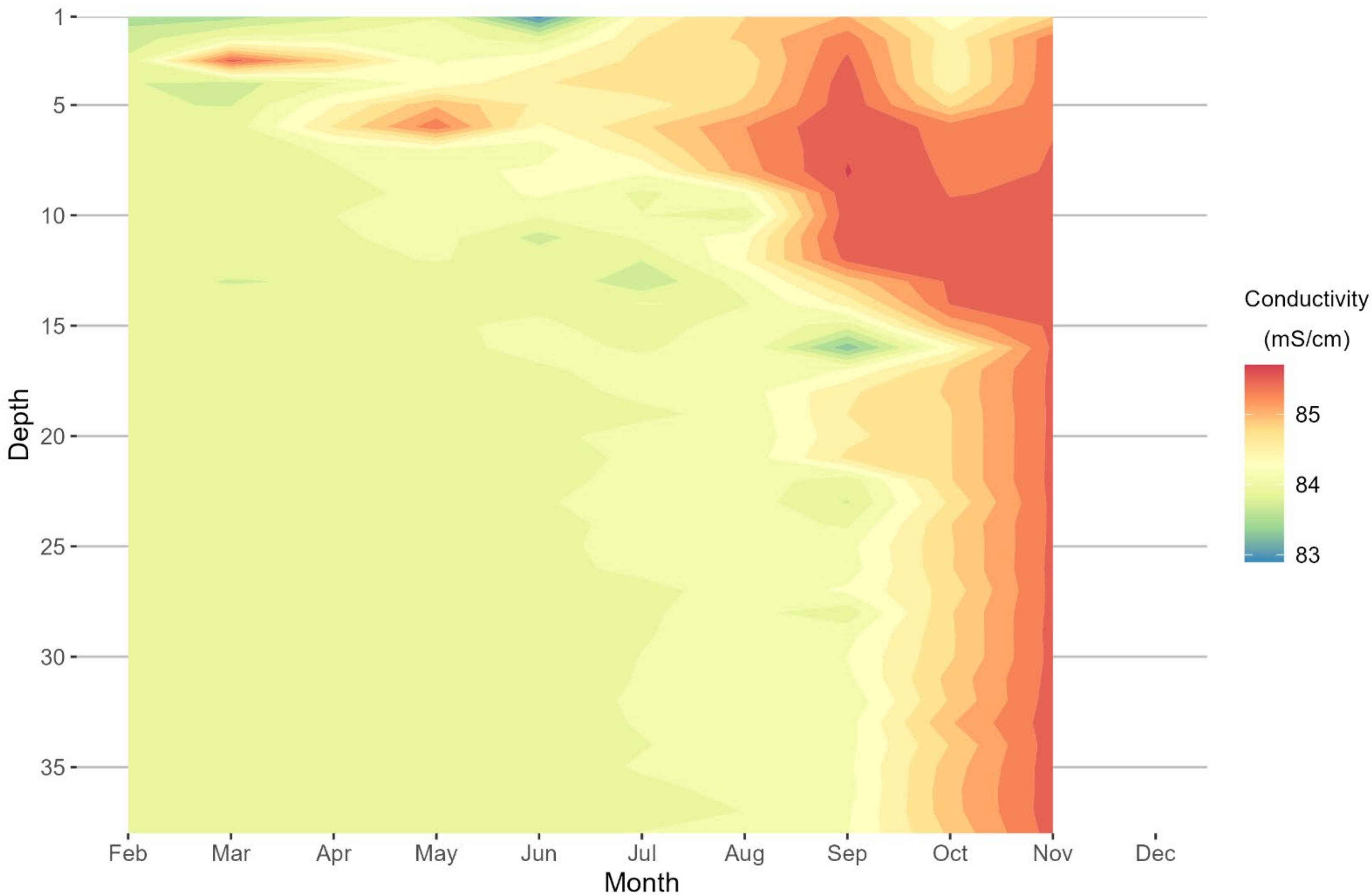


Figure 18. Conductivity (mS/cm) Depth Profile at Station 6 in 2022

Salinity

Salinity expressed in g/L at two different depth classes: epilimnion (between 1 and 10 m) and hypolimnion (below 20 m) are presented in Figure 19 and Figure 20. In 2022, salinity in the epilimnion started at 86.5 g/L in February and increased progressively to 88.8 g/L in November except for a slight dip in June due to the influx of freshwater derived from melting snow. The salinity in the epilimnion has been steadily increasing since the end of the latest meromixis in 2020 with a net increase of 10.9 g/L from the lowest salinity of 77.9 g/L in August, 2019, to 88.8 g/L in November, 2022. Salinity in the hypolimnion also progressively increased from 86.7 g/L in February to 89.0 g/L in November in 2022. The lowest salinity in the hypolimnion was reached at 84.5 g/L toward the end of the latest meromixis in the late 2020 and early 2021, resulting in a net increase of 4.5 g/L. Currently, salinity is 88.8 g/L and 89 g/L in the epilimnion and hypolimnion, respectively, and these values were last observed in 2015 during the five-year drought. Salinity in the epilimnion is expected to decrease in 2023 while hypolimnetic salinity may remain relatively stable in 2023 if Mono Lake enters the sixth known meromictic event due to high snowmelt runoff and precipitation accompanying a series of atmospheric river events last winter.

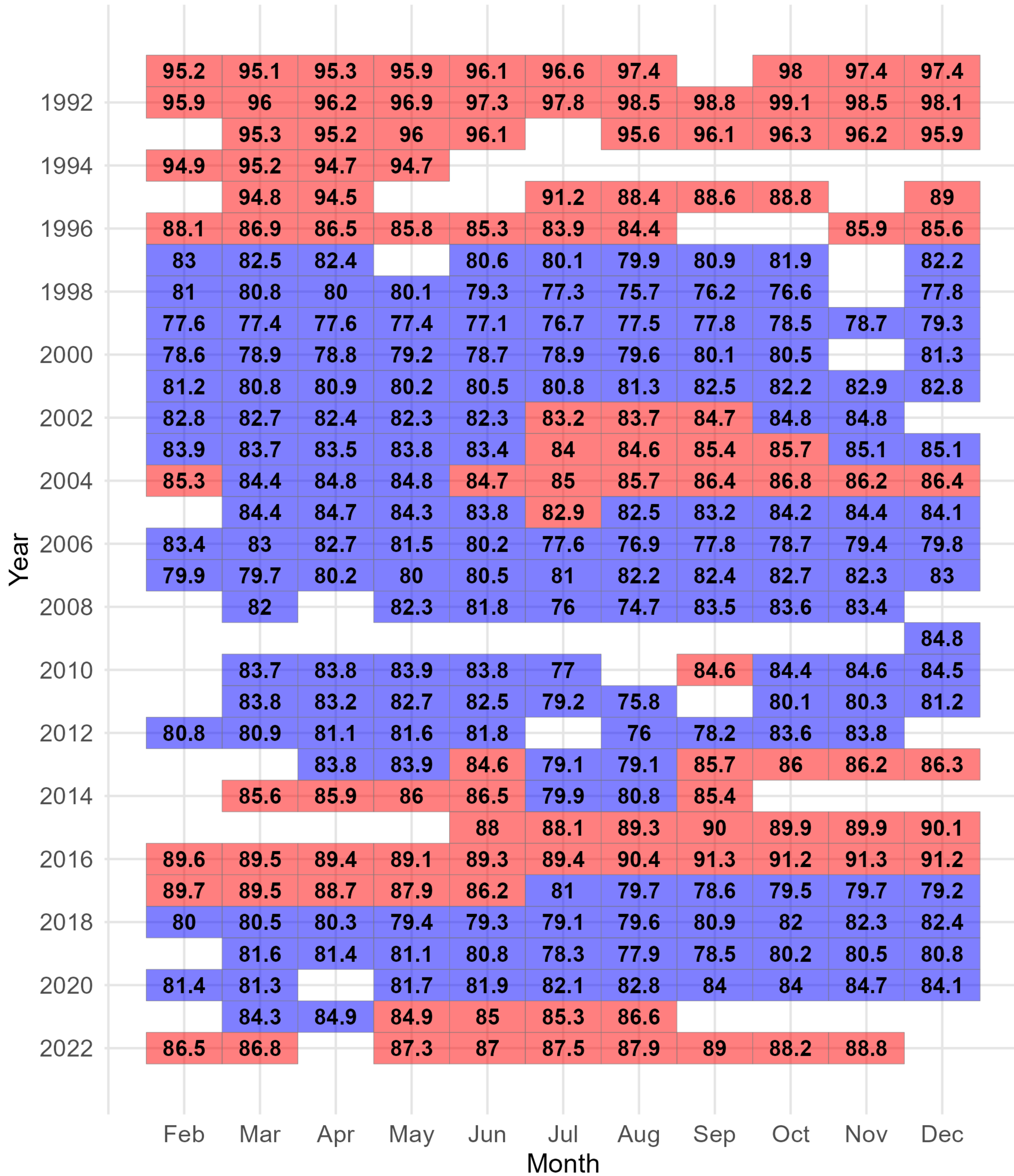


Figure 19. Average Salinity (g/L) between 1 and 10 m at Station 6

Red-colored cells indicate above the long-term average of the respective month while blue-colored cells indicate below the long-term average of the respective month.

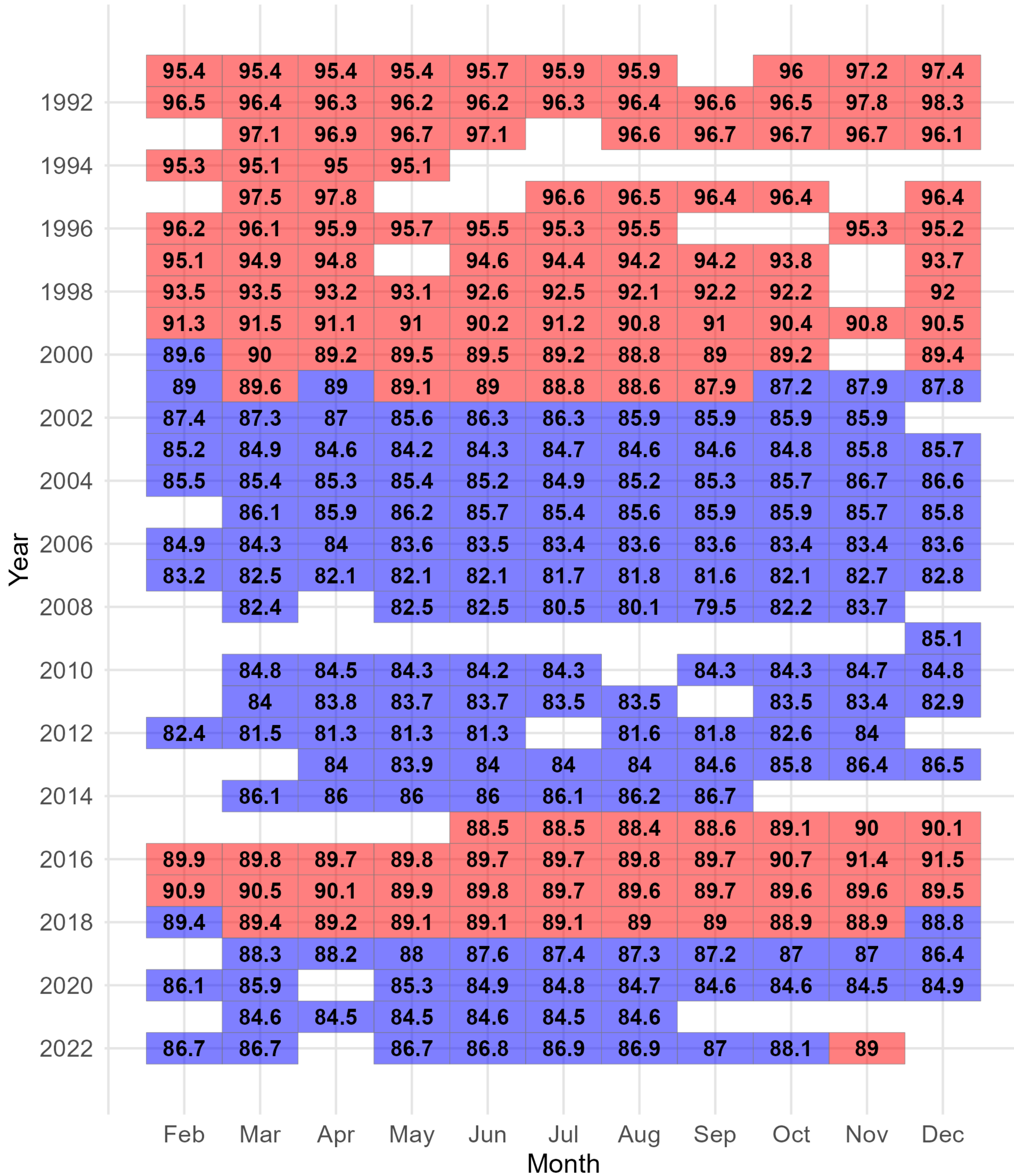


Figure 20. Average Salinity (g/L) below 20m at Station 6

Red-colored cells indicate above the long-term average of the respective month while blue-colored cells indicate below the long-term average of the respective month.

Dissolved Oxygen

Dissolved oxygen (DO) levels in the upper mixed layer (< 10 m) ranged from 4.1 mg/L, the lowest concentration found in July and to 12.8 mg/L, the highest concentration found in May (Table 3.9,

Figure 21). Warming water temperature along with less intense grazing by *Artemia* leads to increased primary production and algal abundance, resulting in more dissolved oxygen in the upper mixing layer. A combination of decreased primary production with intensifying grazing, increased respiration of *Artemia* and other organisms, and higher water temperature decreases dissolved oxygen the upper mixing layer during summer months. Dissolved oxygen concentration was relatively high and uniform throughout the water column in February and March. In May, dissolve oxygen was above 10mg/L from the surface to 7 m. Dissolved oxygen in both epilimnion and hypolimnion declined throughout the summer, and dissolved oxygen below 1 mg/L at the depth of 16 m in June and 8 m in July. From August to November dissolved oxygen continued to rise and became uniform in November at concentrations ranging between 4.0 mg/L to 4.6 mg/L.

Table 3.9. Dissolved Oxygen* (mg/L) Depth Profile at Station 6 in 2022

Depth	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
0	8.8	8.9	-	12.8	6.7	4.1	5.0	-	5.5	4.2	-
1	9.7	9.8	-	12.9	6.5	4.1	5.0	-	5.6	4.6	-
2	9.0	10.0	-	12.8	6.0	4.1	4.8	-	5.2	4.4	-
3	9.1	9.2	-	12.1	5.2	4.0	4.4	-	5.2	4.2	-
4	8.8	8.9	-	12.3	4.8	3.8	4.4	-	4.4	4.1	-
5	8.6	8.2	-	11.7	4.7	3.3	4.4	-	3.7	4.2	-
6	8.3	7.7	-	11.1	3.7	3.0	3.7	-	3.7	4.2	-
7	8.4	7.5	-	10.1	3.5	2.3	2.5	-	3.6	4.2	-
8	8.3	7.5	-	9.7	3.1	0.6	1.6	-	3.5	4.2	-
9	8.3	7.2	-	9.4	2.6	0.3	1.3	-	3.3	4.3	-
10	8.1	7.2	-	9.2	2.4	0.3	1.4	-	3.7	4.2	-
11	7.7	7.2	-	9.1	2.2	0.3	1.4	-	3.9	4.2	-
12	7.4	7.2	-	8.8	1.9	0.3	1.4	-	3.9	4.2	-
13	7.3	6.6	-	8.3	1.8	0.3	1.4	-	4.0	4.2	-
14	8.0	6.6	-	8.3	1.5	0.3	1.5	-	3.9	4.2	-
15	7.7	6.7	-	8.0	1.1	0.3	1.5	-	2.7	4.1	-
16	7.6	6.6	-	7.8	0.9	0.3	1.5	-	2.0	4.0	-
17	7.7	6.8	-	7.5	0.7	0.3	1.5	-	2.0	4.0	-
18	7.6	6.8	-	7.5	0.7	0.3	1.5	-	2.0	4.0	-
19	7.5	6.6	-	7.2	0.6	0.3	1.5	-	2.0	4.5	-
20	7.2	6.2	-	7.2	0.6	0.3	1.5	-	2.0	4.2	-
21	7.1	5.7	-	7.1	0.5	0.3	1.5	-	2.0	4.2	-
22	7.0	5.9	-	7.1	0.2	0.3	-	-	2.0	4.2	-
23	7.2	6.0	-	7.0	0.1	0.3	-	-	2.0	4.2	-
24	6.8	5.7	-	6.9	0.1	0.3	1.5	-	2.0	4.3	-
25	6.7	5.7	-	6.7	0.1	0.3	1.5	-	2.0	4.4	-
26	6.6	5.3	-	6.6	0.1	0.3	1.5	-	2.0	4.3	-
27	6.4	5.3	-	6.5	0.1	0.3	1.5	-	2.0	4.4	-
28	6.2	4.7	-	6.4	0.1	0.3	1.5	-	2.0	4.4	-
29	6.1	4.8	-	6.4	0.1	0.3	1.5	-	2.0	4.4	-
30	6.7	4.9	-	6.3	0.1	0.3	1.5	-	2.0	4.4	-
31	6.2	4.3	-	6.3	0.1	0.3	1.5	-	2.0	4.4	-
32	6.2	4.7	-	6.2	0.1	0.3	1.5	-	2.0	4.5	-
33	6.3	4.7	-	6.0	0.1	0.3	1.5	-	2.0	4.6	-
34	6.2	4.8	-	5.8	0.1	0.3	1.5	-	2.0	4.6	-
35	6.3	4.6	-	4.6	0.1	0.3	1.5	-	2.0	4.5	-
36	6.2	4.7	-	5.4	0.1	0.3	1.5	-	2.0	4.3	-
37	6.3	4.2	-	5.3	0.0	0.3	1.5	-	2.0	4.4	-
38	6.2	4.3	-	5.2	0.0	0.3	1.6	-	2.0	4.4	-
39	-	2.9	-	5.1	0.0	-	1.6	-	-	-	-
40	-	-	-	5.1	-	-	-	-	-	-	-

*YSI probe error (+/- 0.2 mg/L).

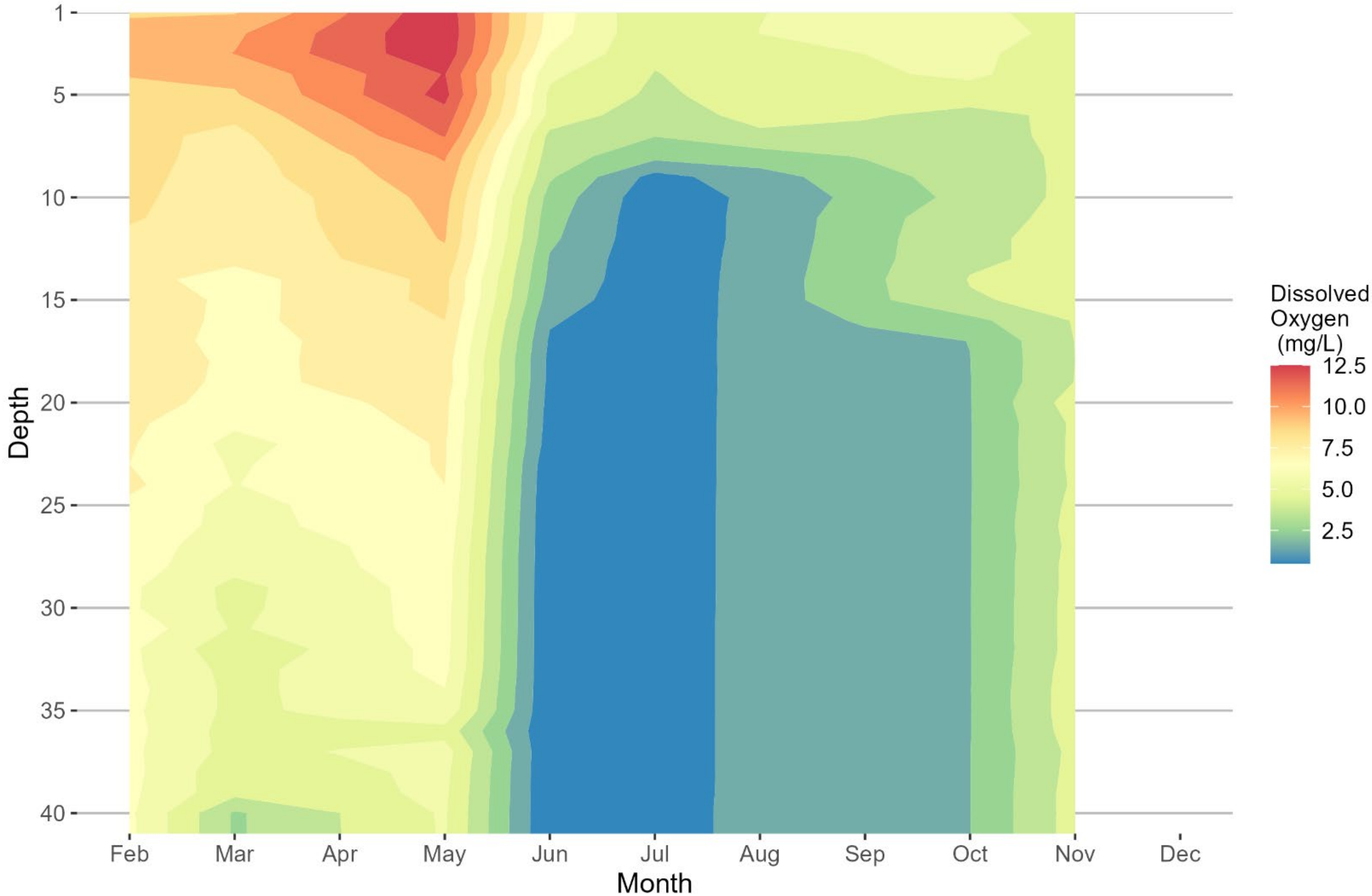


Figure 21. Dissolved Oxygen (mg/L) Depth Profiles at Station 6 in 2022

September DO values were interpolated based on August and October values at each depth.

Artemia Population and Biomass

Artemia population data are presented in Table 3.10 through Table 3.12 with lake-wide means, sector means, associated standard errors and percentage of population by age class. As discussed in previous reports (Jellison and Rose 2011), zooplankton populations can exhibit a high degree of spatial and temporal variability. In addition, when sampling, local convergences of water masses may concentrate shrimp potentially affecting overall means. For these reasons, Jellison and Rose (2011) have cautioned that the use of a single level of significant figures in presenting data is inappropriate, and that the reader should consider the standard error associated with *Artemia* counts when making inferences from the data.

Artemia Population

Hatching of overwintering cysts was observed in February with a high variability among 12 stations in abundance, and accelerated hatching in March (68,699 +/- 14,109 m^{-2}) with the western sector having higher abundance (89,658 +/- 24,244 m^{-2}) compared to the eastern sector (47,740 +/- 10,600 m^{-2}), largely owed to the high abundances found at Station 2 (161,610 m^{-2}) and Station 5 (169,980 m^{-2}). Nauplii during the first two months of monitoring consisted almost exclusively of first and second instars (>98%) (Figure 22). First through fourth stage instars were found in February while only the first three stages were observed in March. For both February and March, the second stage instar was found in higher proportion (62.5% and 58.8%, respectively) than first stage instar. Nauplii abundance continued to increase, and the peak naupliar instar abundance occurred in May (108,856 +/- 27,511 m^{-2}) with the eastern sector showing much higher abundance (181,516 +/- 32,949 m^{-2}) compared to the western sector (36,197 +/- 11,521 m^{-2}).

The first adult was recorded in February at four stations (1 to 2 individuals), but significantly higher numbers of adults did not appear until May (17,807 +/- 2,956 m^{-2}). Adult *Artemia* abundance peaked in July and August as abundance from those two months was almost identical (32,676 +/- 3,312 m^{-2} in July compared to 32,555 +/- 8,613 m^{-2} in August). In July, both western and eastern sectors had similar adult abundances while in August adult abundance in the eastern sector was approximately four times higher than that of western sector.

Fecund females were first recorded in June with 61% of females being ovigerous (carrying eggs). Ovigery increased to 89% in July and remained above 88% through November. Of all ovigerous females, over 90% were found oviparous (carrying cysts) between June and October, ranging from 1,549 +/- 322m^{--2} in May, to the peak of 11,519 +/- 3,633m^{-2} in August, while mostly 2% of ovigerious females were ovoviviparous ranging from 67 +/- 26m^{-2} in October to the peak of 241 +/- 102m^{-2} in

August. Low abundance of ovoviviparous females throughout the year may be enough to replace adult mortality but not enough to result in a secondary or tertiary peak by later generations. It is puzzling that the algal bloom appears to indicate high abundance of food; yet, females carrying cysts was found to be 20 to 50 times higher than ovoviviparous females throughout the year.

The lake-wide mean of brood size ranged from 23.1 eggs per brood in August to 43.1 eggs per brood in October, in spite of August mean female length being 9.9 mm (the second highest mean length to the September mean of 10.3 mm) (Table 3.13, Figure 23). The brood size was 39.6 in June, declined to the lowest size in August, and rebounded since then. The concave pattern inversely related to the adult population abundance.

<u>Biomass</u>

Mean lake-wide *Artemia* biomass rapidly increased from 4.16 g/m^2 in April to the peak of 29.1 g/m^2 in July with an increase in adult proportions (Table 3.14). The lake-wide peak biomass had a much shaper peak in July and quickly dropped by 20% to 23.1 g/m^2 in August while the lake-wide population abundance shows broader peak over July and August with almost identical abundance for all adult categories. The biomass in the eastern sector was consistently found higher than that in the western sector except in March and July. In the latter month, biomass values were almost identical in both sectors, indicating higher productivity in the eastern sector as had been observed in past years. The biomass peak in the eastern sector was observed in August at 37.9 g/m^2, 8.8 g/m^2 higher than the eastern sector peak of 29.1 g/m^2.

Table 3.10. *Artemia* Lake-wide and Sector Population Means (per m^2 or m^{-2}) in 2022

	Instars		Adult	Adult	Adult Female	Ad Female Ovigery Classification				Total
	1-7	8-11	Total	Males	Total	empty	undif	cysts	naup	*Artemia*
Lake-wide										
Feb	18,236	0	8	8	0	0	0	0	0	18,244
Mar	68,699	0	0	0	0	0	0	0	0	68,699
Apr	-	-	-	-	-	-	-	-	-	-
May	108,856	2,213	496	496	0	0	0	0	0	111,566
Jun	15,332	15,855	17,807	14,990	2,817	1,107	80	1,549	80	48,994
Jul	6,653	3,836	32,676	19,624	13,052	1,422	80	11,362	188	43,166
Aug	1,368	543	32,555	19,007	13,548	1,462	325	11,519	241	34,467
Sep	761	439	19,477	10,763	8,714	327	262	7,995	131	20,677
Oct	1,046	218	7,327	3,737	3,590	84	80	3,358	67	8,592
Nov	597	200	780	404	376	42	15	290	29	1,576
Dec	-	-	-	-	-	-	-	-	-	-
Western Sector										
Feb	17,089	0	7	7	0	0	0	0	0	17,096
Mar	89,658	0	0	0	0	0	0	0	0	89,658
Apr	-	-	-	-	-	-	-	-	-	-
May	36,197	80	27	27	0	0	0	0	0	36,304
Jun	10,020	12,314	15,292	12,032	3,260	1,087	161	2,012	0	37,626
Jul	6,600	3,810	34,071	21,435	12,636	1,771	54	10,597	215	44,480
Aug	1,127	255	12,260	8,370	3,890	295	114	3,347	134	13,642
Sep	879	262	16,258	8,930	7,327	64	282	6,922	60	17,398
Oct	443	101	2,616	1,378	1,237	67	13	1,110	47	3,159
Nov	285	80	446	265	181	20	7	144	10	812
Dec	-	-	-	-	-	-	-	-	-	-
Eastern Sector										
Feb	19,383	0	10	10	0	0	0	0	0	19,393
Mar	47,740	0	0	0	0	0	0	0	0	47,740
Apr	-	-	-	-	-	-	-	-	-	-
May	181,516	4,346	966	966	0	0	0	0	0	186,828
Jun	20,644	19,396	20,322	17,948	2,374	1,127	0	1,087	161	60,362
Jul	6,707	3,863	31,281	17,814	13,467	1,073	107	12,126	161	41,851
Aug	1,610	832	52,850	29,645	23,206	2,629	537	19,691	349	55,292
Sep	644	617	22,696	12,596	10,101	590	241	9,068	201	23,957
Oct	1,650	335	12,039	6,097	5,942	101	148	5,607	87	14,024
Nov	909	319	1,113	543	570	64	23	436	47	2,341
Dec	-	-	-	-	-	-	-	-	-	-

Table 3.11. Standard Errors (SE) of *Artemia* Sector Population Means (per m^2 or m^{-2}) from Table 3.10 in 2022

	Instars		Adult	Adult	Adult Female	Ad Female Ovigery Classification				Total
	1-7	8-11	Total	Males	Total	empty	undif	cysts	naup	*Artemia*
Lake-wide										
Feb	2,916	0	4	4	0	0	0	0	0	2,914
Mar	14,109	0	0	0	0	0	0	0	0	14,109
Apr	-	-	-	-	-	-	-	-	-	-
May	27,511	995	237	237	0	0	0	0	0	28,438
Jun	2,936	2,992	2,956	2,781	463	315	53	322	80	7,740
Jul	766	373	3,312	1,864	1,657	261	42	1,498	74	3,640
Aug	356	262	8,613	4,570	4,302	543	123	3,633	102	8,998
Sep	193	174	6,637	3,538	3,115	180	112	2,891	47	6,849
Oct	315	68	2,008	986	1,043	38	36	991	26	2,360
Nov	183	54	187	87	104	13	6	89	9	341
Dec	-	-	-	-	-	-	-	-	-	-
Western Sector										
Feb	4,117	0	4	4	0	0	0	0	0	4,118
Mar	24,244	0	0	0	0	0	0	0	0	24,244
Apr	-	-	-	-	-	-	-	-	-	-
May	11,521	80	27	27	0	0	0	0	0	11,517
Jun	1,663	1,387	1,056	1,042	511	402	93	481	0	2,743
Jul	898	535	5,491	3,416	2,119	340	54	1,885	107	5,815
Aug	108	115	2,093	1,271	885	54	47	845	54	2,127
Sep	286	206	10,592	5,754	4,846	32	207	4,648	53	11,023
Oct	183	78	1,838	881	961	52	13	858	39	2,087
Nov	84	19	98	64	41	13	4	26	7	190
Dec	-	-	-	-	-	-	-	-	-	-
Eastern Sector										
Feb	4,465	0	7	7	0	0	0	0	0	4,459
Mar	10,600	0	0	0	0	0	0	0	0	10,600
Apr	-	-	-	-	-	-	-	-	-	-
May	32,949	1,590	399	399	0	0	0	0	0	34,059
Jun	4,317	5,611	5,953	5,400	781	550	0	331	161	13,634
Jul	1,331	570	4,164	1,521	2,744	369	68	2,468	110	4,879
Aug	723	506	12,541	6,709	6,582	866	215	5,535	197	13,352
Sep	276	280	8,802	4,540	4,294	337	108	3,836	71	8,976
Oct	508	94	2,341	1,129	1,283	59	60	1,250	37	2,889
Nov	318	83	317	146	175	20	10	160	13	490
Dec	-	-	-	-	-	-	-	-	-	-

Table 3.12. Percentage in Different Classes of *Artemia* Population Means from Table 3.10 in 2022

	Instars		Instar	Adult	Adult	Adult Female	Ad Female Ovigery Classification				Ovigerous
	1-7	8-11	%	Total	Males	Total	empty	undif	cysts	naup	Female%
Lake-wide											
Feb	100	0	100	0.05	0.05	0	0	0	0	0	0
Mar	100	0	100	0	0	0	0	0	0	0	0
Apr	-	-	-	-	-	-	-	-	-	-	-
May	98	2	100	0.4	0.4	0	0	0	0	0	0
Jun	31	32	64	36	31	6	39	5	91	5	61
Jul	15	9	24	76	45	30	11	1	98	2	89
Aug	4	2	6	94	55	39	11	3	95	2	89
Sep	4	2	6	94	52	42	4	3	95	2	96
Oct	12	3	15	85	44	42	2	2	96	2	98
Nov	38	13	51	49	26	24	11	5	87	9	89
Dec	-	-	-	-	-	-	-	-	-	-	-
Western Sector											
Feb	100	0	100	0.04	0.04	0	0	0	0	0	0
Mar	100	0	100	0	0	0	0	0	0	0	0
Apr	-	-	-	-	-	-	-	-	-	-	-
May	100	0.2	100	0.1	0.1	0	0	0	0	0	0
Jun	27	33	59	41	32	9	33	7	93	0	67
Jul	15	9	23	77	48	28	14	0.5	98	2	86
Aug	8	2	10	90	61	29	8	3	93	4	92
Sep	5	2	7	93	51	42	1	4	95	1	99
Oct	14	3	17	83	44	39	5	1	95	4	95
Nov	35	10	45	55	33	22	11	4	90	6	89
Dec	-	-	0	-	-	-	-	-	-	-	0
Eastern Sector											
Feb	100	0	100	0.1	0.1	0	0	0	0	0	0
Mar	100	0	100	0	0	0	0	0	0	0	0
Apr	-	-	-	-	-	-	-	-	-	-	-
May	97	2	99	1	1	0	0	0	0	0	0
Jun	34	32	66	34	30	4	47	0	87	13	53
Jul	16	9	25	75	43	32	8	1	98	1	92
Aug	3	2	4	96	54	42	11	3	96	2	89
Sep	3	3	5	95	53	42	6	3	95	2	94
Oct	12	2	14	86	43	42	2	3	96	1	98
Nov	39	14	52	48	23	24	11	5	86	9	89
Dec	-	-	-	-	-	-	-	-	-	-	-

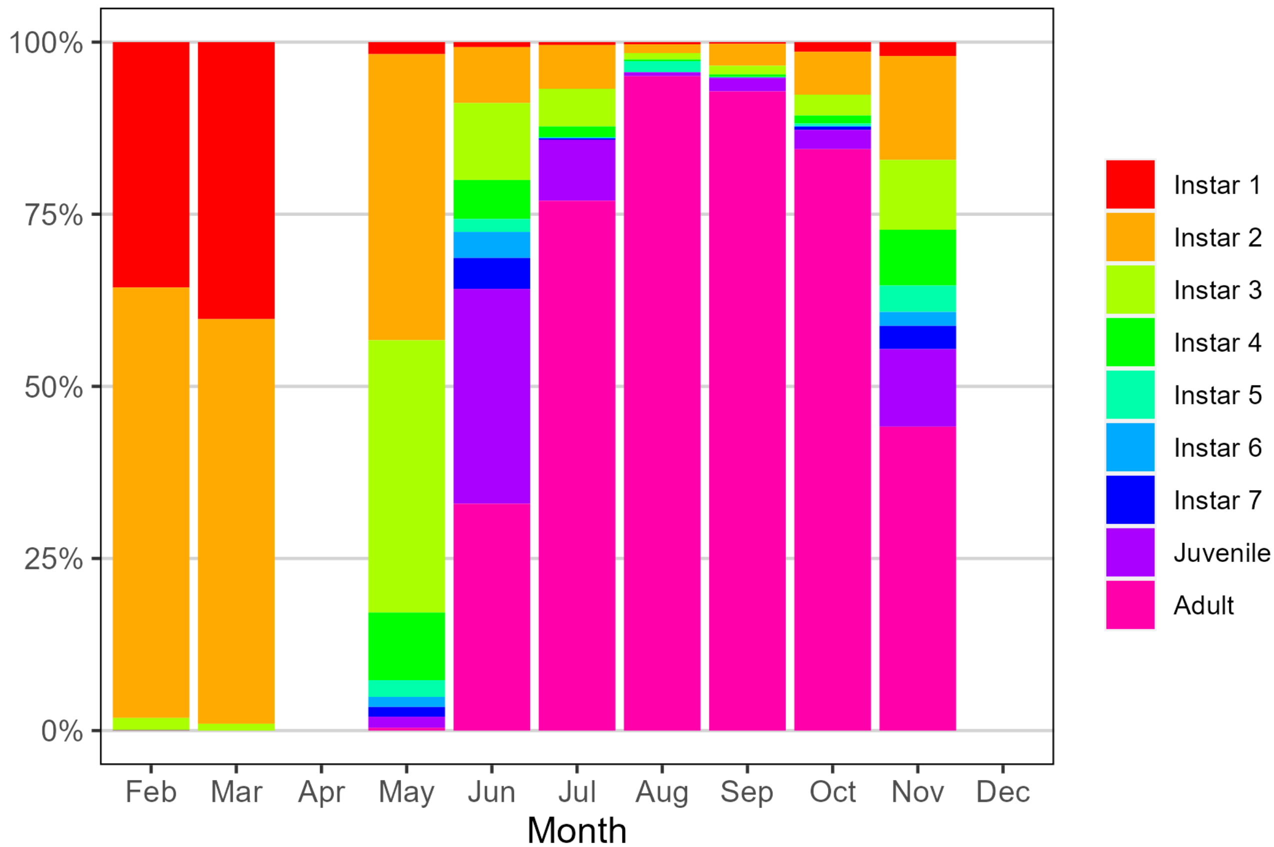


Figure 22. Compositional Changes of *Artemia* Instars and Adults in 2022

Table 3.13. *Artemia* Fecundity Summary in 2022

	# of Eggs/Brood				Female Length (mm)		
Month	Mean	SE	% Cyst	% Indented	Mean	SE	n
Lakewide							
Jun	39.6	2.2	97.5	27.5	9.2	0.1	7
Jul	34.5	1.6	98.6	57.5	9.6	0.1	7
Aug	23.1	1.0	93.8	70.4	9.9	0.1	7
Sep	30.3	1.6	82.4	70.3	10.3	0.1	7
Oct	43.1	3.7	97.7	72.1	9.7	0.1	5
Western Sector							
Jun	38.0	2.6	95.0	20.0	9.1	0.2	4
Jul	34.7	2.2	100	62.2	9.5	0.1	4
Aug	21.3	1.2	91.3	69.6	9.6	0.1	4
Sep	30.5	2.2	81.0	71.4	10.2	0.1	4
Oct	52.0	11.4	100	71.4	9.3	0.2	2
Eastern Sector							
Jun	41.2	3.5	100	35.0	9.3	0.2	3
Jul	34.1	2.4	96.4	50.0	9.8	0.2	3
Aug	25.4	1.6	97.1	71.4	10.3	0.2	3
Sep	30.1	2.2	84.4	68.8	10.5	0.2	3
Oct	41.4	3.8	97.2	72.2	9.8	0.2	3

“n” represents number of stations sampled. 10 individuals were sampled at each station.

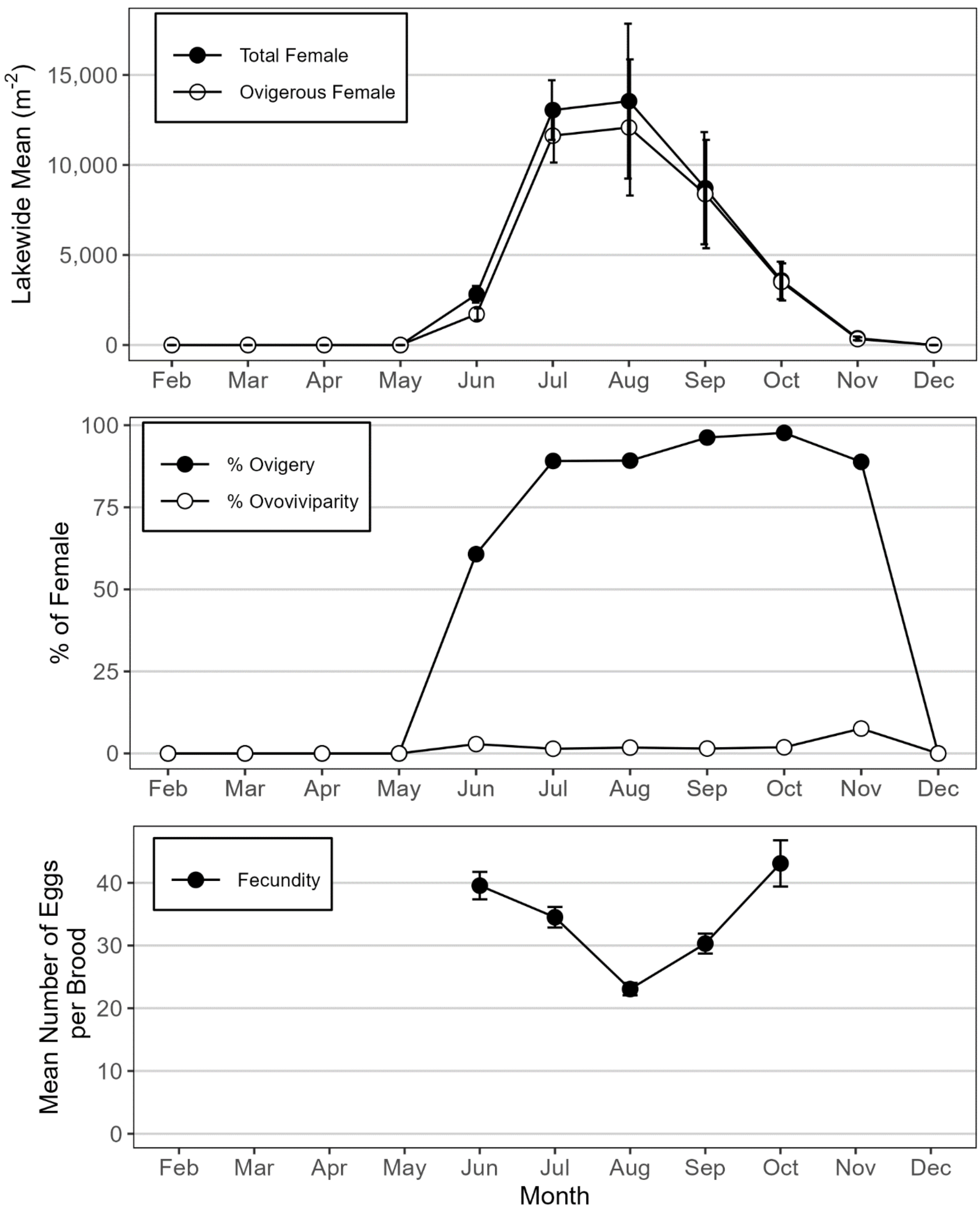


Figure 23. *Artemia* Reproductive Parameters and Fecundity between June and October in 2022

Table 3.14. *Artemia* Mean Biomass (g/m^2) in 2022

Month	Lake-wide	Western Sector	Eastern Sector
Feb	1.42	1.16	1.68
Mar	2.74	3.37	2.11
Apr	-	-	-
May	4.16	1.56	6.77
Jun	18.4	15.7	21.0
Jul	29.1	29.1	29.0
Aug	23.1	8.30	37.9
Sep	14.4	11.8	17.1
Oct	6.29	2.25	10.3
Nov	0.74	0.54	0.93
Dec	-	-	-

Artemia Population Statistics

In 2021, the *Artemia* adult population experienced a post-meromictic peak as the seasonal mean almost doubled from 13,499 m^{-2} in 2020 to 23,908 m^{-2} in 2021 (Table 3.15, Figure 24). The adult population was expected to decline in 2022, as an average decline of 45% in seasonal means has been observed in prior years following a post-meromictic peak. The decline of 6,726 m^{-2}, however, equals only 28% of the decline relative to the peak, the smallest decline among five post-meromictic peaks. The 2022 seasonal mean of 17,182 m^{-2} was higher than what observed in 2010 and 2014, the previous two years which followed the population peak (14,921 m^{-2} and 14,009 m^{-2}, respectively), but lower than the first two years which followed the population peak in 1990 and 2005 (20,223 m^{-2} and 20,031 m^{-2}, respectively). The 2022 seasonal mean was highest since 2011 excluding two post-meromictic peaks in 2013 and 2021, but slightly below the long-term average based on 42 years of record (18,779 m^{-2}) and also the long-term average based on 38 years of non-population peak years (17,450 m^{-2}). The non-population peak year average (38-year average) decreases by 5% to 16,580 m^{-2} when two exceptionally high years (1981 and 1982) are excluded, resulting in 2022 being slightly above average year. In spite of the highest post meromictic peak in 2021 and modest seasonal mean in 2022, the seasonal mean has shown a declining trend since 1979 (Figure 24). The overall decline is 154 m^{-2}/year, equating to a total decline of 6,776 m^{-2} over 44 years ($r = -0.30$, $P = 0.045$). The declining trend is more notable among most meromictic population peaks at 392 m^{-2}/year ($r = -0.97$, $P = 0.0073$) while non-peak years show a moderate decline at 200 m^{-2}/year ($r = -0.46$, $P = 0.003$).

Seasonal peak abundance of 32,676 m^{-2} was highest since 2015 (excluding 2021), but 24% and 17% lower than the 42-year average of 42,758 m^{-2} and the 38-year average of 39,301 m^{-2}, respectively. Seasonal peaks normally coincide with seasonal means ($r = 0.92$), but seasonal peaks show higher variation ranging from 18,511 m^{-2} in 2016 to 105,245 m^{-2} in 1982 with coefficient of variation equating to 48% compared to 35% for seasonal means. Comparing to years which immediately follow the post-meromictic population peak, the 2022 seasonal peak was 24% lower than the average of four such years.

The center of the temporal distribution of adults (centroid) in 2022 was day 223 or August 20, later than day 196 observed in 2021, but again deviating from the declining trend which was present until 2015 (Figure 25). A significant number of adults did not appear until June and peaked in July, but remained at the similar level to this peak through August, shifting centroid into the late August (Figure 26). In 2022, spring hatches started early in February (18,236 m^{-2}, above normal for February), but nauplii abundance did not peak until May (Figure 27). Large spring hatches have been attributed to earlier adult peak or earlier centroid. The May nauplii peak of 108,856 m^{-2}

was highest on record for May, but the timing of the nauplii peak was later than other years. Relatively small spring hatches have been linked to later centroid. Between 2016 and 2019, centroid varied between days 217 and 222 while and nauplii peaks ranged from 45,835 m^{-2} in 2019 to 66,480 m^{-2} in 2017 and remained below the long-term peak nauplii average of 75,600 m^{-2}. The relationship between the size of spring hatches and timing of adult abundance distribution explains some years, but not others, such as 2012 (small nauplii peaks but earlier centroid), 2015 (relatively small nauplii peaks but earlier centroid), and 2020 (relatively large nauplii peaks but later centroid). The 2022 centroid also fell outside this pattern. Nauplii monthly abundance between February and June was compared to adult centroid (Figure 29). Earlier nauplii peaks associated with earlier centroid and small nauplii abundance with later centroid should result in negative correlations in earlier months. Between February and April, correlations were negative with April showing the strongest correlation ($r = -0.41$), supporting the past interpretations, even though correlations were weak. Later nauplii peak (June) appears to be associated with later centroid ($r = 0.43$). Weaker correlations in general, however, have resulted in some deviations from the past interpretations.

A shift of adult seasonal abundance to earlier months appears to have reversed around 2010, especially for May through July, resulting in the S-shaped curve rather than a monotonic increasing trend (Figure 28, Figure 30). From August to November seasonal abundance appears to decrease over time with a slight upward trend toward more recent years, resulting in a convex shape. It appears that around 2010, and after 2010, the trends which have been observed in the past have started to shift for all months.

Table 3.15. Summary Statistics of Adult *Artemia* Abundance

Year	Mean	Median	Peak	Centroid
1979	14,118	12,286	31,700	216
1980	14,643	10,202	40,420	236
1981	32,010	21,103	101,670	238
1982	36,643	31,457	105,245	252
1983	17,812	16,314	39,917	247
1984	17,001	19,261	40,204	212
1985	18,514	20,231	33,089	218
1986	14,667	17,305	32,977	190
1987	24,596	23,722	54,278	225
1988	27,641	25,764	71,630	208
1989	36,121	29,099	92,491	252
1990	20,223	17,563	34,930	229
1991	18,129	19,453	34,565	226
1992	18,947	19,543	34,648	215
1993	14,922	16,266	28,125	217
1994	16,749	19,146	29,018	213
1995	14,968	17,100	26,077	209
1996	17,233	16,465	34,242	215
1997	15,074	16,062	31,791	205
1998	20,185	22,347	37,556	227
1999	19,393	20,572	37,227	225
2000	10,568	9,123	22,384	210
2001	20,031	20,038	38,035	210
2002	12,504	12,679	25,533	200
2003	13,138	13,779	29,510	203
2004	31,108	36,908	75,466	180
2005	20,024	19,803	45,419	193
2006	21,518	20,316	55,748	186
2007	18,826	17,652	41,751	186
2008	12,404	14,140	27,606	188
2009	25,970	17,919	72,086	181
2010	14,921	7,447	46,237	191
2011	21,343	16,893	48,918	194
2012	16,733	11,974	53,813	182
2013	27,068	31,753	57,840	197
2014	14,009	7,942	45,017	194
2015	7,906	6,047	18,699	186
2016	11,157	10,816	18,511	221
2017	15,924	16,372	27,740	222
2018	12,700	12,376	23,239	217
2019	14,241	13,138	28,236	222
2020	13,499	14,035	24,353	210
2021	23,908	24,990	50,731	196
2022	17,182	16,845	32,676	223
Mean	18,779	17,824	42,758	210
Min	7,906	6,047	18,511	180
Max	36,643	36,908	105,245	252

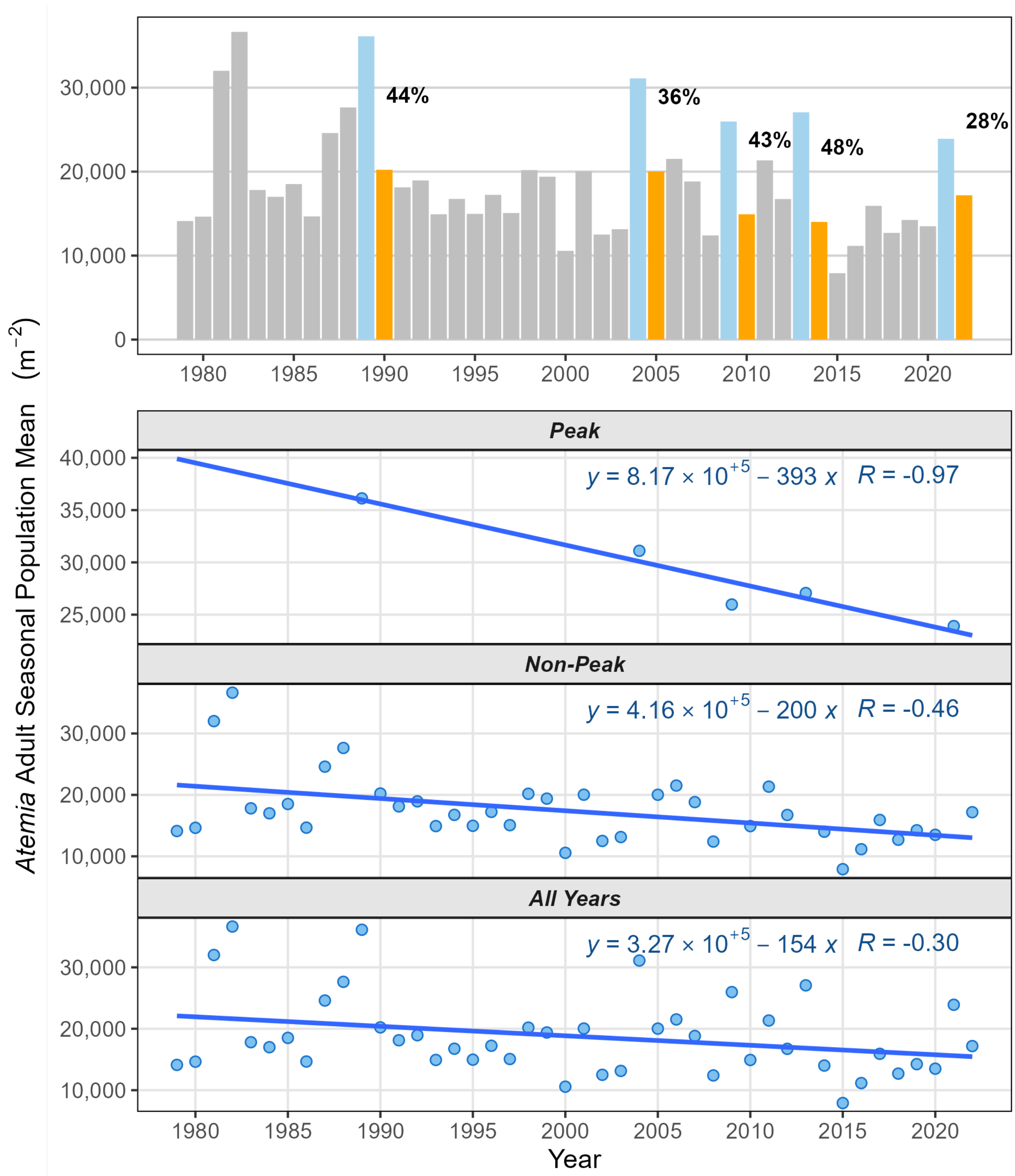


Figure 24. Adult *Artemia* Population Mean since 1979

Bars filled with light blue color indicate years with post meromixis *Artemia* population peaks while bars filled with orange color indicate years which immediately follow the post meromictic peak.

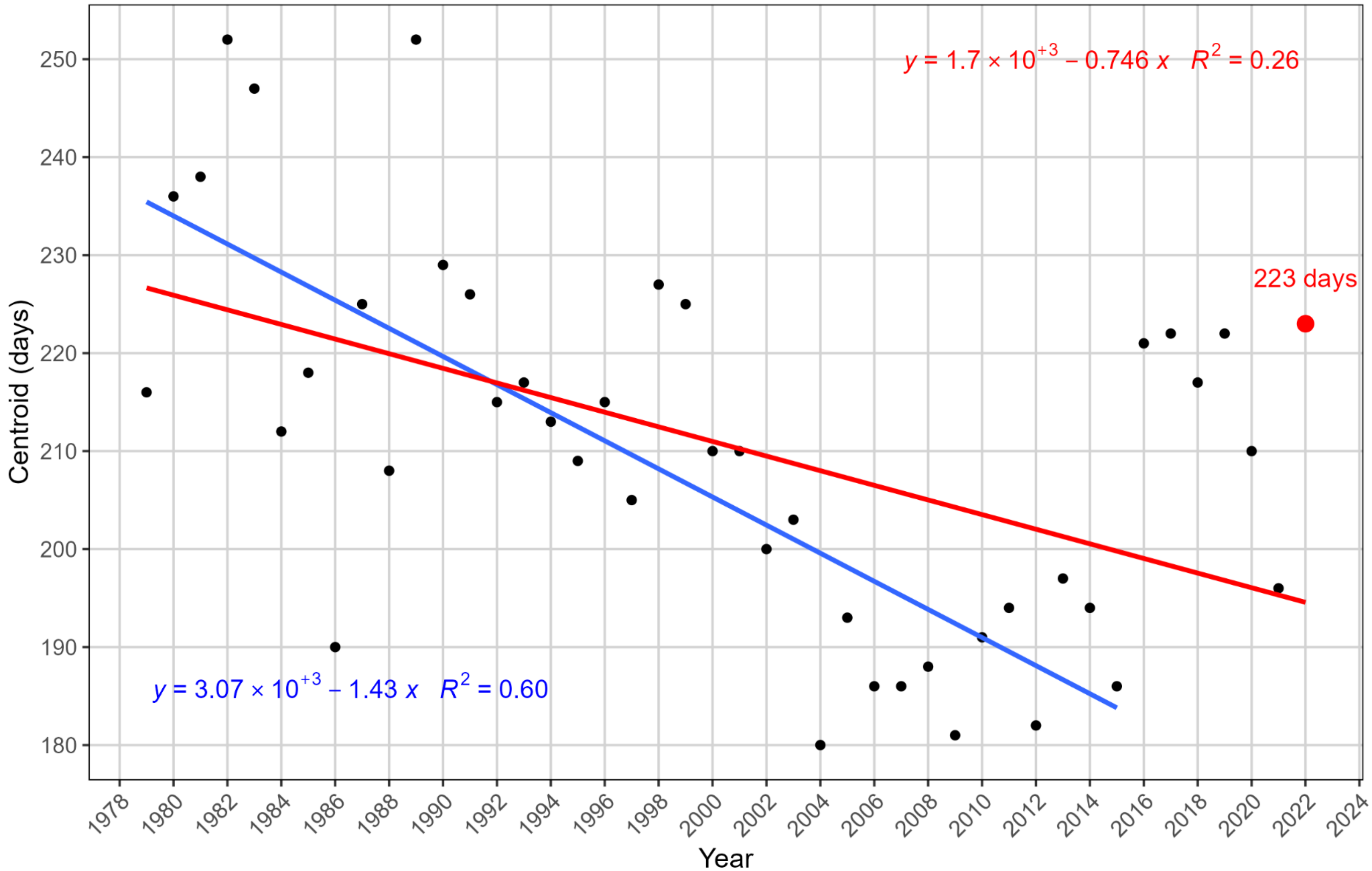


Figure 25. Adult *Artemia* Population Centroid

A red dot indicates a value in 2022. The blue line indicates the linear trend between 1979 and 2015 while the red line indicates the linear trend for all monitoring years.

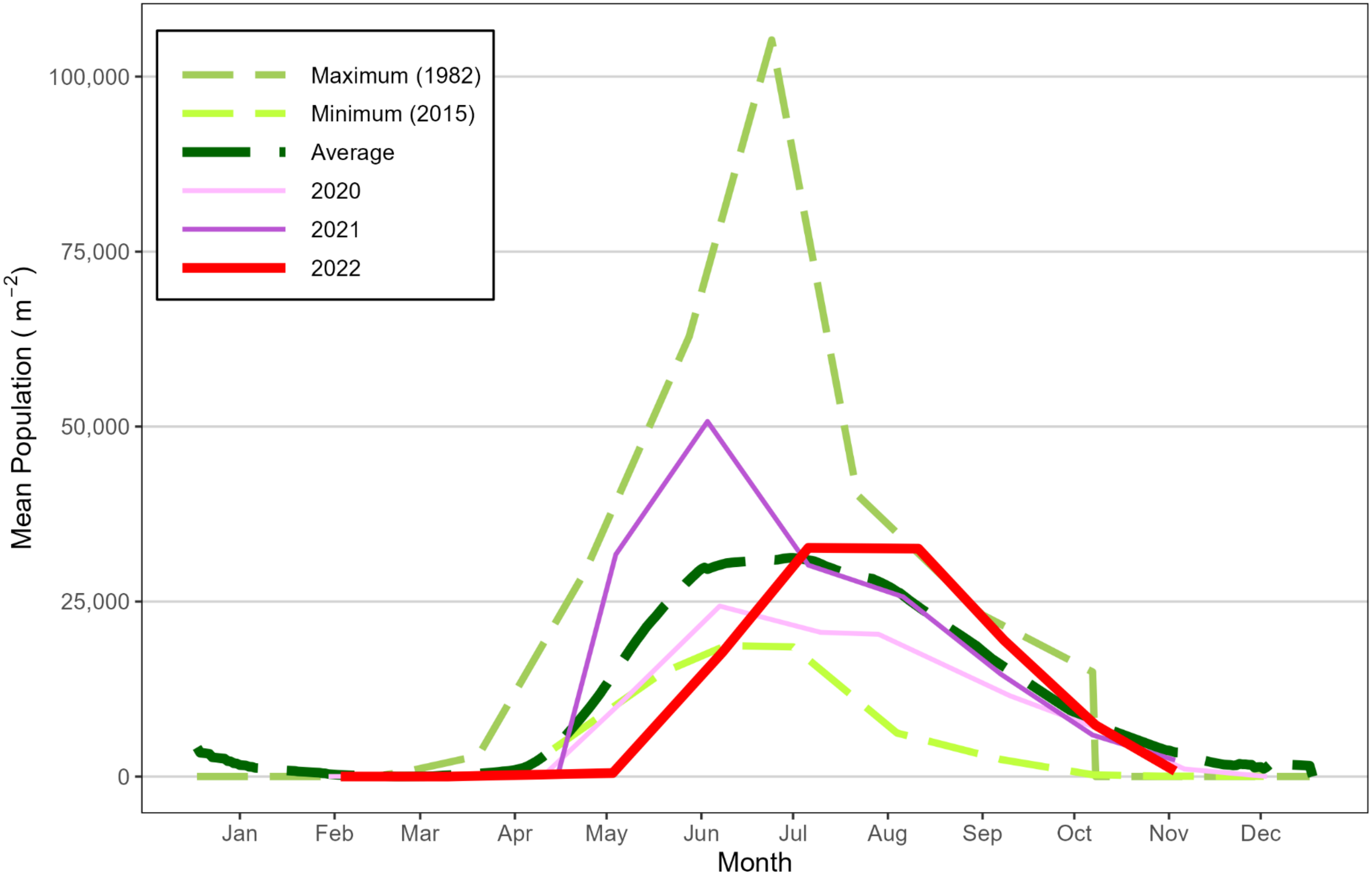


Figure 26. Mean lake-wide Adult *Artemia* Population (m^{-2}) since 1982

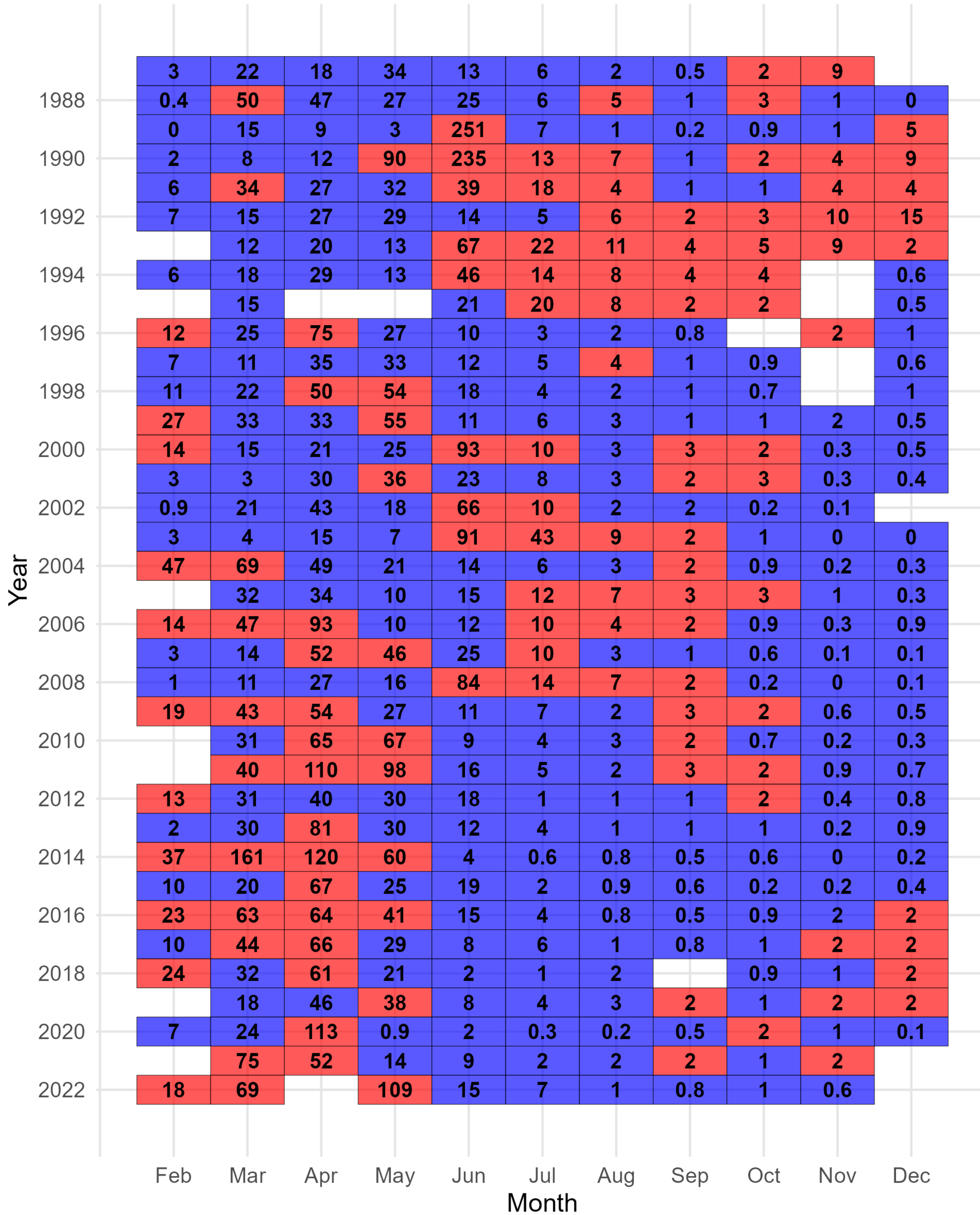


Figure 27. Monthly Average Instars 1-7 *Artemia* Abundance of 12 Stations

Values are in m^{-2} divided by a thousand (e.g. 7.9 = 7,900). Red-colored cells indicate above the long-term average of the respective month while blue-colored cells indicate below the long-term average of the respective month.

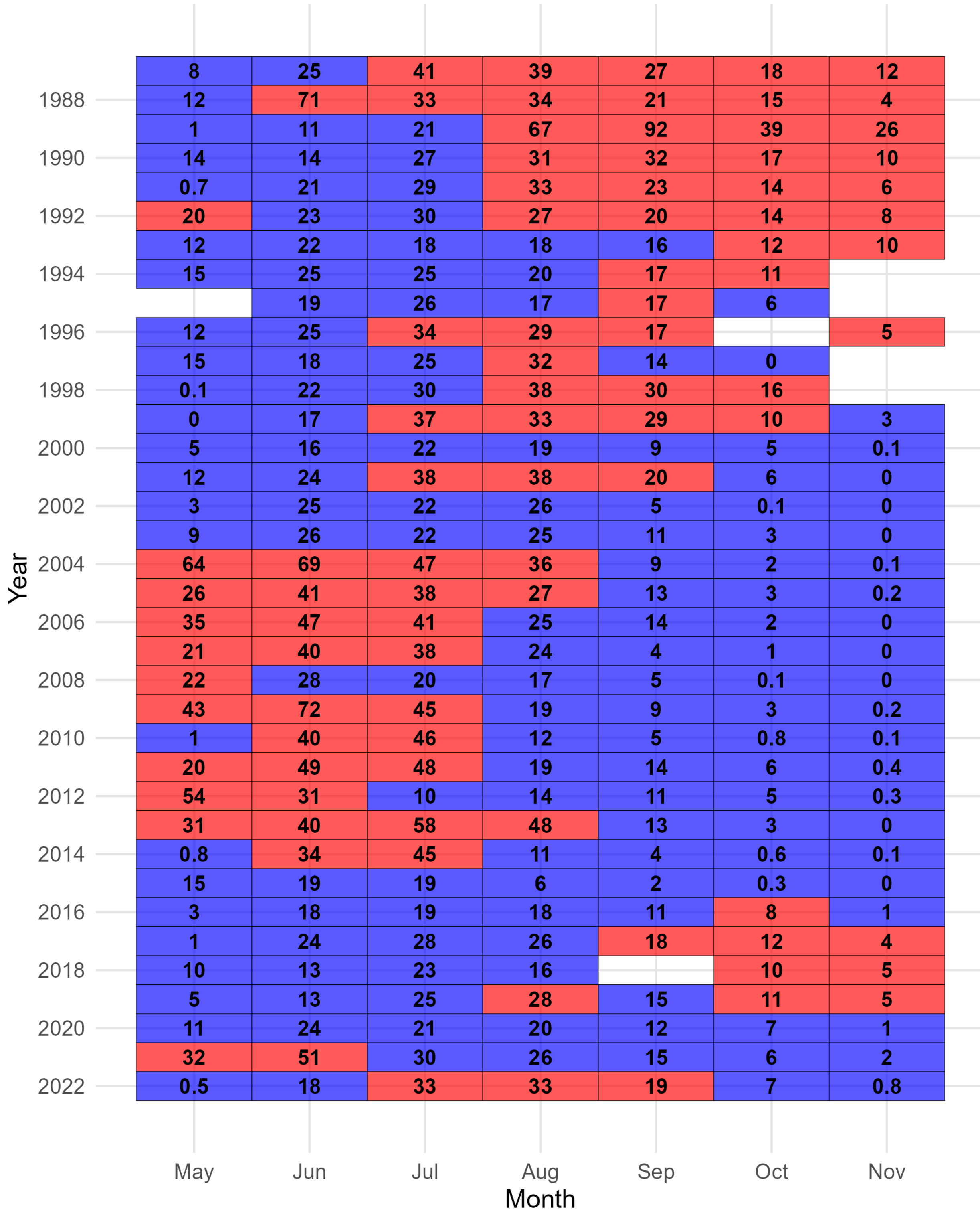


Figure 28. Monthly Average Adult *Artemia* Abundance of 12 Stations

Values are in m^{-2} divided by a thousand (e.g. 7.9 = 7,900). Red-colored cells indicate above the long-term average of the respective month while blue-colored cells indicate below the long-term average of the respective month.

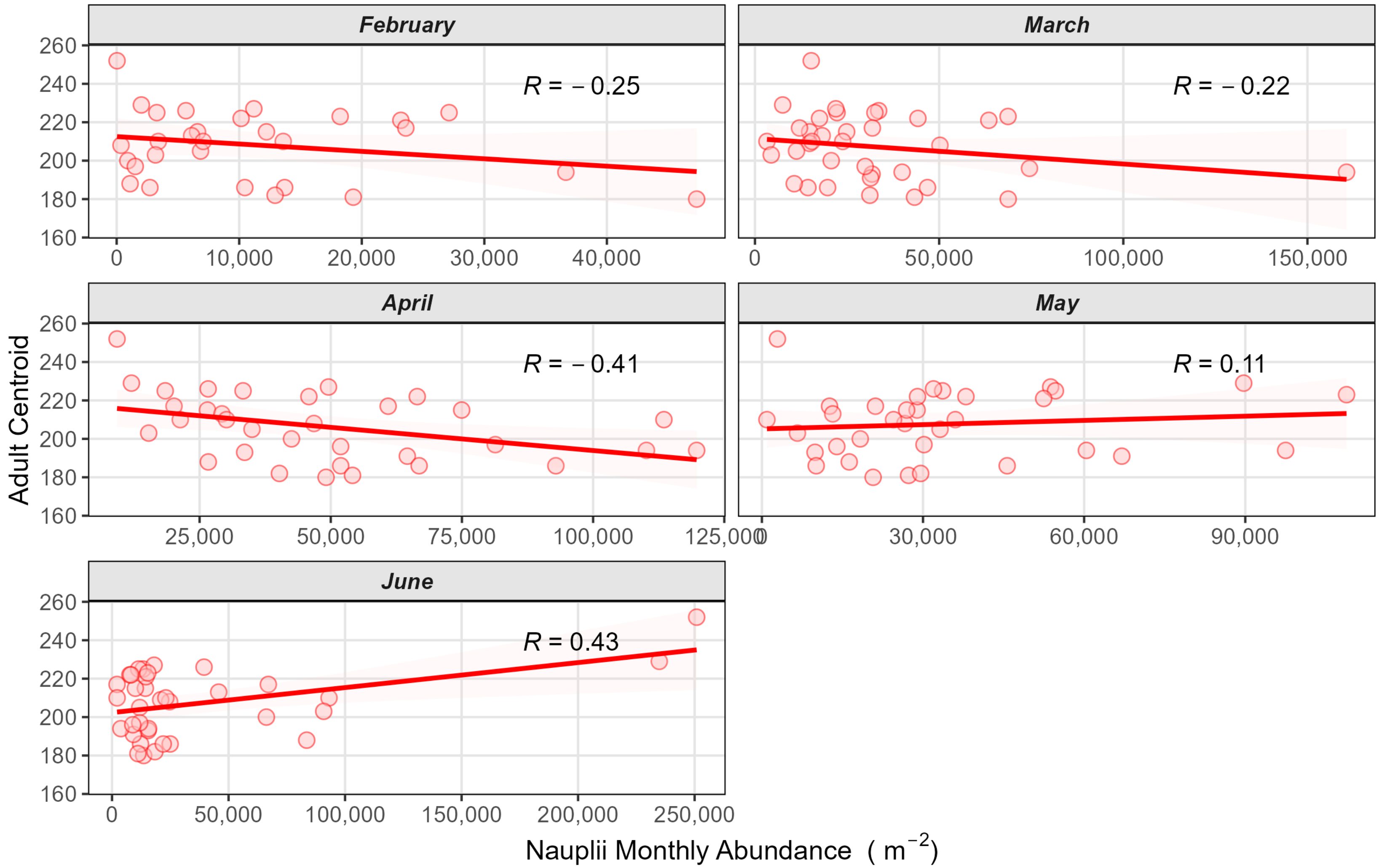


Figure 29. Relationships between Monthly *Artemia* Nauplii Abundance and Adult *Artemia* Centroid

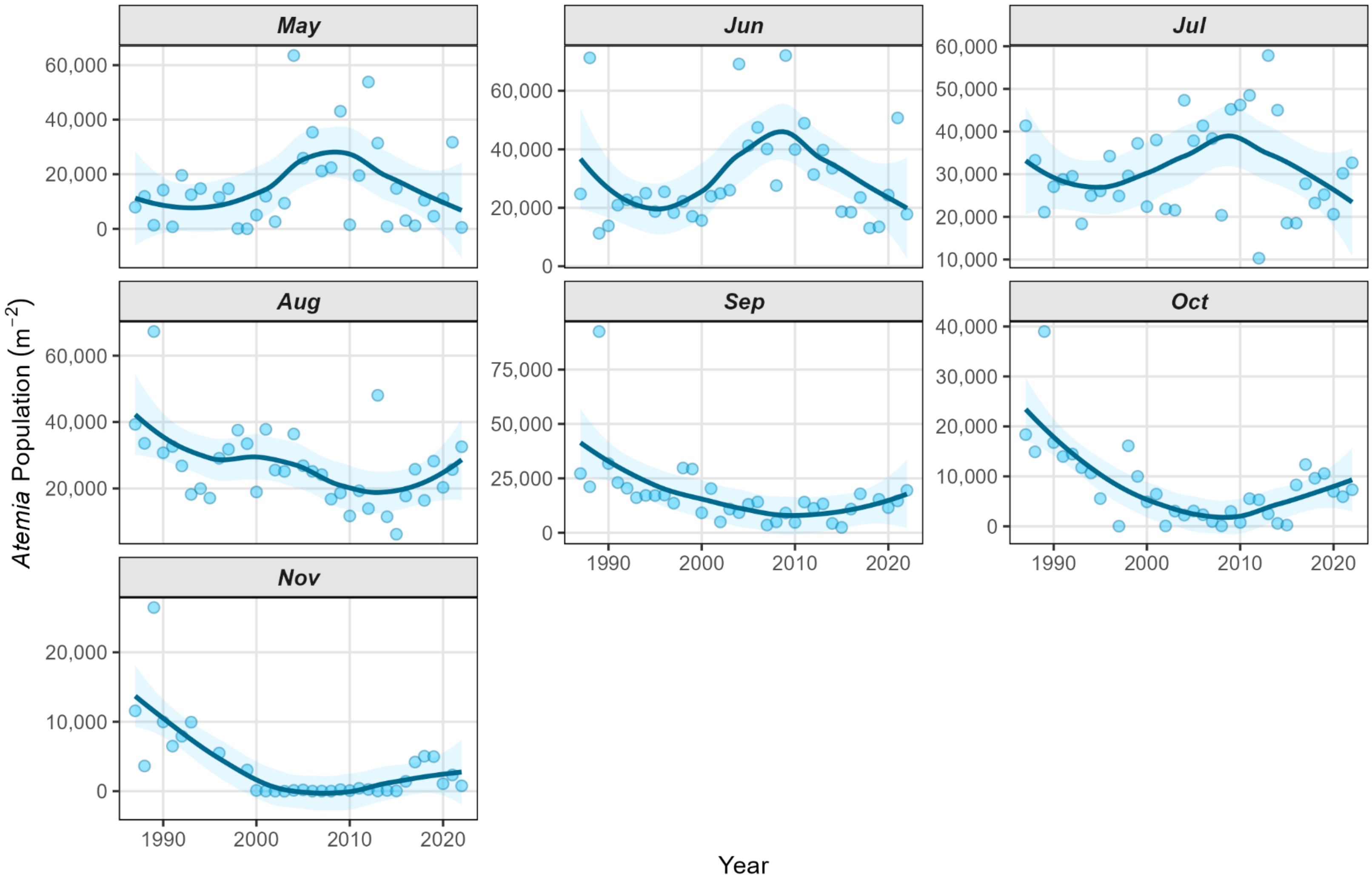


Figure 30. Time Trend of Adult *Artemia* Abundance between May and November

Limnology Data Analysis: Artemia Population and Salinity

Salinity and Mono Lake Elevation/Volume Models

Salinity regression models were built separately accordingly the lake mixing patterns, meromictic phases, and depths, based on the relationship between salinity and lake elevation/cumulative percentage of lake volume between 1991 and 2022 (Table 3-16). A model explains more than 92% of variations in salinity for both upper and lower mixing layers (1 to 10m and below 20m) during monomixis, but the relationship breaks down during meromixis as salinity in mixolimnion decreases quickly, resulting in much lower salinity at a given lake elevation or volume, especially during the formation phase, while salinity in monimolimnion decreases slowly and continuously throughout meromixis regardless of lake elevation. For monomictic years prior to the first recorded meromixis (1983-1987) and between 1988 and 1999, the regression model based on monomictic years between 1991 and 2022 was used to estimate salinity in the upper and lower mixing layers, which explained 92% and 96% of variations in salinity, respectively.

It was stated previously that the first meromixis initiated in 1983, and complete mixing did not occur until November 1988. The Mono Lake elevation data, however, indicate that lake elevation began to rise in July of 1982, instead of 1983; and continuously rose until April of 1984, thus, the duration of the first meromixis was defined as a period starting in July of 1982 and ending in October of 1988. A time period during which lake elevation continuously rises was used as a definition of a formation period, and the remaining time period of the meromixis was used as a definition of a persistence period. For the first meromixis, the formation phase started in July of 1982, and ended in April of 1984, and the persistence phase began in May of 1984, and ended in October of 1988. The regression models were developed based on the phases during the second meromixis whose formation phase started in June of 1995, and ended in August of 1998, and persistence phased started in September of 1988, and ended in October of 2002. Salinity in monimolimnion during meromixis tends to decline monotonically throughout the duration of the meromixis, and this behavior was best described by cumulative freshwater input from two major tributaries (Rush and Lee Vining creeks) among readily available hydrological data. The second order binomial model was developed based on the second meromixis due to slight concavity. In addition to the regression model based on lake elevation and cumulative percentage of lake volume, salinity was also estimated by the salinity-elevation relationship presented by Vorster (1985) following the same steps described above.

Salinity estimates based on elevation and volume were strongly linear and almost identical for the range of Mono Lake elevation since 1991, while Vorster's model consistently resulted in higher salinity estimates except for monimolimnion during meromixis (Figure 31, Figure 32, Table 3-16). Salinity should have been highest in 1981 when the lake level reached its lowest at 6,371.7 ft since the export started, and estimated values for the highest salinity vary among three models, 100.4 g/L ± 2.9 g/L and 100.0 g/L ± 1.9 g/L for mixolimnion and monimolimnion, respectively, based on the elevation model, 99.8 g/L ± 2.9 g/L and 99.4 g/L ± 2.0 g/L for mixolimnion and monimolimnion, respectively, based on the volume model, and 102.6 g/L for Vorster's model. The mixolimnion estimates based on elevation and volume, however, may be slightly underestimating salinity for 1981, as mixolimnion salinity value of 99.1 g/L was recorded in 1992 at the lake level of 6,373.1 ft. Salinity was estimated to rise by 1.7 g/L per each foot of fall in lake elevation in mixolimnion; thus, mixolimnion salinity should have been approximately 2.3 g/L higher than 99.1 g/L, resulting in 101.4g/L, which is higher than the estimated value but within the 95% confidence interval.

The lowest estimated salinity in mixolimnion between 1979 and 1990 should have occurred in 1986 during the first recorded meromixis (1983-1988) at 83.0 g/L ± 5.8 g/L based on the elevation model, at 83.6 g/L ± 5.1 g/L based on the volume model, and at 87.7 g/L based on the Vorster's model, while the lowest estimated salinity in monimolimnion should have occurred in spring of 1989 just after the first recorded meromixis at 92.1 g/L ± 1.9 g/L based on the elevation model, at 92.2 g/L ± 2.0 g/L based on the volume model, and at 94.0 g/L based on the Vorster's model. The values were higher than the lowest observed salinity values of 75.5 g/L in mixolimnion recorded in August, 1988, and 81.3 g/L in monimolimnion recorded in April, 2012.

Table 3.16. Correlation Coefficients and Simple Linear Regression Coefficients and Model Fit

Depth	Mixing	Phase	Variable	r	r^2	Regression
1 to 10m	Monomixis		Elevation	-0.967	0.935	y = -1.7 + 10963
			Volume	-0.965	0.932	y = -0.00004 + 185
	Meromixis	Formation	Elevation	-0.987	0.973	y = -1.84 + 11794
			Volume	-0.987	0.973	y = -0.00004 + 187
		Persistence	Elevation	-0.960	0.919	y = -2.55 + 16366
			Volume	-0.960	0.919	y = -0.00006 + 227
21 to 38m	Monomixis		Elevation	-0.982	0.964	y = -1.6 + 10314
			Volume	-0.980	0.961	y = -0.00004 + 179
	Meromixis		Input	-	0.987	$y = -4.1X^2 - 26.39X + 91.3$

Formation and Persistence indicate the formation and persistence phases of the second recorded meromixis, which lasted from June, 1995, to August, 1998, and from September, 1999, to October, 2002, respectively.
Meromixis for Variable, Input, refers to the second recorded meromixis (1995-2002).
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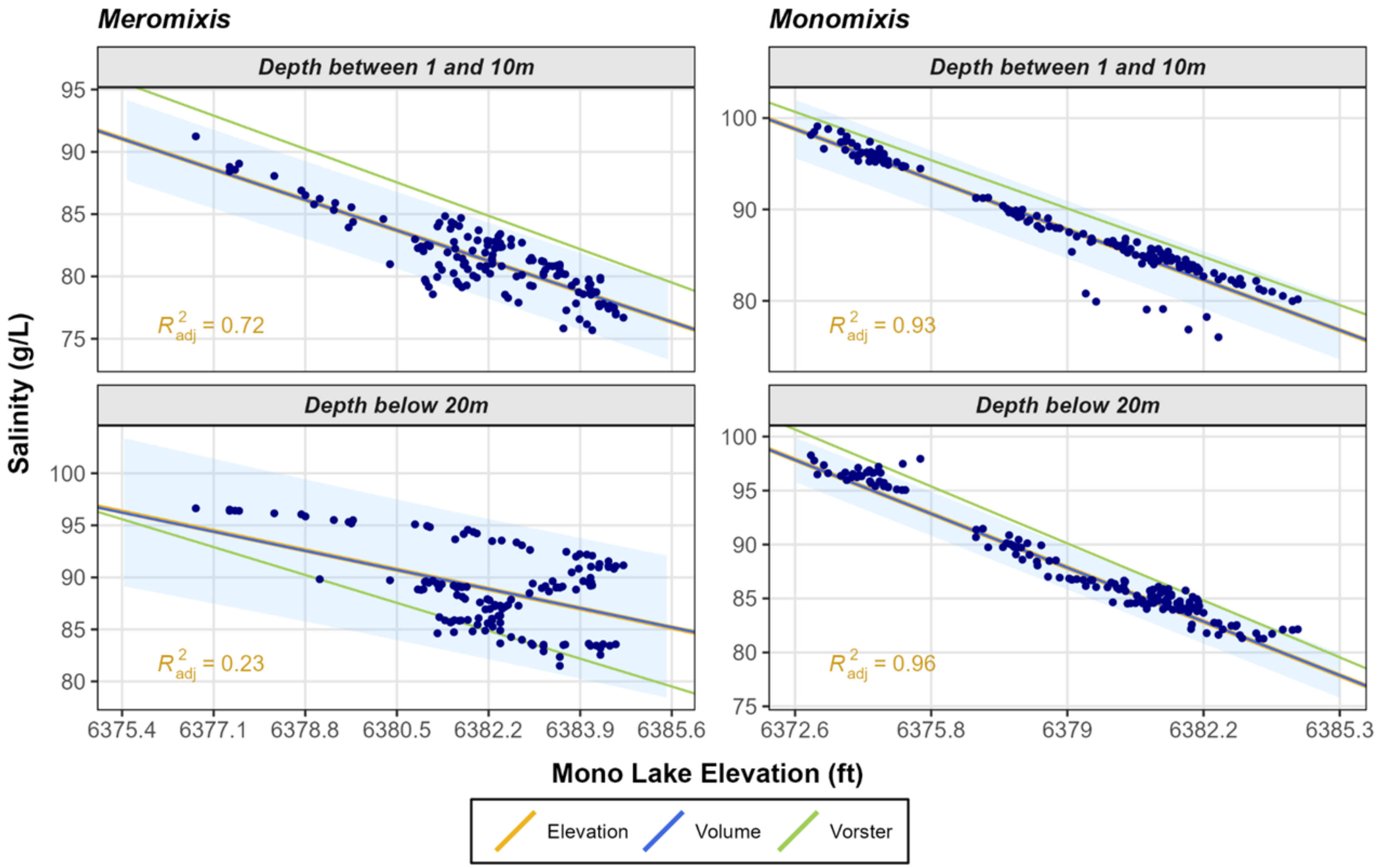

Figure 31. Simple Linear Regression between Salinity and Mono Lake Elevation during Meromixis and Monomixis at Depths between 1 and 10m and below 20m

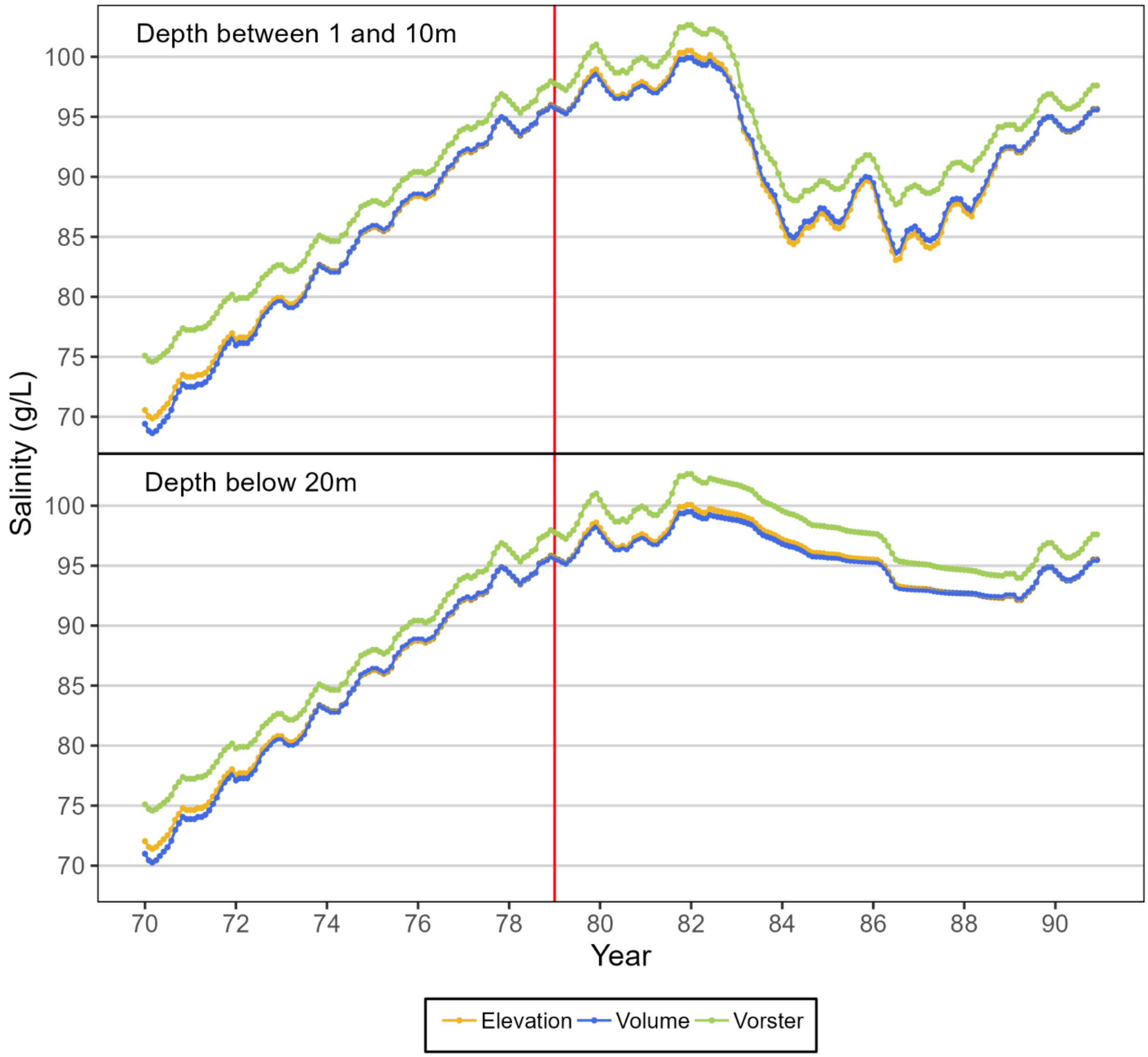

Figure 32. Salinity Estimates based on Three Models between 1970 and 1991
Red line indicates year 1979 since when Artemia population statistics are available.

Artemia Population and Salinity

Since 1979, the first year of *Artemia* seasonal statistics availability, there are moderately strong negative temporal trends for salinity in both the mixolimnion and monimolimnion, and moderately strong positive time trends for Mono Lake elevation[1] while *Artemia* seasonal mean shows moderate negative temporal trends (Figure 33). Among the five known post meromixis population peaks (1989, 2004, 2009, 2013, and 2021), there exists a strong declining trend in seasonal mean and peak as demonstrated previously ($r = -0.97$). In spite of a small sample size ($n = 5$), correlations were significant ($P = 0.0073$ and $P = 0.0037$ for seasonal mean and peak, respectively). Elevation and salinity show a moderately-strong declining trend among those five years with correlation coefficients ranging from 0.74 to 0.79 for elevation and from -0.72 to -0.85 for salinity (results not shown).

Artemia seasonal means and peaks prior to 1990 were significantly higher compared to years since 1990, and the same trends were observed for salinity and Mono Lake elevation (Table 3.17). The average of *Artemia* seasonal means prior to 1990 was 23,069 m^{-2} compared to 17,348 m^{-2} ($t = 3.68$, $P = 0.0007$) while the average of *Artemia* seasonal peaks prior to 1990 was 58,510 m^{-2} compared to 37,506 m^{-2} since 1990 ($t = 4.13$, $P = 0.0002$). The average of annual mean, maximum, and minimum salinity in mixolimnion prior 1990 was 92.1 g/L, 94.2 g/L, and 90.5 g/L, respectively, and these values were significantly higher than the respective values since 1990 by 7.1 g/L to 7.7 g/L. Mono Lake elevation shows a very similar trend to that of salinity.

Salinity in monimolimnion prior to 1990 was also significantly higher. The average annual maximum salinity prior to 1990 was 94.2 g/L and 96.4 g/L for mixolimnion and monimolimnion, respectively, and these values were higher than maximum salinity values observed in 2015 when the lowest *Artemia* seasonal mean and peak were recorded (90.1 g/L in both mixolimnion and monimolimnion). Since 1979 the highest salinity of 99.9 ± 2.9 g/L (could be 2.3 g/L higher than this) most likely occurred in 1981 when Mono Lake hit its lowest elevation, and *Artemia* seasonal population mean and peak were 32,010 m^{-2} and 101,670 m^{-2}, respectively, which were ranked third and second, respectively, among 44 years of available data. The first and largest observed post meromixis population peak occurred in 1989 when salinity in mixolimnion was estimated to range from 92.1 g/L to 95.0 g/L while salinity in monimolimnion was estimated to range from 92.2 g/L to 94.9 g/L.

Artemia seasonal population means and peaks show positive relationships with salinity averaged between 1 and 10 m in depth, especially the means and peaks during post

[1] The essentially same relationship was observed when using cumulative percentage based on the hypsometric curve; thus, the result was not presented here.

meromixis population peak years, since 1979 (Figure 34). A post meromictic population peak has occurred one or two years after meromixis breaks up regardless of salinity or lake elevation; however, magnitudes of post meromictic peaks show strong positive correlation with salinity ranging from -0.63 to -0.85, depending on *Artemia* seasonal and salinity statistics. These correlation coefficients were translated into coefficients of determination (r^2) ranging from 0.20 to 0.63. For *Artemia* seasonal means in post-meromixis peak years, up to 63% of variations was explained by annual mean salinity between 1 and 10m while 41% of variations in seasonal peaks in those years was explained by the annual mean salinity between 1 and 10m.

Prior to 1979, Mono Lake elevation declined from 1944 on, with a brief interruption between 1967 and 1970, when the lake elevation rose by 2.7 ft. Meromixis may have formed during this period; however, the preceding salinity in mixolimnion would have been ~75 g/L - lower than the initial salinity of five recorded meromixis events - and elevated freshwater input was discontinuous and lasted only briefly; thus, the chemocline would likely have been weaker and short-lived. Post-meromixis population peak after this meromixis would have been smaller compared to five known peaks if such a peak were to occur.

Correlations during non-peak years are weakly positive with salinity and significant at $\alpha = 0.1$ for *Artemia* seasonal means with all salinity measures between 1 m and 10 m. Salinity, however, explain at most 9% of variations associated with *Artemia* seasonal metrics, indicating other factors influencing *Artemia* population abundance in Mono Lake since 1979.

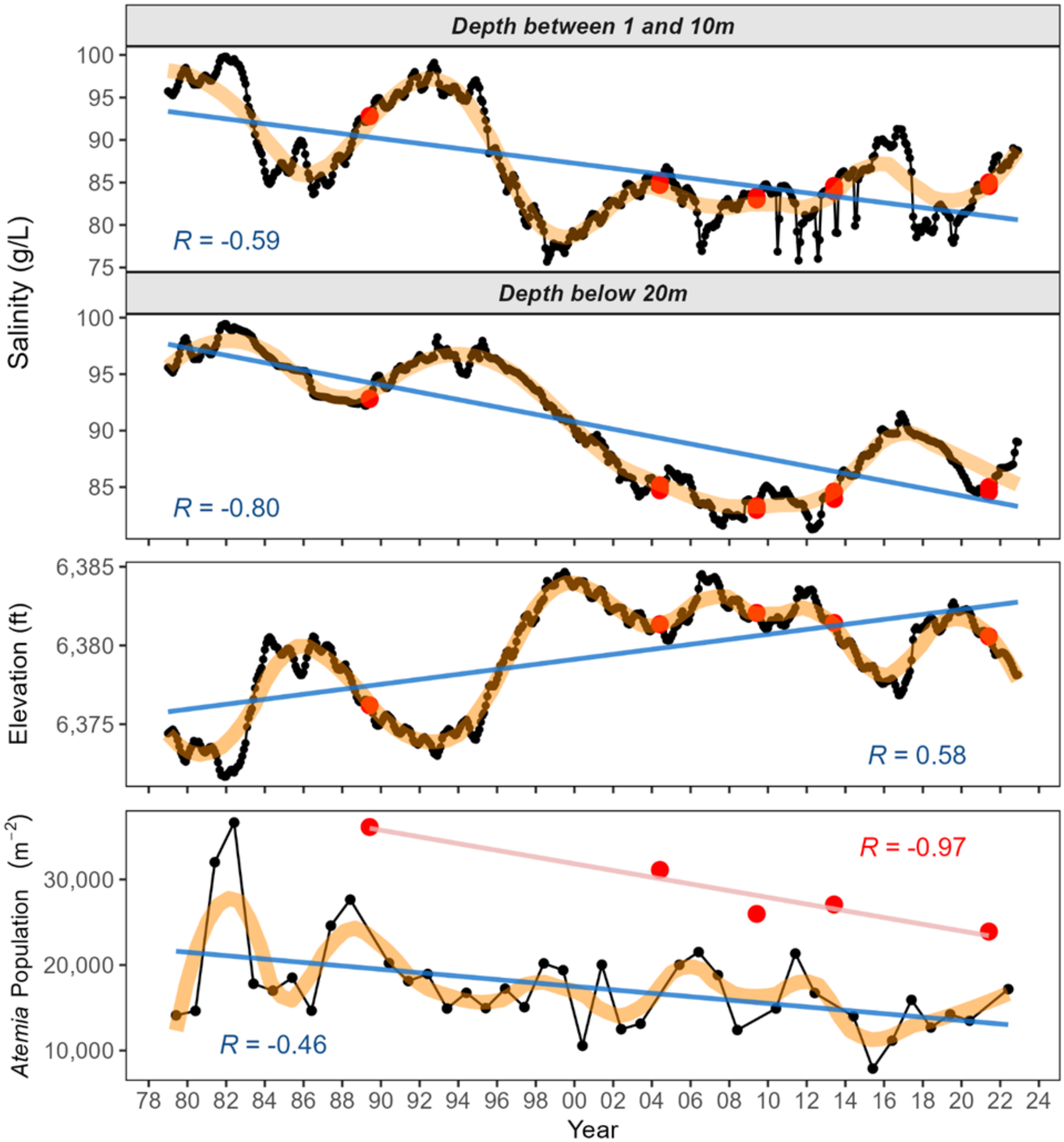

Figure 33. Salinity in Depth between 1 and 10m and Depth below 20m, Mono Lake Elevation, and Artemia Seasonal Population Mean

Red dots indicate values observed during post meromixis *Artemia* population peaks.

Table 3.17. Statistical Comparisons between before and since 1990 in Artemia Adult Seasonal Population Mean and Peak (m-2), Annual Mean, Max, and Min Mono Lake Elevation (ft), and Annual Mean, Max, Min Salinity for Depth between 1 and 10m and below 10m (g/L)

Variable	Type	Before	Since	Difference	*t*	*P*
Artemia	Mean	23,069	17,348	5,721	3.68	0.0007
	Peak	58,510	37,506	21,004	4.13	0.0002
Elevation	Mean	6,376	6,380	-4.0	-4.16	0.0004
	Max	6,377	6,381	-3.8	-3.85	0.0009
	Min	6,375	6,379	-4.2	-4.33	0.0003
Salinity between 1 and 10m	Mean	92.1	85.1	7.1	4.12	0.0005
	Max	94.2	87.1	7.1	4.37	0.0002
	Min	90.5	82.8	7.7	4.13	0.0004
Salinity below 20m	Mean	95.6	88.6	7.0	6.70	<0.0001
	Max	96.4	89.6	6.8	6.40	<0.0001
	Min	94.9	87.9	7.0	6.71	<0.0001

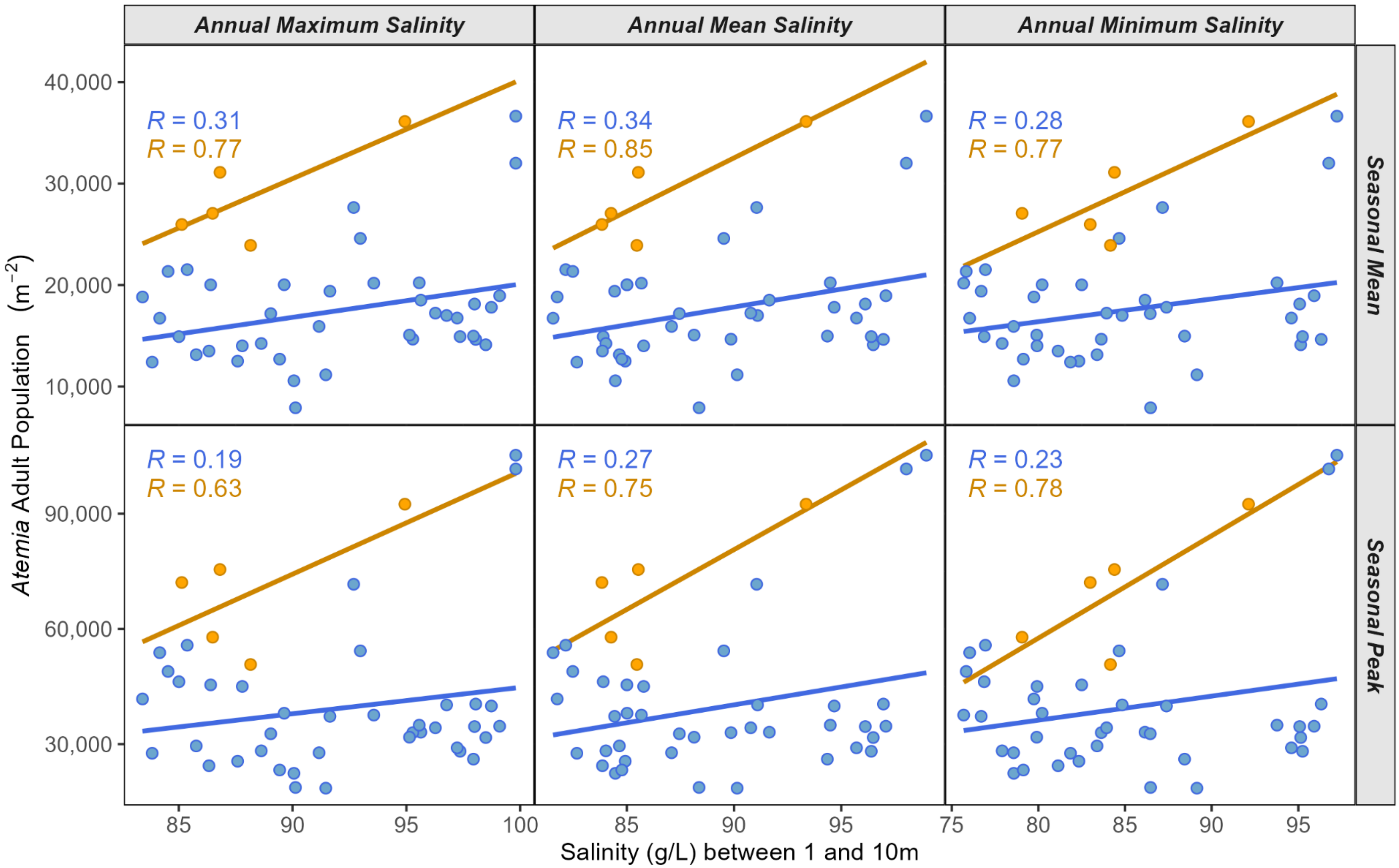

Figure 34. Scatter Plots between Artemia Population Seasonal Statistics (Mean and Peak) and Salinity between 1 and 10m (Annual Mean, Max, and Mean)

Orange dots and lines indicate post-meromictic population peak years while blue dots and lines indicate non-peak years.

3.2.4 Limnology Discussion

2022 Condition

Mono Lake elevation declined from 6379.5 ft in January to 6378.1 ft in December as runoff was 51% of average, and Mono Lake remained monomictic. The 2022 monitoring year occurred one year after the post-meromixis *Artemia* population peak in 2021. The *Artemia* seasonal mean decreased to 17,182 m^{-2} from 23,908 m^{-2} but by 28%, smaller than what has been observed during four past such occasions. The seasonal mean was not far below the long-term average of 18,779 m^{-2}, but highest since 2011, excluding two post-meromixis population peaks. The seasonal peak declined to 32,676 m^{-2} from 50.731 m^{-2} in 2021 (36% decline), and was 23% below the long-term average of 42,758 m^{-2}. The *Artemia* population centroid was on day 223 and for the sixth time in seven years the centroid exceeded day 200 after the long-term decline was reversed in 2016. The nauplii peak occurred in May, and was also the highest May nauplii abundance since 1987, contributing to a later adult peak and centroid.

Clarity of the lake remained below 1 m for the third year. The lower salinity following the large runoff in 2017 was reversed in 2021 and continued to increase in 2022. Salinity in the upper mixing layer (1 to 10 m in depth) increased from 86.5 g/L in February to 88.8 g/L in November while salinity below 20 m increased from 86.7 g/L in February to 89 g/L in November when the lake became holomictic. Epilimnetic water temperature in 2022 was above the long-term average in spring and summer due to warmer than normal monthly ambient temperatures, while hypolimnetic water remained above the long term-average between May and November. In September and October, hypolimnetic water temperatures were the highest recorded for the respective month since 1991.

Long-Term Trend

The *Artemia* population has been declining since 1979, and this declining trend is much stronger among five post-meromixis population peaks (Figure 34, Table 3.15). Magnitude of the 2021 post-meromixis peak was the lowest among five such peaks in spite of longer duration of the fifth meromixis and higher ammonium accumulation than the previous two meromictic periods as discussed in the previous compliance report (LADWP 2022). There has been a temporal shift in peak abundance of *Artemia* nauplii but a trend for *Artemia* adults is not clear, as the shift toward earlier peak reversed sometime between 2005 and 2010 (Figure 27). Food sources are not more abundant and spring water temperature has not been warmer; yet, instar nauplii are hatching earlier. There is no clear relationship between nauplii abundance and timing of adult peak or centroid (Figure 29).

The lake-wide Secchi readings in summer started to decline sometime between 2005 and 2010, coinciding with higher summer chlorophyll α concentration, which has been rising since the second meromixis (Figure 13, Figure 14). Mono Lake remained green throughout the year for the first time in 2015 during the driest five-year period on record: the lake-wide maximum Secchi reading remained below 1 m. Higher algae abundance in summer months does not appear to have translated in higher food abundance or at least, higher *Artemia* populations. Salinity is lower relative to conditions found prior to the second meromixis (1995-2002), but started to rise after the fifth meromixis ended in 2020. This rising trend in salinity, however, is expected to be short-lived, and should be reversed in 2023 due to a series of atmospheric river events beginning in late December 2022, and anticipated high runoff.

Artemia season means and peaks were higher prior to 1990 when salinity in mixolimnion between 1 and 10m is estimated to have exceeded 100 g/L and averaged around 92.1 g/L compared to 85.1 g/L since 1990 (Table 3.17). Herbst (1988) proposed a hypothetical model describing a relationship between salinity and alkali flies (*Ephydra hians*) and an idea of ecological and physiological limitations along the salinity gradient. Invertebrate species, which have adapted to higher salinity conditions, such as *Ephydra hians* and *Artemia monica*, experience lower abundance under lower salinity conditions due to more competition for food and increased predation by less salt-tolerant species. At higher salinities, these species may also show lowered abundance due to increased energy expense for osmoregulation reducing growth and reproductive rates. In between these two limitations there lies the optimum environmental conditions: salinity is too high for other specie,s but not too high for themselves. Salinity prior to water diversion by the city of Los Angeles was estimated to be around 50 g/L (Vorster 1985), and at that salinity, predatory species, such as *Trichocorixa verticalis*, may have been able to decrease *Artemia* populations, as has been observed in Great Salt Lake (Wurtsbaugh 1992). Kelts (1979) found good survivability of adult and juvenile up to 55 g/L and sudden increase in mortality at 65 g/L. Hence, *Artemia monica* populations may have been lower pre-diversion at the salinity levels near or below 50 g/L, due to these ecological limitations. During the second meromixis, rotifers (*Hexarthra jenkinae* and *Branchionus plicatilis*) were collected at the salinity around 75 g/L in the upper water column (Jellison et al. 2001). Herbst (1986) also reported increased abundance and numbers of species from 1983 to 1984 when salinity decreased from 96.6 g/L to 84.4 g/L. Zooplankton community shifts with changes in salinity among seasonally flooded dust-controlled cells were also recorded in Owens Lake (D. House, personal communication).

In the lab experiments, increasing salinity was found adversely to affect body sizes, growth rates, and brood size of *Artemia* (Dana and Lenz 1986, Dana et al. 1983), and Dana and Lenz (1986) concluded that 133 g/L was the upper limit for hatching success:

hence, survivability of the species. Based on *Artemia* seasonal statistics of past 44 years, however, four of the top five years in *Artemia* seasonal means, excluding post-meromixis population peak years, occurred before 1990, with salinity in mixolimnion ranging from 86.2 g/L to 98.9 g/L. The largest post-meromixis population peak occurred in 1989 when salinity in mixolimnion ranged from 92.1 g/L to 94.9 g/L. On the other hand, the lowest seasonal mean and peak occurred in 2015 when salinity in mixolimnion ranged between 84.5 g/L and 90.1 g/L. The year 2015 was the third year of the five-year drought, the driest five period on record. These findings may suggest that either salinity is not a primary factor or a covariate for *Artemia* population at the range of salinity since 1979. Further, there exists a slight positive trend between *Artemia* seasonal metrics and salinity among non-post meromixis population peak years, suggesting that the optimum range for *Artemia monica* in Mono Lake may fall toward the higher end (99 g/L) of the observed range of salinity. Other ecological factors, such as water temperature, food availability, and strength and duration of meromixis, also influence *Artemia* population; however, the interpretation of data, if focused solely on salinity, does not support the conclusion that salinity is the driving factor, at least at the range of salinity since 1979.

An analysis, which uses “time” as a variable, is subject to vary depending on a time period used; such that observed trends or results may become stronger or weaker or disappear altogether if *Artemia* population statistics prior to 1979 or new data were included. During the time period starting in 1979 to present, salinity and elevation also shows moderately strong correlations with monitoring years, suggesting that “time” is a covariate for analyses using these two variables. Hence, caution is required in interpreting time series trends of *Artemia* population, salinity and lake elevation or volume. It is imperative to continue to monitor *Artemia* population and salinity in order to see whether the current trend persists or not.

One notable change observed recently is sustained high abundance of phytoplankton starting in 2015. Mono Lake transparency normally improves in summer months, exceeding 10 m in the past, due to intensifying *Artemia* grazing. Starting in 2015, however, the lake transparency remained less than 1 m throughout summer month, and this phenomenon has occurred four times since then, including 2021- the latest post-meromixis *Artemia* population peak. A shift in algal community may have begun during the five-year drought and continue to progress. Algal samples should be taken periodically to determine the species composition.

3.3 Saltcedar Eradication

3.3.1 Overview of Saltcedar Eradication

Saltcedar (*Tamarix* spp.) is a fast-growing, highly prolific invasive, widely-distributed nonnative large shrub to shrubby tree that can be found in the Mono Basin. The California Invasive Plant Council (Cal-IPC) considers saltcedar as a plant with the potential to have severe impacts to ecological systems including physical processes and biological communities (Cal-IPC 2006). Saltcedar can influence native plant communities by increasing soil salinities, displacing native vegetation, or increasing fire frequency and intensity (University of California 2010).

The control of saltcedar and other invasive weeds in the Mono Basin has been a cooperative effort conducted largely by California State Parks and the Mono Lake Committee. LADWP staff have informed State Parks personnel of new noxious weed populations found while conducting fieldwork in the Mono Basin, and have undertaken tamarisk removal. Although multiple entities have contributed to weed control, these efforts have largely remained undocumented in the annual Mono Basin reports.

A recommendation put forth in the 2018 Periodic Overview Report was to improve the sharing of information between LADWP and California State Parks regarding tamarisk locations and treatment efforts so that efforts are not duplicated, and to assist in assessing the progress toward eradication efforts (LADWP 2018). In 2020, we began reporting on the Saltcedar Eradication Program.

3.3.2 Saltcedar Eradication Methodologies

Since 2016, a tamarisk surveillance and treatment program has been implemented by California State Parks, with the work conducted primarily by a contractor. In 2021, the Waterfowl Director contacted California State Parks regarding their tamarisk control program in order to provide documentation to the California State Water Resources Control Board regarding the status of tamarisk control efforts, and increase coordination between agencies. California State Parks provided a brief overview of their program, and a Calflora website link of their observations (https://www.calflora.org/entry/observ.html#srch=t&taxon=Tamarix&cols=b&inma=t&y=38.0065&x=-118.9794&z=11). Locations of all tamarisk on the Calflora website since 2016 were downloaded and displayed in ArcGIS. Tamarisk locations were associated with a shoreline location using the waterfowl survey lakeshore segment boundaries. Tamarisk treatment sites were summed by year and shoreline segment.

3.3.3 Saltcedar Eradication Results

Total tamarisk treatment sites represent the number of sites treated per year, and may include plants found previous years. Most of the tamarisk has been found in the western basin, including Mill Creek, Ranch Cove, and Rush Creek. The total number of saltcedar treatment sites was highest in 2016 (151), when Mono Lake was at its most recent low point. Since 2016, the number of sites decreased dramatically, and only five sites were treated in 2022. Of these five sites, four were new and one site was retreatment of a plant previously treated (Joe Woods, pers. comm.).

Table 3.18. Total Tamarisk treatments sites by year and shoreline segment area

	Year							Total Treated per
Shoreline Area	2016	2017	2018	2019	2020*	2021	2022	Shoreline Area
Bridgeport Creek	2		1	1				4
Lee Vining Creek	8	2	2	1				13
Mill Creek	62	7	8	6		2	2	87
Ranch Cove	30	9	6	5			1	51
Rush Creek	23	8	10	6		1		48
South Shore Lagoons	6	5	4	4				19
South Tufa	2			8		1		11
West Shore	8	4	4	5	1	1	1	24
Wilson Creek	10					1	1	12
Yearly Total Treated	151	35	35	36	1	6	5	269

*Surveys were not conducted in the southern portion of the Mono Basin due to a wildfire closure.

3.3.4 Saltcedar Eradication Discussion

The saltcedar eradication program conducted by California State Parks over the past six years has been very effective. The high number of treatment sites in 2016 occurred during a time of reduced lake level, and a high level of recruitment was observed (D. House, pers. obs.) This flush of new recruitment was effectively controlled as only 35 sites were located in 2017. Although four new plants were found in 2022, this number is small compared to previous years.

3.4 Waterfowl Population Surveys and Studies

Waterfowl population surveys are conducted to monitor the response of waterfowl populations to restoration. Although very limited historic quantitative data were available, evidence presented to the SWRCB suggested that Mono Lake was at one time a major concentration area for migratory waterfowl, and supported a much larger waterfowl population prior to out-of-basin diversions (SWRCB 1994). The SWRCB determined that diversion-induced impacts to waterfowl were more significant than for other waterbird species.

Waterfowl population monitoring in 2022 included summer ground counts at Mono Lake and fall surveys at Mono Lake, Bridgeport Reservoir, and Crowley Reservoir (Figure 85). The Mono Basin Waterfowl Director, along with assistance from LADWP Watershed Resources staff, have conducted waterfowl population monitoring annually at these three sites since 2002. Mono Lake, Bridgeport Reservoir, and Crowley Reservoir are the main areas of waterfowl concentration in Mono County, and combined, account for the overwhelming majority of waterfowl numbers in the county (D. House, pers. obs.). These data not only provide local site data, but serve as an index to regional waterfowl populations level.

3.4.1 Waterfowl Population Surveys – Survey Areas

Mono Lake

Mono Lake is almost centrally located in Mono County and lies just east of the town of Lee Vining (Figure 35). Mono Lake is a highly productive, deep-water saline lake. Invertebrate foods predominate, and Mono Lake brine shrimp (*Artemia monica*) and alkali flies (*Ephydra* spp.) are virtually the only aquatic invertebrates in the open water. Although the highly saline water, overall depth, and low diversity of food items limit habitat quality for waterfowl- nearshore and onshore resources for waterfowl include several perennial creeks and their deltas along the west shore, numerous fresh and brackish springs scattered around the perimeter, and small, temporary and semi-permanent fresh and brackish ponds. The fall migratory waterfowl community at Mono Lake is dominated by species able to exploit these invertebrate resources, and Northern Shoveler and Ruddy Duck (*Oxyura jamaicensis*), together comprise approximately 90% of all waterfowl (House and Honda 2018).

<u>Shoreline subareas and Cross-lake Transects</u>

Waterfowl spatial distribution during surveys was recorded using a combination of shoreline subareas and cross-lake transect zones (Figure 36). The entire Mono Lake shoreline was divided into 15 shoreline subareas, generally following those established by Jehl (2002). Open-water areas of Mono Lake were sampled by means of cross-lake transects. The sampling grid established in 2002 to survey open-water areas of Mono Lake consists of eight parallel transects spaced at one-minute (1/60th of a degree, approximately one nautical mile) intervals that were further divided into a total of 25 sub-segments of approximately equal length.

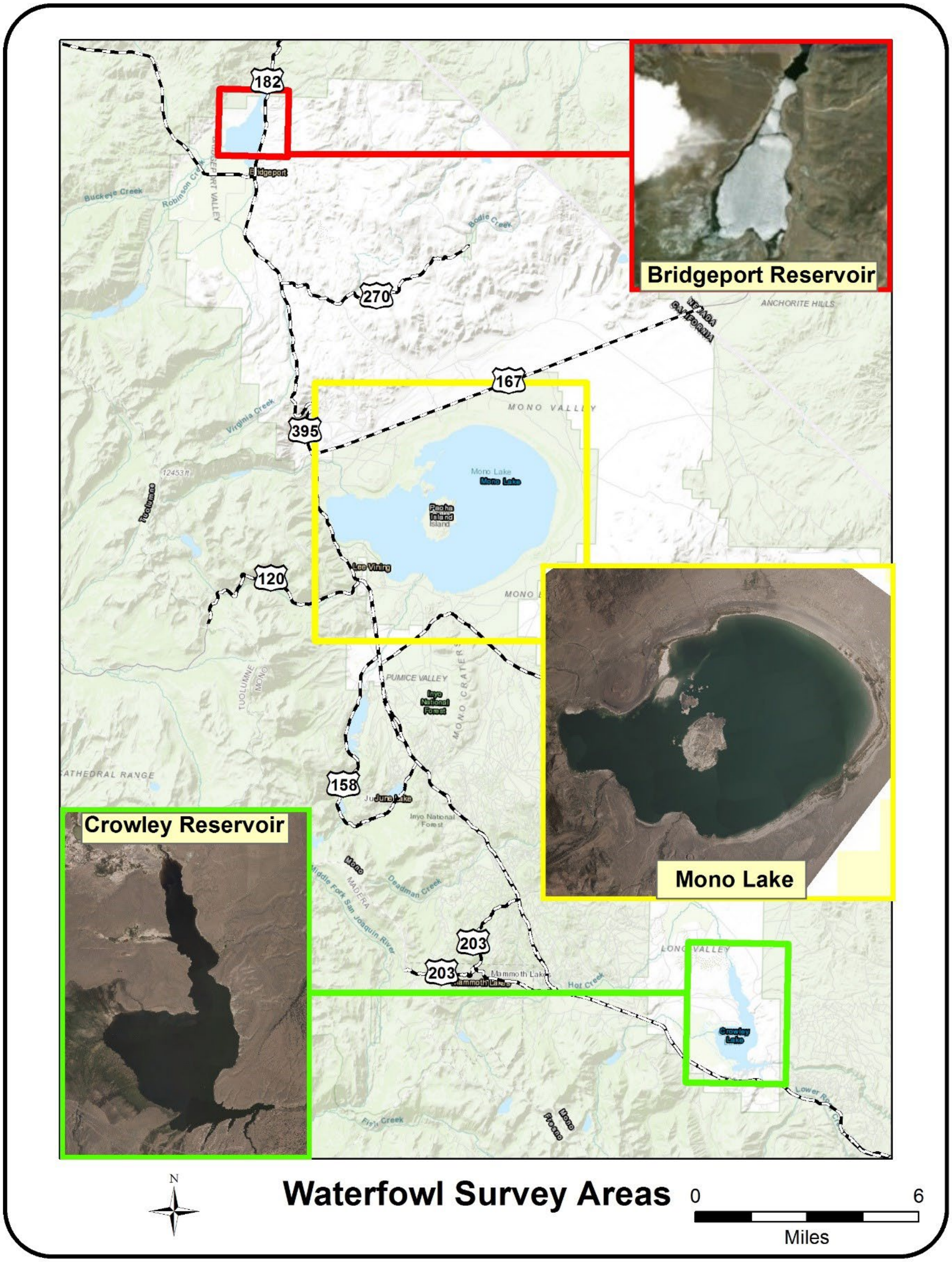

Figure 35. Waterfowl Survey Areas

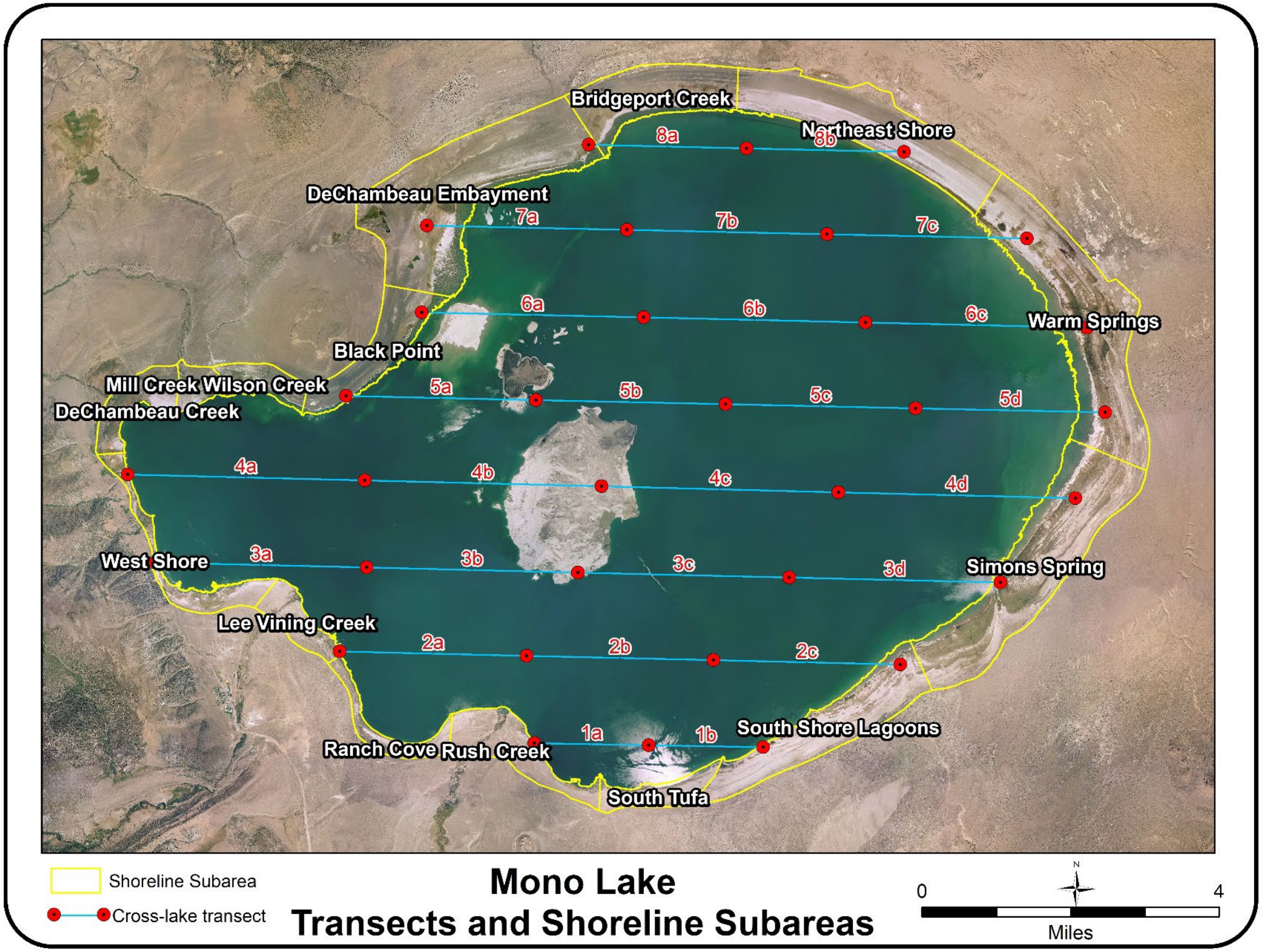

Figure 36. Mono Lake Shoreline Subareas and Cross-lake Transects

Mono Basin Restoration Ponds

The Mono Basin Restoration Ponds are located on the north side of Mono Lake, near the historic DeChambeau Ranch, and upgradient of the DeChambeau Embayment shoreline area (Figure 37). The Restoration Pond complex consists of the five DeChambeau Ponds and two County Ponds.

The DeChambeau Ponds are a complex of five artificial ponds of varying size. The DeChambeau Ponds were initially created at the onset of trans-basin diversions in the 1940s (LADWP 1996a) and restored in the mid-1990's (LADWP 2018). Project goals for the restoration included the creation of seasonal waterfowl habitat consisting of semi-permanent ponds (Ponds 1 and 2), and seasonal impoundments (Ponds 3-5), as well as adjacent seasonal wet meadow and willow habitat (LADWP 1996a, USDA Forest Service 2005). Management has seemingly differed from these original goals, as some ponds (Ponds 2 and 4) have been continuously inundated and Ponds 1 and 5 infrequently flooded. Failing infrastructure has also altered management.

There are two water sources currently supplying water to the DeChambeau Ponds (DEPO). Most of the water for the DeChambeau Ponds is from Wilson Creek and delivered via an underground pipe averaging 1-2 cfs recently (N. Carle, pers. com.). The underground piping flows water from DEPO1 to DEPO5. The second source is water from a hot artesian source adjacent to DEPO4. Hot spring water is delivered to each of the five ponds through piping. A leak developed around 2008 or 2009 in the pipe supplying the ponds (N. Carle, pers. com.), and for several years, hot spring water was only delivered to DEPO4. In 2021, repairs to the piping had restored the ability to deliver spring water to additional ponds in the DeChambeau Pond complex.

The County Pond complex consists of two ponds – County Pond East (COPOE) and County Pond West (COPOW). The two County Ponds lie in a natural basin and former lagoon that is approximately 20 acres in total area (LADWP 1996). The lagoon dried as the lake level dropped below 6,405 feet in the 1950s. The County Ponds were temporarily re-flooded on an occasional basis after that time with water diverted from Wilson Creek, until an underground pipeline was installed to deliver water from DEPO4 to the pond complex (USDA Forest Service 2005) in the late 1990s. A clay sealant was also applied to COPOE in order to reduce water use. A diverter box at the County Ponds allows some control over water releases to the individual ponds. The County Ponds have been dry the last three years.

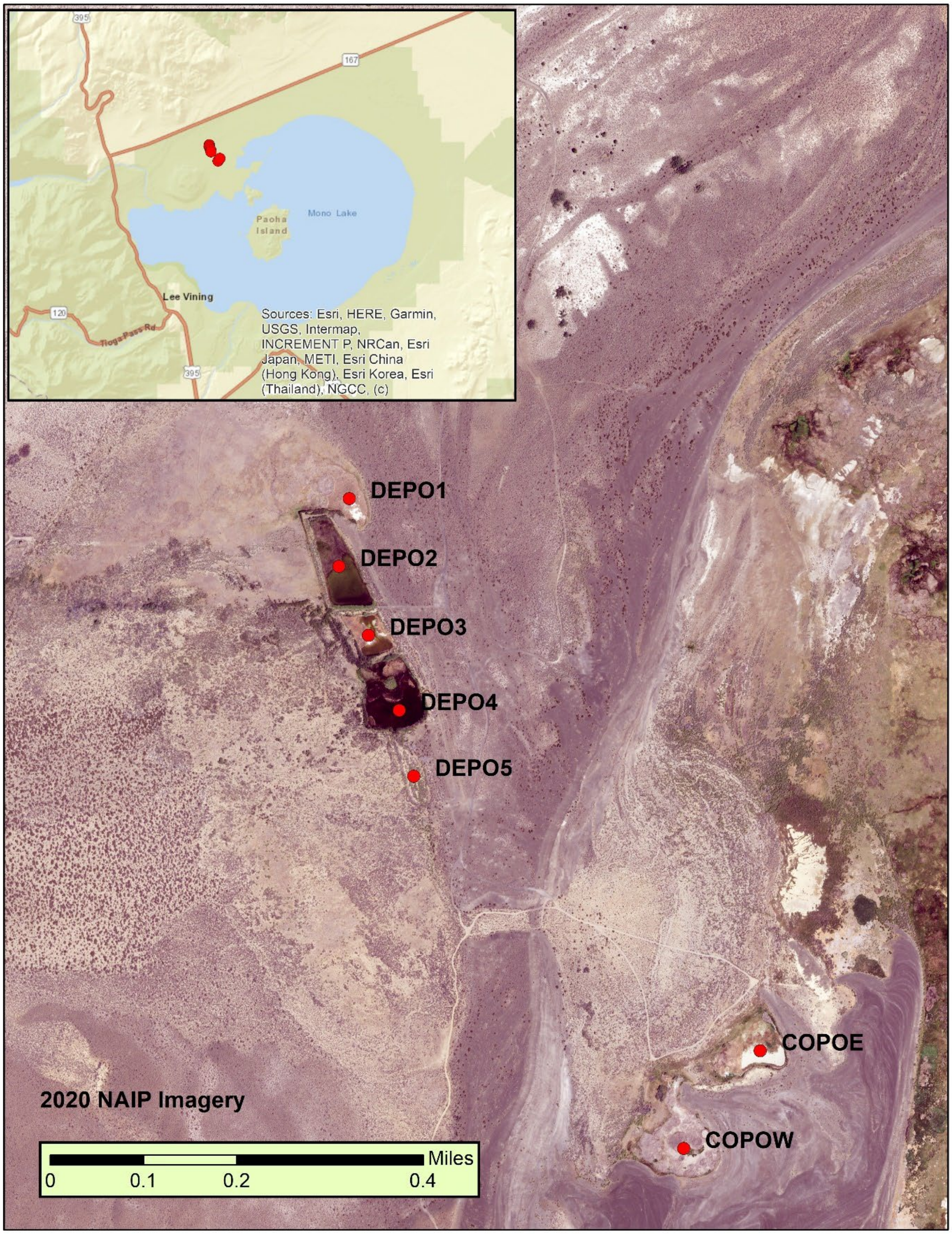

Figure 37. Mono Basin Restoration Ponds Locator Map

Bridgeport Reservoir

Bridgeport Reservoir is approximately 22 miles northwest of Mono Lake near the town of Bridgeport (Figure 35). Bridgeport Reservoir is located in Bridgeport Valley in northern Mono County, California, at an elevation of 6,460 feet. Bridgeport Valley has an arid continental climate (Zellmer 1977) and experiences relatively cool, mild summers and cold, snowy winters. The average July temperate is 63°F, and the maximum July temperature is in the low 90s. Winters are cold as the average minimum January temperature is 9.1°F, and the average maximum is 42.5°F. Precipitation averages 10 inches, mostly in the form of snow, and Bridgeport averages only 65 frost-free days a year. Bridgeport Reservoir typically freezes over in the winter for varying lengths of time. The mid-November surveys are generally ice-free, however in some years, a thin layer of ice is present in some areas of the reservoir.

Bridgeport is part of the hydrologically-closed Walker River Basin, which spans the California/Nevada border. Bridgeport Reservoir, completed in 1923, provides irrigation water to Smith and Mason Valleys in Nevada (Sharpe et al. 2007). Numerous creeks originating from the east slope of the Sierra Nevada drain toward Bridgeport Reservoir (Figure 38). These tributaries are used for upslope irrigation of Bridgeport Valley to support the primary land use of cattle grazing. The creeks directly tributary to the reservoir are the East Walker River, Robinson Creek and Buckeye Creek. Downstream of Bridgeport Reservoir Dam, the East Walker River continues flowing into Nevada, joining the West Walker River, ultimately discharging into the terminal Walker Lake, Nevada (House 2021). In Nevada, the Walker River system supports extensive agricultural operations.

Bridgeport Reservoir is a small- to moderately-sized reservoir with a surface area of approximately 7.4 square miles and a storage capacity of 42,600 acre-feet. In September 2022, Bridgeport Reservoir held 8,121 acre-feet (https://cdec.water.ca.gov/dynamicapp/QueryMonthly?s=BDP), which is approximately 28% below the long-term 2002-2021 average of 11,314 acre-feet. The September 2022 storage level was almost twice that of the September of 2021 storage level of 4,221 acre-feet.

The reservoir is rather shallow with a mean depth of 15 feet and a maximum depth of 43 feet (Horne 2003). Due to the shallow-sloping topography of the southwestern portion of the valley, reservoir level greatly influences surface area (House 2021).

Flood-irrigated pastures border the gently-sloping south and southwestern portion of Bridgeport Reservoir, while Great Basin scrub is dominant along the more steeply-sloped north arm and east shore. In shallow areas and creek deltas, submergent aquatic vegetation is abundant, including broad beds of water smartweed (*Persicaria amphibia stipulacea*). Marsh, dense wetlands, or woody riparian vegetation are lacking in the immediate vicinity of the reservoir and Bridgeport Valley proper. The reservoir is eutrophic due to high nutrient loading and experiences summer blooms of colonial forms of cyanobacteria that form dense floating scum (Horne 2003). An algal bloom in early fall of 2022 resulted in the issuance of a temporary recreational advisory due to the presence of cyanobacteria and associated toxins.

The shoreline of Bridgeport Reservoir was subdivided three shoreline survey areas (Figure 38).

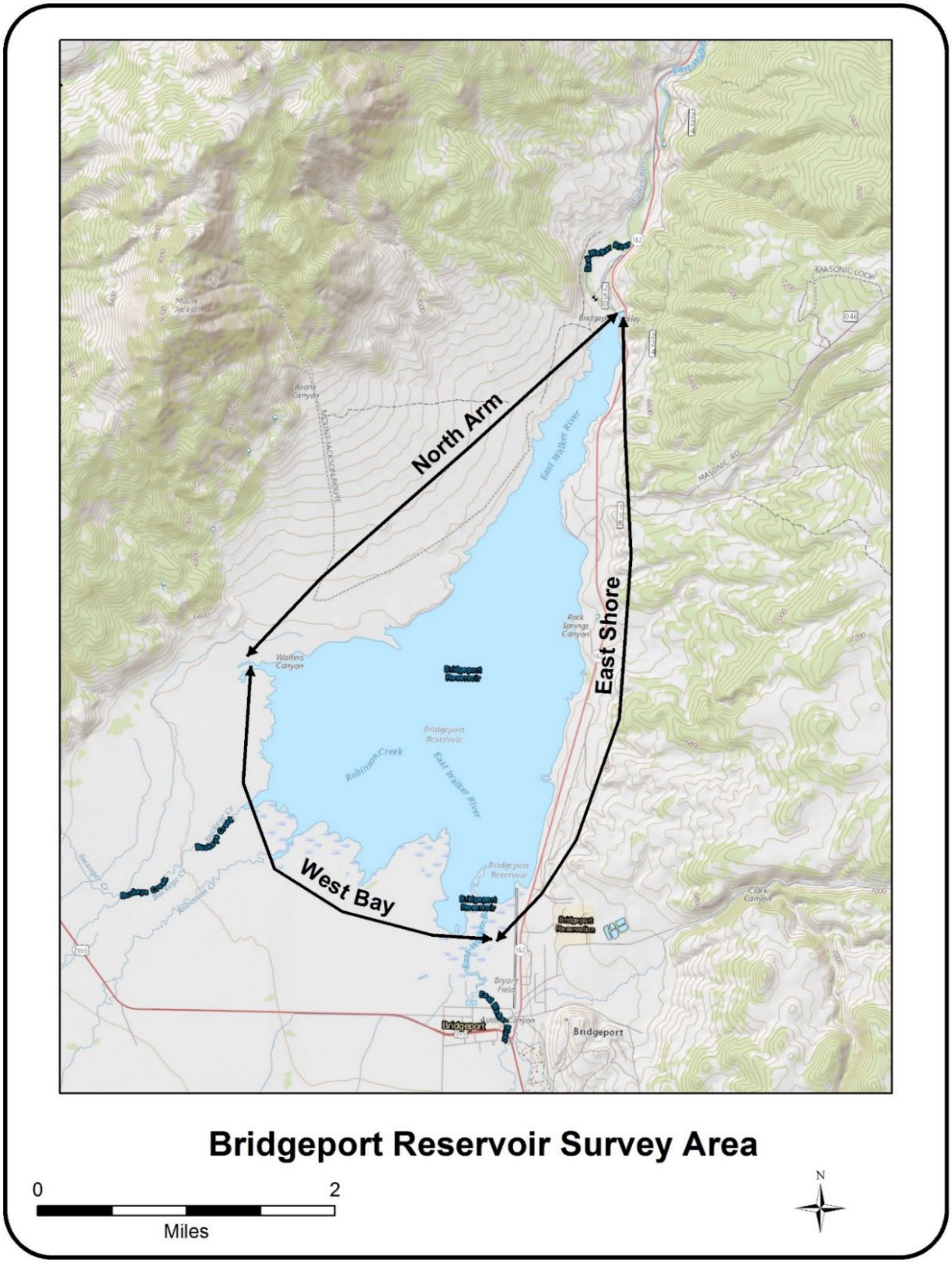

Figure 38. Bridgeport Reservoir Shoreline Subareas

Crowley Reservoir

Crowley Reservoir is approximately 31 miles southeast of Mono Lake, and 12 miles southeast of the town of Mammoth Lakes (Figure 35). Crowley Reservoir is located in Long Valley, at an elevation of 6,780 feet. Created by the construction of the Long Valley Dam in 1941, Crowley Reservoir is the second largest lake in Mono County, and the largest reservoir in the county, averaging 13.2 square miles. The primary source of fresh water input to Crowley Reservoir is the Owens River. Other fresh water input includes flows from McGee Creek, Convict Creek, Hilton Creek, and Crooked Creek. Crowley Reservoir also receives spring flow from Layton Springs along the northeast shoreline, and unnamed springs and subsurface flow along the west shore. Crowley is much deeper than Bridgeport Reservoir, with a mean depth of 35 feet and a maximum depth of 125 feet (Corvallis Environmental Research Laboratory and Environmental Monitoring Support Laboratory 1978).

Crowley Reservoir is moderately-sized with a storage capacity of 183,465 acre-feet. In September 2022, Crowley Reservoir held 102,223 acre-feet (https://cdec.water.ca.gov/dynamicapp/QueryMonthly?s=CRW), which is close to the long-term 2002-2021 average of 105,314 acre-feet. The September 2022 storage level was almost 20% greater than the September of 2021 level of 86,550.

Crowley Reservoir is eutrophic and experiences summer blooms of the nitrogen-fixing cyanobacteria *Gloeotrichia* in summer, and late-summer and fall season blooms of the cynaobacteria *Aphanizomenon* (Jellison et al. 2003). An algal bloom in early fall resulted in the issuance of a temporary recreational advisory due to the presence of cyanobacteria and associated toxins. In shallow areas near the deltas, submergent aquatic vegetation is abundant. Crowley Reservoir is known for supporting a healthy population of midges (Chironomidae).

The shoreline of Crowley Reservoir was subdivided into seven shoreline survey areas (Figure 39).

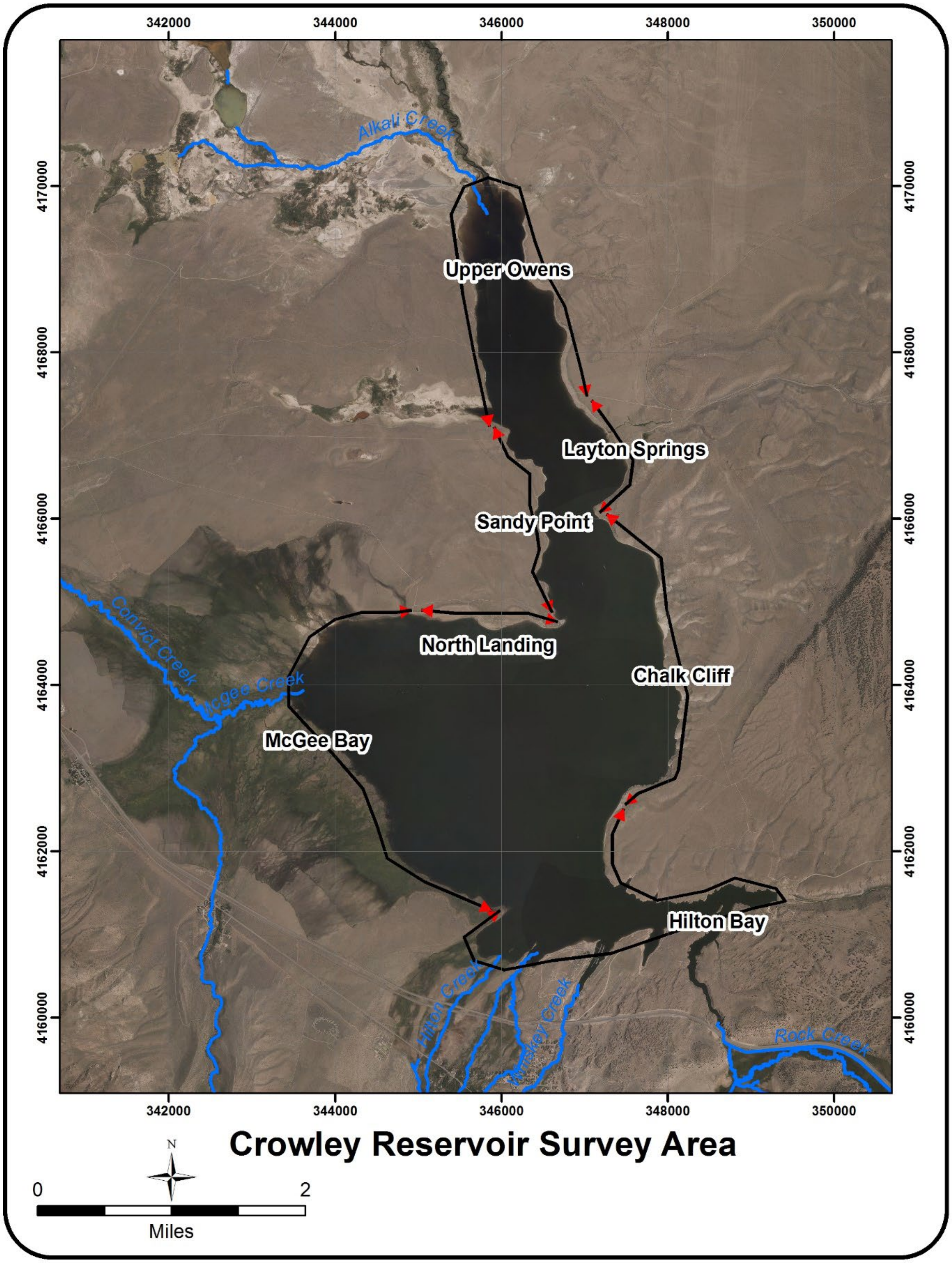

Figure 39. Crowley Reservoir Shoreline Subareas

3.4.2 Waterfowl Population Monitoring Methodologies

Mono Lake Waterfowl Surveys

Summer Surveys

Summer ground surveys were conducted in the Mono Basin along the shoreline of Mono Lake and at the DeChambeau and County Pond complexes. Nine of the 15 shoreline subareas were surveyed: South Tufa (SOTU), South Shore Lagoons (SSLA), Simons Spring (SASP), Warm Springs (WASP), Wilson Creek (WICR), Mill Creek (MICR), DeChambeau Creek Delta (DECR), lower Rush Creek and Rush Creek Delta (RUCR), and lower Lee Vining Creek and delta (LVCR).

Three summer ground-count surveys were conducted at each of these nine shoreline subareas and all seven restoration ponds in 2022. Surveys were conducted at three-week intervals beginning in early June (Table 3.19). Surveys were conducted by walking at an average rate of approximately 1 mile/hr, depending on conditions, and recording waterfowl species as they were encountered. Surveys started within one hour of sunrise, and all shoreline areas were surveyed over a 3-5-day period. The order in which subareas were visited was varied in order to minimize the effect of time-of-day on survey results. For each waterfowl observation, the following was recorded: time of the observation; the habitat type being used; and an activity code indicating how the bird, or birds were using the habitat. Examples of activities recorded include resting, foraging, flying over, nesting, brooding, sleeping, swimming, or calling.

Table 3.19. Summer ground count survey dates, 2022

	2022 Survey Number and Date		
Subarea	**Survey 1**	**Survey 2**	**Survey 3**
DECR	7-Jun	27-Jun	18-Jul
LVCR	7-Jun	27-Jun	18-Jul
MICR	7-Jun	27-Jun	18-Jul
RUCR	6-Jun	28-Jun	19-Jul
SASP	8-Jun	30-Jun	20-Jul
SOTU	7-Jun	28-Jun	19-Jul
SSLA	6-Jun	28-Jun	22-Jul
WASP	9-Jun	29-Jun	21-Jul
WICR	7-Jun	27-Jun	18-Jul
COPO	9-Jun	27-Jun	18-Jul
DEPO	9-Jun	27-Jun	18-Jul

While conducting these summer ground counts at Mono Lake, emphasis was placed on finding and recording all waterfowl broods. Because waterfowl are easily flushed, and females with broods are especially wary, the shoreline was scanned frequently well ahead of the observer in order to increase the probability of detecting broods. Information recorded for broods included species, size, GPS coordinates (UTM, NAD 83, Zone 11, CONUS), habitat use, and age. Broods were aged based on plumage and body size (Gollop and Marshall 1954).

Since summer surveys were conducted at three-week intervals, any brood assigned to Class I, using the Gollop and Marshall age classification scheme (which includes subclasses Ia, Ib, and Ic), would be a brood that had hatched since the previous visit. Assigning an age class to broods allowed for a determination of the minimum number of "unique broods" using the Mono Lake wetland and shoreline habitats.

Habitat use was recorded in order to document habitat use by waterfowl at Mono Lake. Habitat use was recorded using the mapped landtype categories. Two additional habitat types: open water near shore (within 50 meters of shore), and open water offshore (>50 meters offshore), were added to the existing classification system in order to more completely represent areas used by waterfowl.

Salinity measurements of lake-fringing ponds were taken using an Extech EC400 Conductivity/TDS/Salinity probe in order to aid in the classification of fresh versus brackish ponds when recording habitat use. Ponds with a salinity of less than 500 ppm were classified as fresh. Ponds with vegetation present and a salinity of greater than 500 ppm were classified as brackish. Ponds with a measured salinity greater than 10 ppt (the maximum range of the probe) lacking vegetation and subsurface or surface freshwater inflow were classified as hypersaline.

Fall Surveys

The fall 2022 surveys included the entire shoreline of Mono Lake, a subset of the cross-lake transects, and all seven restoration ponds. Five fall surveys were completed at two-week intervals between August 30 and November 9 (Table 3.20). Survey 2 was missed due to extended staff illness.

Helicopter-based shoreline surveys were completed by flying the perimeter of Mono Lake, maintaining a distance of approximately 500-800 feet from the shoreline. The beginning and ending points for each shoreline area were determined using both landscape features and the mobile mapping program Avenza®. Waterfowl not identifiable to species were recorded as the next identifiable taxa higher (e.g. *Aythya* spp.)

The open-water cross-lake transects were surveyed by boat using a 17-foot Boston Whaler. The areas surveyed in 2022 were: 4b, 5a, 5b, 6a, 7a, 7b, 7c, 8a, and 8b. These nine subsections of the cross-lake transects were sampled as they have been highly predictive of both total lakewide Ruddy Ducks (r^2=0.990, p<0.001) and offshore Ruddy Duck detections (r^2=0.831, p<0.001). Boat surveys of the cross-lake transects were conducted by cruising slowly at a speed of 8-10 knots along each transect subsection. The beginning and ending points for each shoreline or cross-lake transect area were determined using both landscape features and the mobile mapping program Avenza®. Slower speeds were used when waterfowl flocks were encountered, or when shallow conditions and/or the presence of submerged objects required reduced speeds for safety. On occasion, we stopped on the open water to prevent flushing, or to allow observers improved viewing of waterfowl. In some areas we could not follow the transect for the entire length due to low water depths or the presence of submerged objects including tufa or pumice blocks.

The restoration ponds were surveyed on foot, spending a minimum of 5 minutes at each pond to record any waterfowl and broods present.

In 2022 fall waterfowl surveys were conducted by the Mono Basin Waterfowl Program Director, Deborah House, and LADWP Watershed Resources Specialists, Bill Deane and Erin Nordin.

Table 3.20. Fall 2022 Mono Lake Survey Dates

Survey Period	Shoreline	Cross-lake	Restoration Ponds
Survey 1	30-Aug	31-Aug	30-Aug
Survey 2	No survey		
Survey 3	28-Sep	29-Sep	28-Sep
Survey 4	11-Oct	14-Oct	13-Oct
Survey 5	27-Oct	26-Oct	27-Oct
Survey 6	9-Nov	No survey	9-Nov

Bridgeport Reservoir Fall Waterfowl Surveys

The fall 2022 surveys included the entire shoreline and open water areas of Bridgeport Reservoir. Six fall surveys were completed at two-week intervals between August 30 and November 9 (Table 3.21). With the exception of Survey 6, which was a helicopter-based survey, all surveys were ground-based. Survey 6 was completed by helicopter due to a heavy snowfall event in early November limiting access on foot. High flows in the East Walker River combined with a higher reservoir level in 2022 prevented surveyors from crossing the river to access the shoreline west of the East Walker River. Thus, surveys were conducted with binoculars and a spotting scope from stationary viewing locations accessed from Highway 182, arriving shortly after dawn when lighting conditions were best.

Table 3.21. Fall 2022 Bridgeport Reservoir Survey Dates

Survey Period	Shoreline
Survey 1	30-Aug
Survey 2	14-Sep
Survey 3	28-Sep
Survey 4	11-Oct
Survey 5	27-Oct
Survey 6	9-Nov

Crowley Reservoir Fall Waterfowl Surveys

The fall 2022 surveys included the entire shoreline and open water areas of Crowley Reservoir. Five fall surveys were completed at two-week intervals between August 30 and November 9 (Table 3.22). All surveys were conducted by boat, except for Survey 6 which was conducted by helicopter due to a heavy snowfall event in early November limiting access by boat. Survey 2 was missed due to extended staff illness. Boat surveys work well at Crowley Lake and are a preferred method as water depth is not an issue, and because it is a time efficient way to completely survey the reservoir. Boat surveys were conducted from by paralleling the shoreline at low speeds, stopping when lighting and viewing conditions were most favorable to count waterfowl in the different shoreline areas. In 2022 however, the extreme growth of widgeongrass (*Ruppia* sp.) made boat travel very difficult until later in the season when this aquatic vegetation started to die off. Because of good visibility and access from the shoreline, boat and ground survey are comparable in terms of providing good coverage of the lake.

Table 3.22. Fall 2022 Crowley Reservoir Survey Dates

Survey Period	Shoreline
Survey 1	1-Sep
Survey 2	No survey
Survey 3	27-Sep
Survey 4	12-Oct
Survey 5	28-Oct
Survey 6	9-Nov

Aerial Photography of Waterfowl Habitats

The shoreline configuration of Mono Lake is dynamic, as seasonal and annual changes in lake level influence the development and presence of ponds, the amount of shoreline exposed, and other features important to waterfowl. Due to the dynamic nature of the Mono Lake shoreline, the aerial or satellite imagery studies and subsequent mapping performed at five-year intervals do not adequately capture annual changes that may influence waterfowl use. In order to document annual changes, aerial photographs are taken yearly in fall, in order to provide more complete information to assess shoreline changes at Mono Lake.

In 2022, digital photographs were taken from a helicopter to document shoreline conditions. Photos of all three waterfowl survey areas were taken 21 October 2022. At each waterfowl survey area, representative photos were taken of each shoreline subarea established for use in evaluating the spatial distribution of waterfowl. For reference, the elevation of Mono Lake in October 2022 was 6,378.3 feet. This work was conducted by Deborah House, Mono Basin Waterfowl Program Director.

3.4.3 Waterfowl Data Summary and Analysis

Mono Lake

Summer Waterfowl Community
The summer waterfowl community data summary includes all breeding, migrant, and non-breeding/oversummering species observed in 2022. Waterfowl species were classified as breeding or nonbreeding based on whether a territorial pair, nest, or brood has been observed over the entire length of the study. The 2022 summer waterfowl survey data were summarized by survey number.

Breeding Population Size and Composition
The size of the Mono Lake breeding waterfowl population was estimated by averaging the sum of all breeding waterfowl over the three surveys. Waterfowl totals for the Restoration Ponds will be reported separately and not included when estimating population size. The 2022 breeding waterfowl population total was compared to the long-term 2002-2021 mean. The breeding waterfowl community composition was evaluated by comparing 2022 values to the 2002-2021 mean plus standard error for each breeding species. The 2020 data were not included as only two surveys were completed in that year.

Brood totals for shoreline surveys will be used as an index of waterfowl breeding productivity. Brood number totals were determined by eliminating broods potentially double-counted over the season. Brood species, age, size and location were used to determine which broods to eliminate from the total. The calculation of brood parameters included all nesting species except Canada Goose. Canada Goose initiates nesting earlier than the other waterfowl species and family groups can be difficult to approach closely on foot except in areas where they have become habituated to humans. These factors combined with the tendency of this species to be highly mobile has made ageing broods accurately and determining the minimum number of Canada Goose broods difficult. Waterfowl brood totals were compared to the long-term 2002-2021 means +/- SE. Brood totals for the Restoration Ponds will be reported separately and not included in the shoreline counts.

The spatial distribution of breeding waterfowl was evaluated by calculating the total number of broods observed for each shoreline area in 2022. The total broods observed per shoreline subarea was compared with the long-term averages by shoreline subarea.

Habitat Use

Habitat use data were summarized for each breeding species by both modeled and mapped vegetation types (LADWP 2018).

Factors Influencing Waterfowl Breeding Populations

The influence of lake level and *Artemia* populations on the breeding population was initially evaluated using Pearson Correlation analysis. Correlation coefficients between monthly lake level, monthly *Artemia* population, monthly *Artemia* biomass and annual breeding population size annual brood counts, were generated, and scatterplots examined. When significant correlations existed, simple linear regression was used to further evaluate the relationship between variables. A t-test was use to further investigate the relationship between lake level and brood numbers.

Fall Surveys

Fall Waterfowl Population Size and Species Composition

Waterfowl species totals were summed by survey area and survey period. Survey totals were compared for each of the six surveys by site. Waterfowl community composition was described by classifying species into three groups: geese and swans, dabbling ducks, and diving ducks, and then determining the proportional abundance of each group.

Spatial distribution

The spatial distribution was evaluated by summing the total waterfowl by survey area and shoreline subarea.

Factors Influencing Mono Lake Fall Waterfowl Populations

The long-term trend in total waterfowl number at Mono Lake were evaluated for the time period 2002-2022. The influence of lake level and *Artemia* populations on the fall migratory populations was evaluated using Pearson Correlation analysis and simple linear regression. Correlation coefficients between September lake level each year, mean annual *Artemia* population, and total fall waterfowl were generated, and scatterplots examined. *Artemia* and waterfowl totals were log10 transformed to meet the test assumptions of normality.

Comparison with Reference Data

Waterfowl use of Mono Lake was compared to the reference sites by first calculating annual means +/- SE. The 2022 results were compared across the sites. The relative importance of Mono Lake within the local area was assessed by comparing the proportion of the Mono County population of fall Northern Shoveler and Ruddy Duck that use each survey area.

Aerial Photography of Waterfowl Habitats

The annual photographs of waterfowl habitats at Mono Lake, Bridgeport Reservoir and Crowley Reservoir were reviewed and compiled. Representative photos from each shoreline subarea are included in this report. The annual photos, combined with field notes taken over the summer and fall survey periods, were used to evaluate and subjectively describe shoreline conditions in 2022.

3.4.4 Waterfowl Population Survey Results

Mono Lake Summer Surveys

<u>Summer Waterfowl Community</u>

In 2022, 731 waterfowl and 10 identified waterfowl species were observed over the three summer shoreline surveys (Table 3.23) including seven breeding and three non-breeding species. Breeding waterfowl comprised the overwhelming majority of waterfowl present in summer (724 of 731). Waterfowl numbers were highest on Survey 1 and lowest on Survey 3. Of the breeding species, Gadwall was most abundant, comprising 53% of breeding waterfowl at Mono Lake in 2022.

Table 3.23. Summer Ground Count Waterfowl Detections in 2022.

Mono Lake breeding waterfowl species are in bold type.

Species	Survey 1 June 6-9	Survey 2 June 27-30	Survey 3 July 18-22	Total Detections
Canada Goose	36	82	53	171
Blue-winged Teal	1			1
Cinnamon Teal	11	2	4	17
Gadwall	154	151	81	386
Mallard	70	24	19	113
Green-winged Teal	14	11	6	31
Unidentified dabbling duck	1			1
Redhead		1		1
Lesser Scaup	4			4
Common Merganser	1			1
Ruddy Duck	3		2	5
Total waterfowl by survey	295	271	165	731

Breeding Population Size and Composition

The breeding waterfowl population at Mono Lake in 2022 is estimated to have been 241, or approximately 120 pairs. The 2022 breeding population was significantly lower as compared to the long-term mean of 308.1 +/- 18.8 SE or 154 pairs. Breeding was confirmed for Canada Goose, Cinnamon Teal, Gadwall, Green-winged Teal, and Mallard. Canada Goose numbers were slightly above average in 2022 (Figure 40), and with the exception of Green-winged Teal, numbers of all other breeding species were below their respective long-term means.

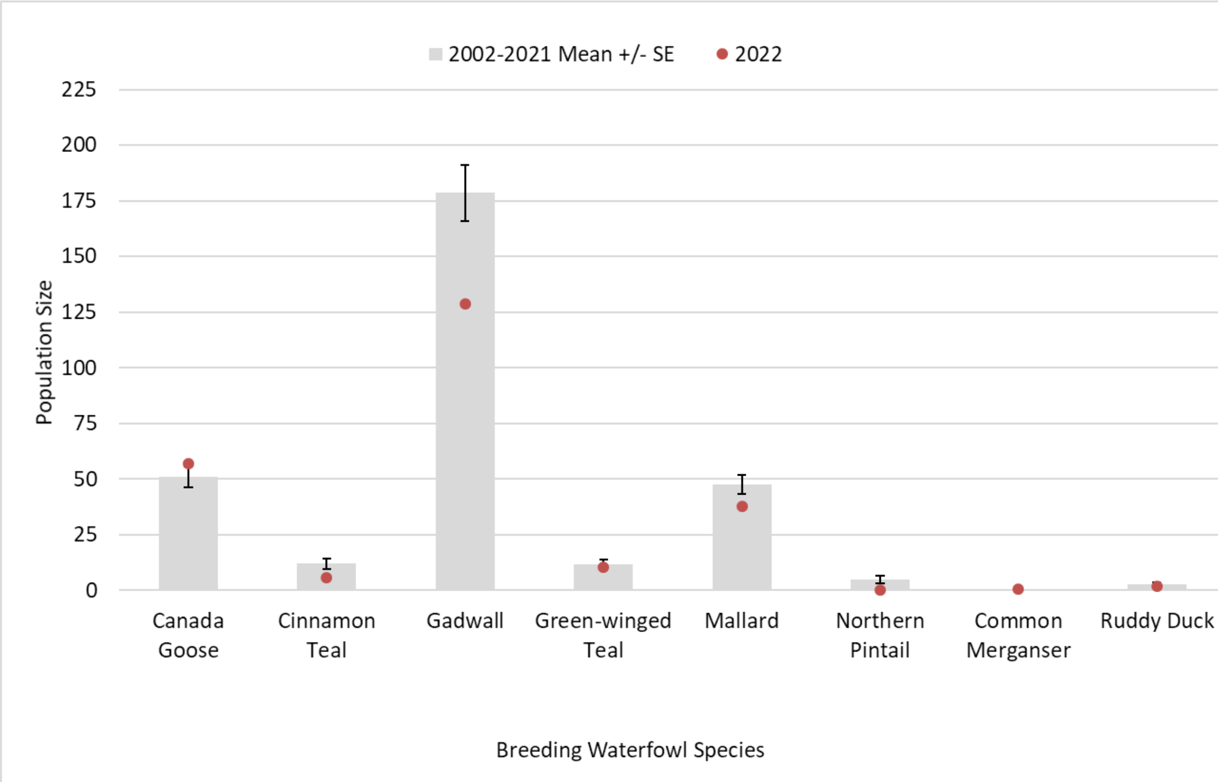

Figure 40. 2022 Breeding Waterfowl Population vs. Long-term Mean

A total of 51 waterfowl broods were found on the three surveys conducted in 2022, including seven Canada Goose and 44 dabbling duck. Breeding was confirmed for five species, with brood numbers highest for Gadwall (Table 3.24). In 2022, broods were found at all shoreline areas, except South Tufa, South Shore Lagoons, and Warm Springs. The majority of broods (18; 55%) were found along the northwest shore which includes the DeChambeau Creek, Mill Creek, and Wilson Creek shoreline areas. Other areas supporting a large proportion of the broods were Simons Spring, and Lee Vining Creek.

The total number of waterfowl broods found (exclusive of Canada Goose) in 2022 (44) was slightly below the long-term average of 48.6 +/- 3.8 of all three surveys combined. While conducting Survey 3, two additional Gadwall hens and a Green-winged Teal were acting broody, but no brood was seen. No other pairs or females without broods were observed.

Table 3.24. Waterfowl Broods by Shoreline Area, 2022

Breeding Waterfowl Species	DECR	LVCR	MICR	RUCR	SASP	SOTU	SSLA	WASP	WICR	Total 2022 Broods
Canada Goose	3	1		1	2					**7**
Cinnamon Teal									1	**1**
Gadwall	3		7	16	3				6	**35**
Green-winged Teal	2			1					1	**4**
Mallard	1	1	2							**4**
Total broods per shoreline area	**9**	**2**	**9**	**18**	**5**	**0**	**0**	**0**	**8**	**51**

<u>Habitat Use</u>

Most dabbling duck activity was concentrated in and around nearshore water features including freshwater ponds, and ria (Table 3.25). Secondarily, mudflats and alkaline wet meadow were used frequently by Mallard. The habitat use patterns of Canada Goose differed from the dabbling duck species in their greater use of mudflats and more consistent use of alkaline wet meadow and open water. Waterfowl with broods were seen most frequently in ria. Dabbling ducks fed most often in ria, and freshwater ponds. Canada Goose foraged most frequently on mudflats, alkaline wet meadow, and open water. Although classified as mudflats due to the very low cover, the areas where Canada Goose forage often support fresh young growth of wetland plants.

Table 3.25. Proportional Habitat use by Breeding Waterfowl Species, 2022

Landtypes		Breeding Waterfowl Species				
Modeled	Mapped	Canada Goose	Cinnamon Teal	Gadwall	Green-winged Teal	Mallard
Meadow Marsh		10%	0%	6%	3%	24%
	Marsh	*0%*	*0%*	*1%*	*0%*	*0%*
	Wet Meadow	*0%*	*0%*	*2%*	*3%*	*0%*
	Alkaline Wet Meadow	*10%*	*0%*	*3%*	*0%*	*24%*
	Dry Meadow/Forb	*0%*	*0%*	*0%*	*0%*	*0%*
Water		55%	88%	34%	71%	52%
	Freshwater Stream	*0%*	*0%*	*1%*	*10%*	*8%*
	Streambar	*10%*	*0%*	*5%*	*10%*	*1%*
	Freshwater Pond	*0%*	*88%*	*17%*	*48%*	*20%*
	Brackish Pond	*0%*	*0%*	*5%*	*0%*	*1%*
	Hypersaline Pond	*0%*	*0%*	*0%*	*0%*	*0%*
	Mudflat	*45%*	*0%*	*6%*	*3%*	*22%*
Ria		10%	0%	47%	23%	21%
Open Water		15%	6%	13%	0%	1%
Riparian		6%	0%	0%	0%	0%
Barren Lake Bed		4%	6%	0%	0%	1%
Upland		0%	0%	0%	3%	0%
		0%	0%	0%	0%	1%

<u>Restoration Ponds</u>

Summer Surveys

In 2022, a total of 25 waterfowl and four species were seen at the Restoration Ponds (Table 3.26). DEPO4 and DEPO3 were the most heavily used ponds. DEPO1 had water in it, but no waterfowl were observed using it. The County Ponds and DEPO5 continued to be dry. The water in DEPO1 was warm and had a heavy growth of algae all summer.

Five waterfowl broods were observed at the Restoration Ponds including three at DEPO3 and two at DEPO4 (Table 3.27). Gadwall, Mallard, and Ruddy Duck bred at the Restoration Ponds, and - as was the case for the shoreline areas - Gadwall broods were most abundant.

The number of waterfowl and broods at the Restoration Ponds in 2022 were significantly lower than the long-term 2002-2021 mean (Figure 41).

Table 3.26. Total Summer Waterfowl by Pond and Species, 2022

Species	DEPO1	DEPO2	DEPO3	DEPO4	DEPO5	COPOW	COPOE	Species Total
Cinnamon Teal			2		Dry	Dry	Dry	2
Gadwall		2	2	7				11
Mallard			1					1
Ruddy Duck			6	5				11
Pond Totals	0	2	11	12				25

Table 3.27. Waterfowl Broods at the Restoration Ponds, 2022

Species	DEPO3	DEPO4	Species Total
Gadwall	2	1	3
Mallard	1		1
Ruddy Duck		1	1
Total Broods	3	2	5

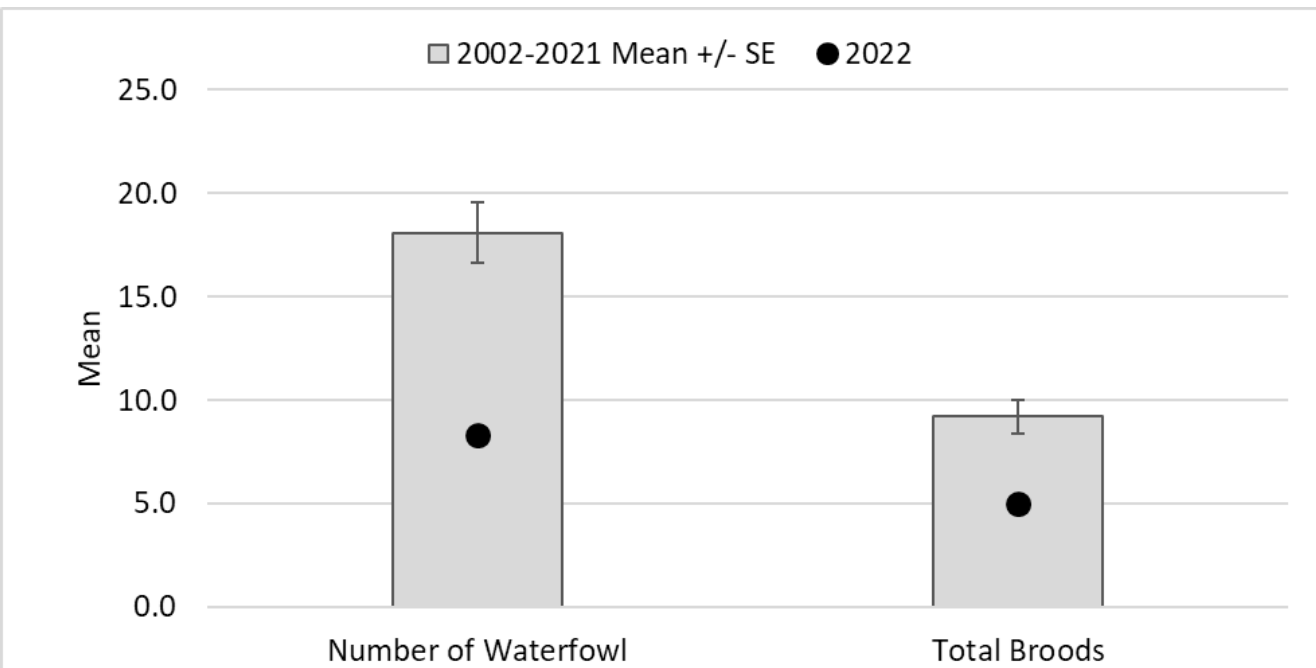

Figure 41. Mean Number of Waterfowl and Total Broods-Restoration Ponds, 2022

Factors Influencing Waterfowl Breeding Populations

The breeding waterfowl populations at Mono Lake were influenced by lake elevation. The size of the annual breeding population was most strongly correlated with lake level in April (Figure 42) (r^2_{adj} = 0.499, p=<0.01). The total number of broods was most strongly correlated with the lake elevation in June. An examination of the scatter plot showed a different response to lake level above and below a threshold of about 6,382 feet. The total number of broods produced at Mono Lake when the lake is below 6,382 feet has been significantly fewer than when the lake elevation is at or above 6,382 feet (Figure 43) (Student's t-statistic = -4.21, df=16, p<0.001).

Artemia biomass was not found to be correlated with either breeding waterfowl totals, or annual brood numbers.

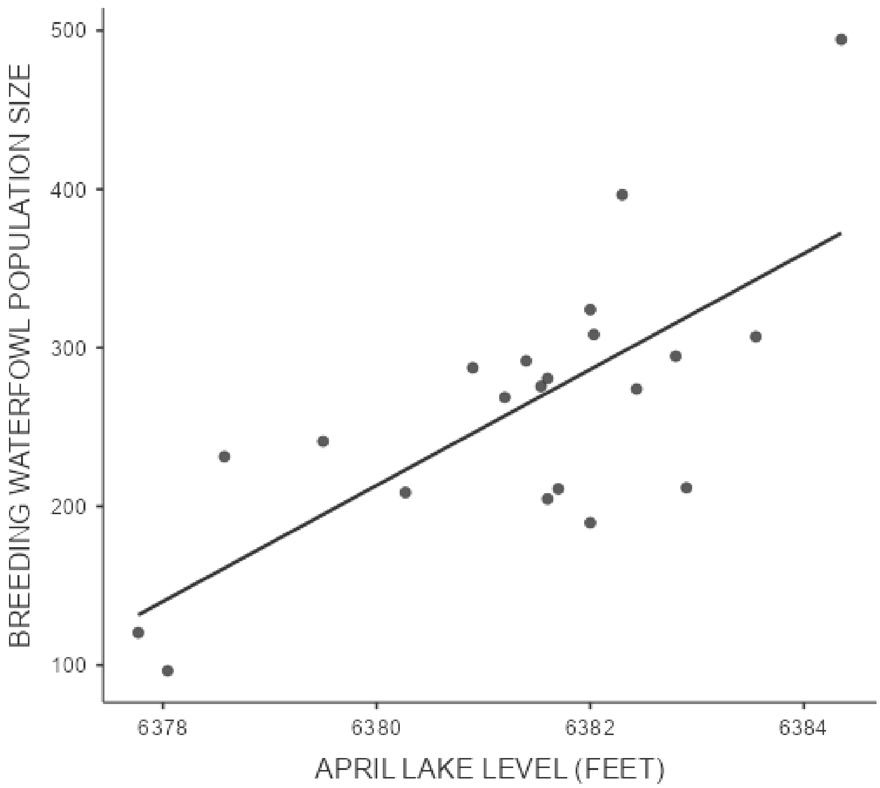

Figure 42. Annual Breeding Population Size vs. April Lake Elevation

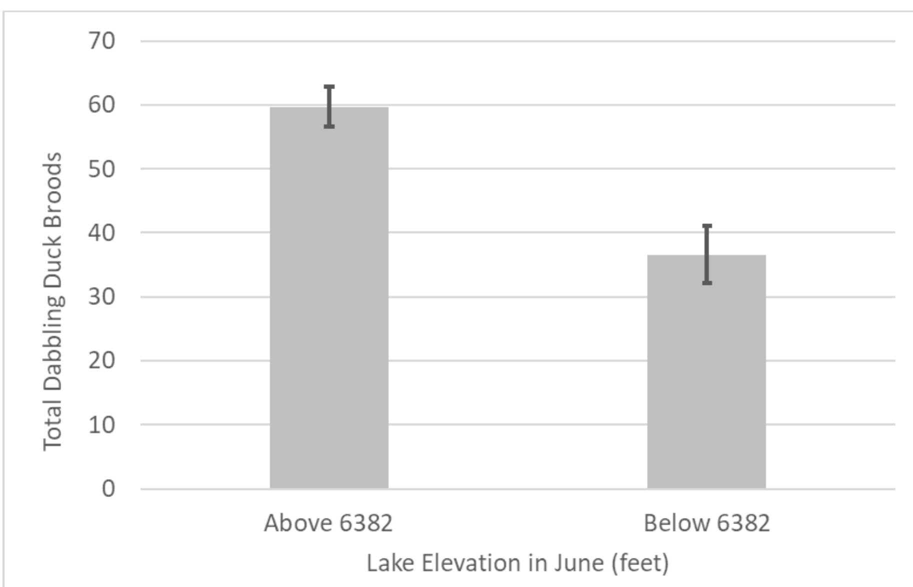

Figure 43. Total broods vs. Lake Level

Mono Lake Fall Surveys

<u>Fall Waterfowl Population Size and Species Composition</u>

A total of 13 identifiable waterfowl species and 18,351 individuals were detected during the five 2022 Mono Lake fall surveys (Table 3.28). Using interpolation to provide an estimate for the total number of waterfowl on missing Survey 2, an estimate of the total waterfowl population for 2022 is 25,032, which does not differ from the long-term mean of 25,261 +/-2691 SE. Northern Shoveler and Ruddy Duck were the most abundant species, and combined, comprised 87% of all waterfowl. Northern Shoveler have typically shown a seasonal peak in numbers on the Early- or Mid-September survey, followed by a dramatic decline through the remainder of the season (House and Honda 2018). In 2022, the highest count was on the End-of-September survey. A second pulse of Northern Shoveler arrived at Mono Lake at the end of October, such that total waterfowl numbers were higher than Early-September, due to the combined presence of Ruddy Ducks which were essentially absent in Early September. Ruddy Duck numbers typically show a seasonal peak the end of September through the end of October, and in 2022, peak numbers of Ruddy Duck were observed on the End-of-September survey, followed by a gradual decline in numbers through Mid-November. Mallard and Green-winged Teal were regularly encountered throughout fall. Gadwall were seen in small numbers, despite their presence throughout summer as the most abundant breeding species.

Table 3.28. Species Totals, 2022 Mono Lake Fall Waterfowl Surveys

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Snow Goose		No Survey			10		10
Ross's Goose					1		1
Greater White-fronted Goose					1		1
Canada Goose	16		6	25	50	20	117
Cinnamon Teal	55		6				61
Northern Shoveler	2234		3159	1040	2450	4	8887
Gadwall	32		7	4		12	55
American Wigeon					4		4
Mallard	111		65	100	4	96	376
Northern Pintail	12			18	10	70	110
Green-winged Teal	50		4	294	736	415	1499
Unidentified Dabbling Duck	2				100	4	106
Ring-necked Duck	1						1
Ruddy Duck	2		3582	2174	1257	108	7123
Totals	2515		6829	3655	4623	729	18351

The Early-September count in 2022 was below average. However, waterfowl counts were comparable to the long-term mean from the End-of-September through Mid-October. The second pulse of Northern Shoveler observed on the End-of-October count brought overall numbers to well above average. The greatly-decreased Mid-November count indicates a movement of waterfowl out of the basin.

Total waterfowl numbers have typically demonstrated a clear seasonal pattern at Mono Lake as numbers have been highest in early fall (Survey 1 through 3, Early September through the End-of-September) and lower in late fall (October to Mid-November (Figure 44). This early season peak has been largely due to the abundance of Northern Shovelers, an early season migrant in the region. In 2022, waterfowl numbers generally followed this pattern, however, the second large and late pulse of Northern Shoveler resulted in an unusually high End-of-October count (see Table 3.28). By Mid-November, following an early-season cold storm that blanketed Mono County in snow waterfowl numbers at Mono Lake dropped substantially, driven largely by departure of Northern Shovelers from the Mono Basin, and a significant reduction in the number of Ruddy Duck.

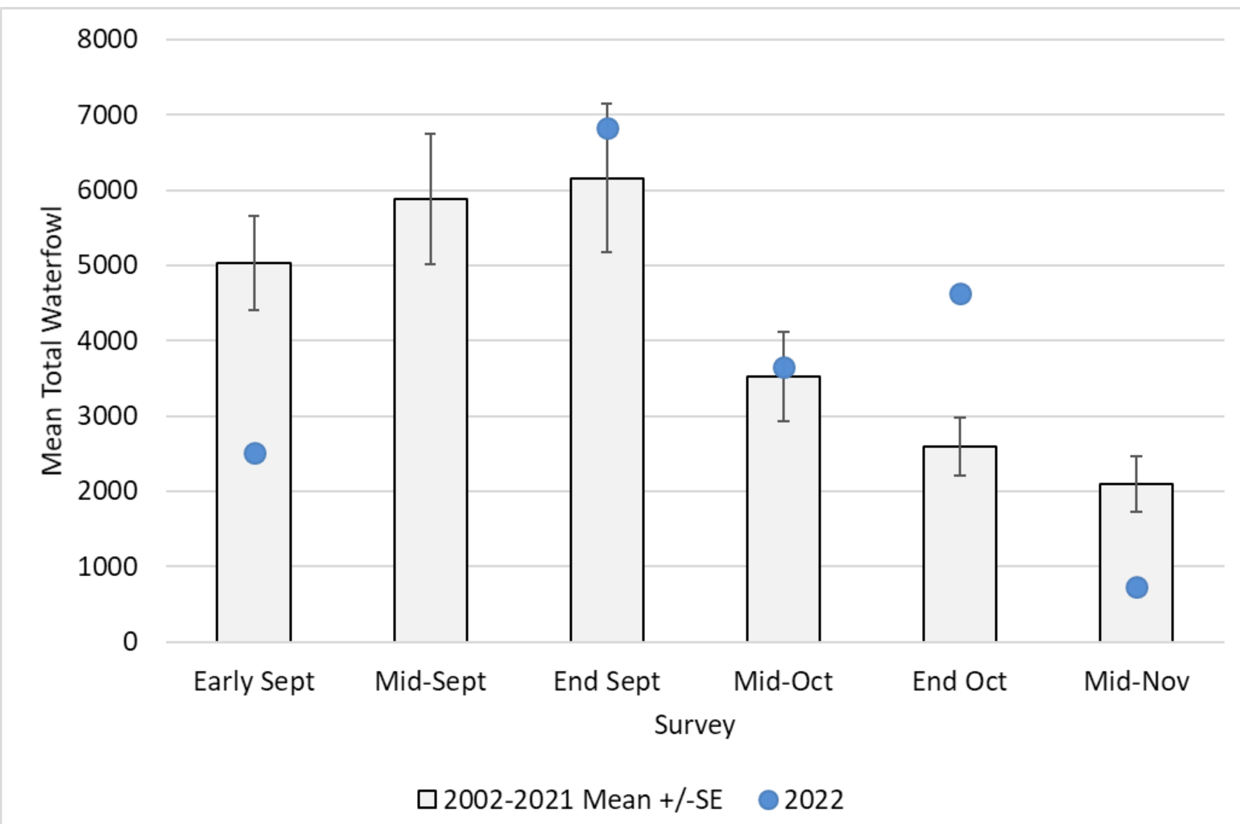

Figure 44. 2022 Mono Fall Waterfowl Survey Totals and 2002-2021 Means

Spatial Distribution

On average, approximately 78% of all fall waterfowl are recorded on the shoreline surveys, and 23% offshore on cross-lake transects. For the 2002-2022 time period, over half (~52%) of all waterfowl at Mono Lake in fall have occurred either in the Wilson Creek area of the shoreline (~30%), or seen Offshore (~23%). Wilson Creek is typically the main staging area in fall for Northern Shoveler, and frequently the only shoreline location where large numbers of this species are seen (several 1,000). Since Northern Shoveler are the dominant species at Mono Lake in fall, their frequent use of Wilson Creek results in this shoreline area supporting a large proportion of all waterfowl. Ruddy Duck, the other dominant species at Mono Lake occurs primarily off-shore, thus driving the trend of off-shore areas contributing to the other main area of use.

All other shoreline areas have attracted far fewer waterfowl, on average. The next most important areas, supporting between 5% and 8% of waterfowl have been Mill Creek, Simons Springs, DeChambeau Embayment, South Shore Lagoons, and DeChambeau Embayment.

In 2022, the main areas of use by waterfowl in fall were Wilson Creek, Offshore, Simons Spring, and DeChambeau Embayment. Almost 37% of all fall waterfowl in 2022 (6,714) were seen in Wilson Creek. The other three main areas of use in 2022 supported one-third that observed in Wilson Creek, or between 2,332 and 2,684 waterfowl. In 2022, Ruddy Ducks were most numerous in the DeChambeau Embayment area, covered by cross-lake transects 7A and 6A (Table 3.29).

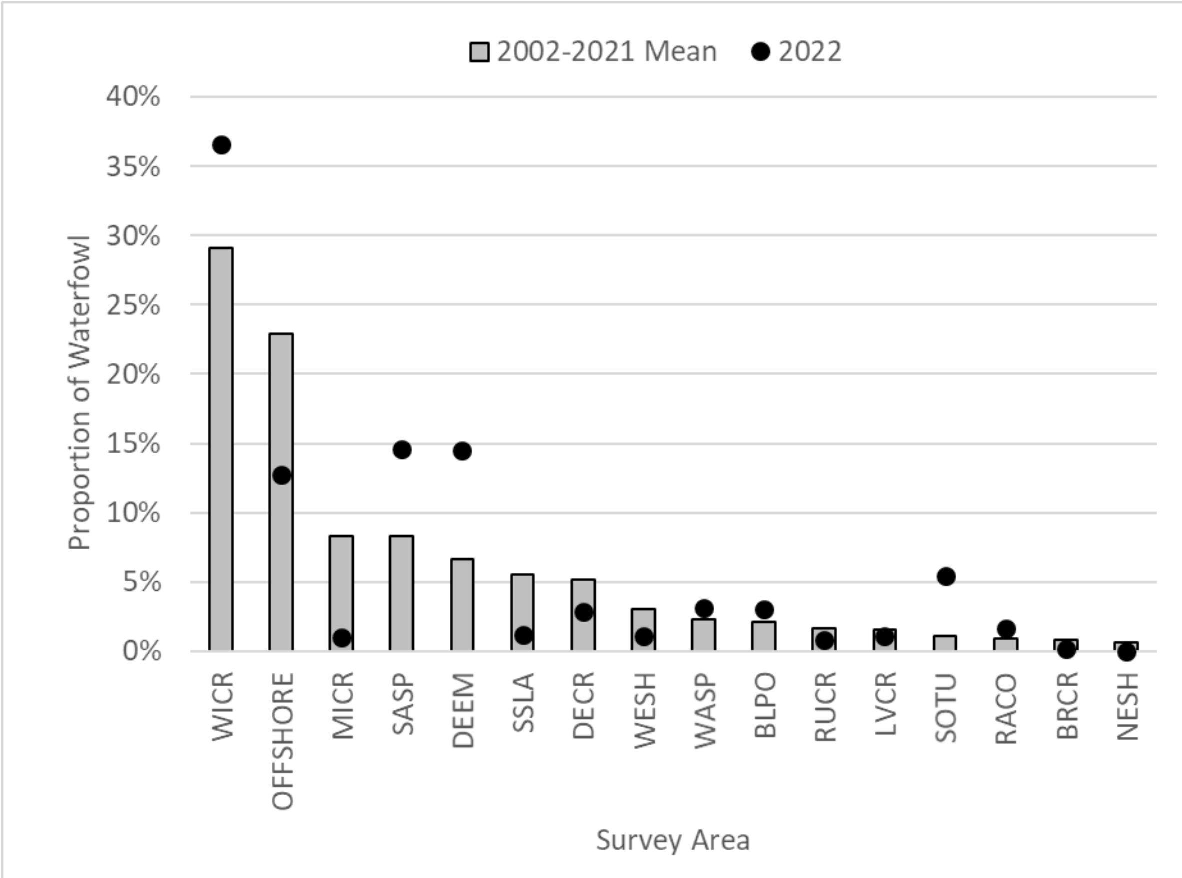

Figure 45. Fall Spatial Distribution of Waterfowl at Mono Lake

Table 3.29. Distribution of Ruddy Ducks on Cross-lake Transects, 2022

Cross-lake subsection	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Total
4B		No Survey		8	1	Transects not Surveyed	**9**
5A			33	115	106		**254**
5B				10			**10**
6A			60	147	93		**300**
7A			966	157	57		**1180**
7B			5	16	3		**24**
7C			2		1		**3**
8A			201		21		**222**
8B			8	74	147		**229**
Total Offshore	**0**		**1275**	**527**	**429**		**2231**

Restoration Ponds

Pond conditions were the same in fall as in the summer in that DEPO5, COPOW, and COPOE remained dry, and DEPO1 algae-covered. A total of 54 waterfowl of six species were seen over the six fall surveys (Table 3.30). Unlike shoreline surveys, where Northern Shovelers dominate, Gadwall were the most abundant waterfowl in fall at the Restoration Ponds. Similar to summer, DEPO4 attracted the most waterfowl. The 2022 total of 54 waterfowl over the six surveys was significantly below the 2002-2021 mean of 291.3 +/-118.0.

Table 3.30. Fall Waterfowl Totals by Pond, 2022

Species	DEPO1	DEPO2	DEPO3	DEPO4	DEPO5	COPOW	COPOE	Species Total
Cinnamon Teal		4	2	4	Dry	Dry	Dry	10
Gadwall		6		18				24
Greater White-fronted Goose				1				1
Green-winged Teal				16				16
Ring-necked Duck				1				1
Ruddy Duck			1	1				2
Pond Totals	0	10	3	41	0	0	0	54

Factors Influencing Mono Lake Fall Waterfowl Populations

The total number of waterfowl observed at Mono Lake in fall has varied from a low of 8,755 in 2018 to a high of 51,494 in 2004. There has been a weak linear downward trend in total waterfowl numbers over time (r^2= 0.2216, p=0.02, Figure 46). Unlike the breeding waterfowl population at Mono Lake, within the range observed, lake level has had no direct effect on the size of the fall migratory waterfowl population (r^2=<0.01, p=0.685); however, waterfowl numbers in years when the lake is monomictic have been higher on average than when the lake is meromictic (Figure 47). The fall migratory population has been positively correlated with mean *Artemia* population size (r=0.463) (Figure 48). Variation in annual mean *Artemia* population size has explained just 20% of the variation in total waterfowl number, however this relationship has been significant (p=0.046). The relationship between *Artemia* populations and annual Northern Shoveler numbers has been stronger (D. House, in prep), than when considering all waterfowl species as was done here, which is not unexpected given their ability to filter feed on small species such as *Artemia* (Kooloos et al. 1989).

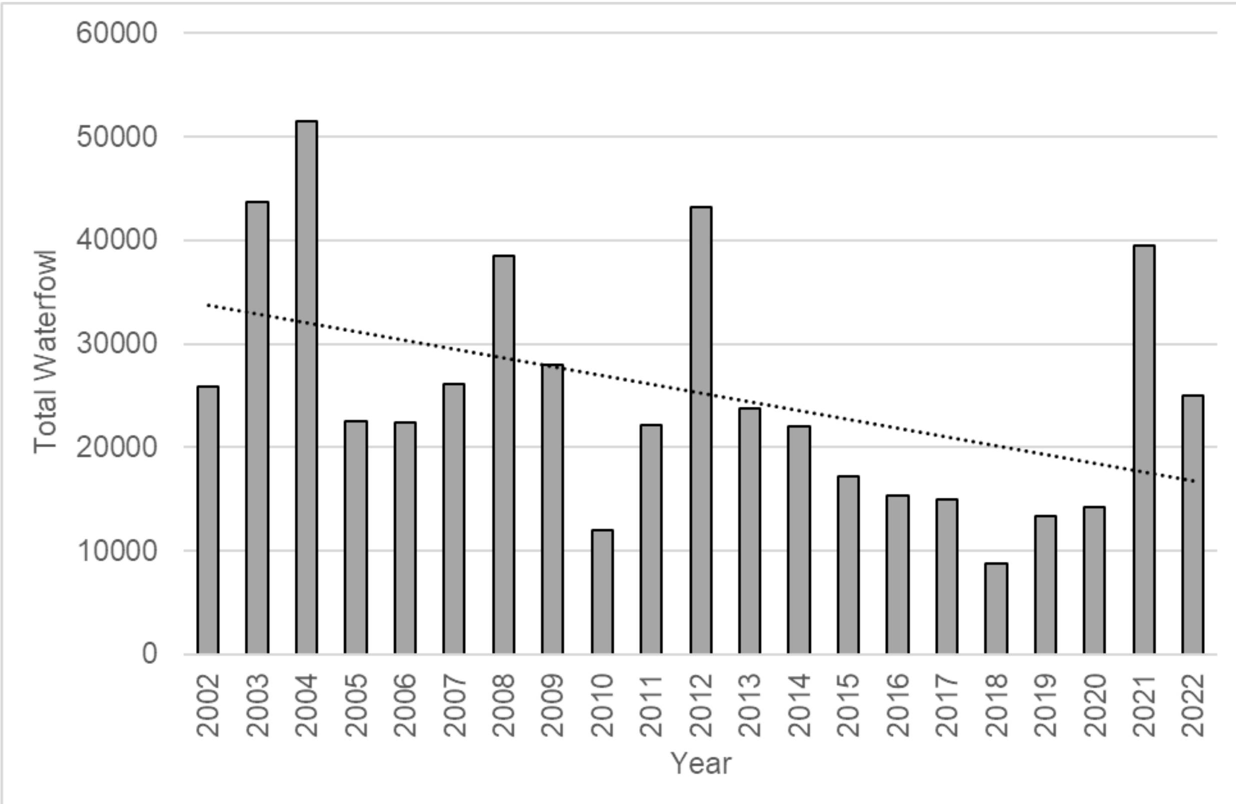

Figure 46. Trend in Total Fall Waterfowl Numbers at Mono Lake, 2002-2022

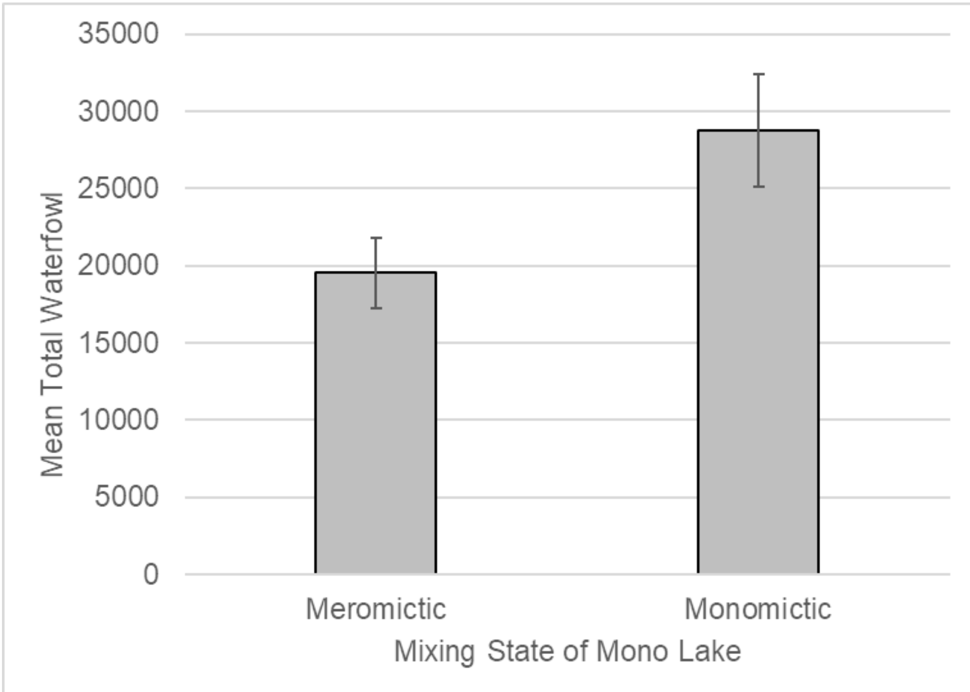

Figure 47. Mean Total Fall Waterfowl Under Meromictic vs. Monomictic Conditions

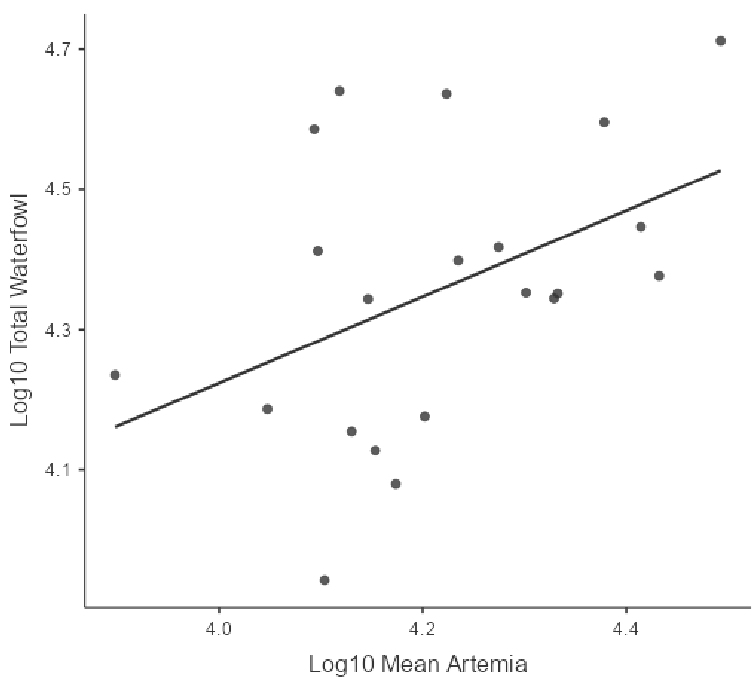

Figure 48. Total Fall Waterfowl vs. Mean Lakewide *Artemia*

Bridgeport Reservoir

Fall Waterfowl Totals and Species Composition

A total of 17 waterfowl species were identified, and 21,112 individuals recorded at Bridgeport Reservoir over the six fall surveys in 2022 (Table 3.31). Geese and swans comprised approximately 11% of all waterfowl, and of this group, only Canada Goose was abundant and present on all surveys. Dabbling ducks totaled 77% of all waterfowl, and of the seven dabbling duck species identified, Northern Shoveler and Green-winged Teal were most abundant. The seven species of divers as a whole comprised approximately 12.1% of all waterfowl.

Table 3.31. Species Totals, 2022 Bridgeport Reservoir Fall Waterfowl Surveys

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Greater White-fronted Goose				3			3
Cackling Goose						4	4
Canada Goose	454	74	570	524	487	285	2,394
Cinnamon Teal	4	9					13
Northern Shoveler	1,184	625	300	203	2		2,314
Gadwall	1,557	308	1,156	77	12	20	3,130
American Wigeon		17	30	30	48		125
Mallard	65	133	221	72	135	355	981
Northern Pintail	12	65	132	22	79	2	312
Green-winged Teal	430	214	620	583	1,362	500	3,709
Unidentified Dabbling Duck	200	5,250		20	50	204	5,724
Canvasback					7		7
Redhead	4		72				76
Ring-necked Duck			4	2	1		7
Lesser Scaup				10			10
Bufflehead		9	8	3	4	1	25
Common Merganser	36	20	68	75	17	45	261
Ruddy Duck	85	185	696	486	565		2,017
Totals	4,031	6,909	3,877	2,110	2,769	1,416	21,112

Spatial distribution

Of the three subareas at Bridgeport Reservoir, waterfowl numbers were highest in the West Bay throughout the season (Table 3.32), with the exception of the mid-September count in which more birds were seen in the East Shore area (particularly the East Walker River bay). Waterfowl were found throughout the West Bay, particularly among the deltas and inlets of Buckeye Creek and Robinson Creek. Geese were most often found out on the irrigated pastures and meadows south of the reservoir, away from the water's edge. Waterfowl use in the East Shore subarea occurred primarily in the southern half of this segment area, in proximity to inflow from the East Walker River, where shallow water feeding areas and mudflats occur. In the North Arm, waterfowl were few in number and scattered along the immediate shoreline area.

Table 3.32. Bridgeport Reservoir, Spatial Distribution by Survey, 2022

Survey	**EASH**	**NOAR**	**WEBA**
Early September	682	38	3,311
Mid-September	2,037	272	4,600
End of September	2,250	104	1,523
Mid-October	745	84	1,281
End of October	678	17	2,074
Mid-November	225	185	1,006
Total waterfowl by shoreline segment	**6,617**	**700**	**13,795**

Crowley Reservoir

Fall Waterfowl Totals and Species Composition

The number of waterfowl at Crowley Reservoir in fall of 2022 was nothing short of extreme. A total of 19 waterfowl species were identified, and 102,244 individuals recorded at Crowley Reservoir over the five fall surveys in 2022 (Table 3.33). Geese and swans comprised only 0.8% of all waterfowl and only Canada Goose was present on all surveys. Dabbling ducks totaled 70% of all waterfowl, and of the seven dabbling duck species identified, Northern Shoveler and Gadwall were far more abundant than the other species in 2022. Eight species of diving ducks were observed and divers as a whole comprised approximately 31% of all waterfowl. Ruddy Duck was overwhelmingly the most abundant of the divers.

Table 3.33. Species Totals, 2022 Crowley Reservoir Fall Waterfowl Survey

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Snow Goose		No Survey			2	34	36
Greater White-fronted Goose				21	80	60	161
Canada Goose	32		108	75	156	254	625
Tundra Swan					4	4	8
Cinnamon Teal	43		9				52
Northern Shoveler	3,900		9,513	8,192	2,140	280	24,025
Gadwall	6,653		8,642	1,653	134	830	17,912
American Wigeon	4		38	22	6	90	160
Mallard	716		866	781	687	6,150	9,200
Northern Pintail	42		1,123	352	283	490	2,290
Green-winged Teal	372		1,526	3,188	1,633	2,640	9,359
Unidentified Dabbling Duck	1,780			2,400	900	2,000	7,080
Canvasback			2		60	96	158
Redhead	2		40	130	80	4	256
Ring-necked Duck	2		4	40	32	20	98
Lesser Scaup				2	39	20	61
Surf Scoter				2	8		10
Bufflehead	8		47	18	128	142	343
Red-breasted Merganser					1		1
Ruddy Duck	482		1,816	14,929	11,788	1,394	30,409
Totals	14,036		23,734	31,805	18,161	14,508	102,244

Spatial Distribution

During the 2022 surveys, the largest waterfowl concentrations at Crowley Reservoir were in the McGee Bay area, where almost 60% of all waterfowl were tallied. The delta of the Owens River (Table 3.34), was the second most used area of Crowley Reservoir, supporting 26% of all waterfowl. Waterfowl in McGee Bay used the entire shoreline area, although higher densities were observed near the McGee Creek delta and spring outflow areas. The other area of waterfowl concentration was the Upper Owens River delta where flows from the Owens River enter the reservoir. Except at very high reservoir levels, this area has extensive mudflats for loafing, shallow feeding areas, and quiet backwater bays. During early season surveys, waterfowl generally avoid the Chalk Cliffs area as there are limited feeding opportunities due the deep water and lack of fresh water inflow. However, waterfowl continued to show a pattern of late-season use of the Chalk Cliffs area when increased numbers of dabbling ducks are seen offshore or loafing along the narrow, dry beach. Yearly, this late-season increase in use of the Chalk Cliffs area has coincided with the opening of waterfowl hunting season, and waterfowl may be seeking refuge in this area, which is difficult for hunters to access. Hilton Bay has good waterfowl habitat with adjacent meadows, some fresh water inflow, and shallow waters, but supports fewer numbers of waterfowl than areas of comparable quality, likely because of its smaller size. Waterfowl use of the Layton Spring subarea is usually concentrated near the spring inflow. Birds may also be scattered in smaller numbers along the mudflats or nearshore throughout the remainder of the subarea, which is primarily sandy beach. North Landing is another shoreline area with no direct fresh water inflow, and limited shallow water areas near shore and typically supports lower waterfowl use. The Sandy Point subarea is also an area of limited use by waterfowl due to a lack of freshwater input and limited shallow feeding areas.

Table 3.34. Crowley Reservoir, Spatial Distribution by Survey, 2022

Survey	CHCL	HIBA	LASP	MCBA	NOLA	SAPO	UPOW
Early September	-	199	86	11,938	35	15	1,763
Mid-September	No Survey						
End of September	-	688	134	16,392	348	41	6,131
Mid-October	-	956	327	21,152	94	10	9,266
End of October	147	675	1,266	9,219	308	24	6,522
Mid-November	1,352	957	2,177	3,671	1,262	1,148	3,941
Total waterfowl by shoreline segment	1,499	3,475	3,990	62,372	2,047	1,238	27,623

CHCL=Chalk Cliffs HIBA=Hilton Bay LASP= Layton Springs MCBA= McGee Bay
NOLA=North Landing SAPO=Sandy Point UPOW=Upper Owens Delta

Comparison to Reference Sites

The long-term mean annual waterfowl totals have differed among sites (Figure 49). Despite its much larger size, on average, Mono Lake supports fewer total waterfowl than either Bridgeport or Crowley Reservoirs. Crowley Reservoir has accounted for 46% of all waterfowl and waterfowl numbers have been significantly higher at Crowley Reservoir than the other two sites. Bridgeport Reservoir has supported 30% of all waterfowl, and waterfowl totals have also been significantly higher than Mono Lake. Waterfowl totals at Mono Lake have accounted for 24% of all survey areas. In 2022, waterfowl use of Bridgeport Reservoir was well below the long-term mean, and Mono Lake slightly below. The total number of waterfowl at Crowley Reservoir in 2022 was more than double the long-term mean.

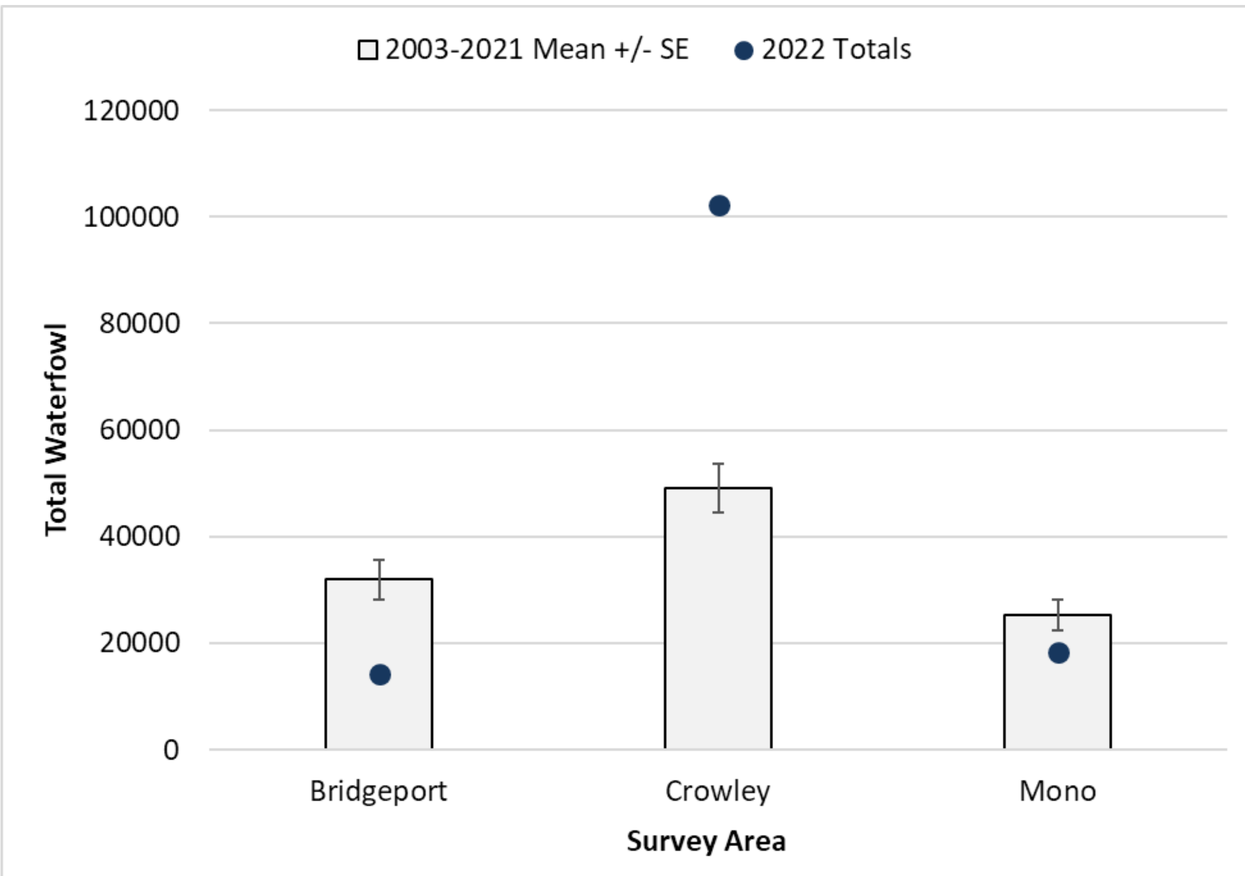

Figure 49. Comparison of Mean Fall Waterfowl at each of the Three Surveys Areas, with 2022 Totals Shown

The species composition of the waterfowl community at Mono Lake also differs notably from the other two survey areas in that it is dominated primarily by two species typically associated with saline lakes – Northern Shoveler and Ruddy Duck. In contrast, the waterfowl communities of Bridgeport and Crowley Reservoirs are more diverse, and have numerous codominant species as is typical of fresh water systems.

Although Bridgeport and Crowley support larger and more diverse waterfowl populations, Mono Lake has supported a significant proportion of the local, Mono County Northern Shoveler and Ruddy Duck fall migratory populations. Mono Lake has on average attracted the largest proportion of the Mono County Northern Shoveler population (44%) (Figure 50). In 2022, Crowley Reservoir attracted a very large number of shovelers, and a significantly larger proportion of the local population than is typical, and more than either Bridgeport Reservoir or Mono Lake. Ruddy Duck totals at Mono Lake have accounted for approximately 45% of the total for all three survey areas, roughly equal to that observed at Crowley Reservoir (
Figure 51). In 2022, Crowley Reservoir also accounted for the majority of Ruddy Ducks in Mono County, and Mono Lake supported a lower proportion of the total than is typical.

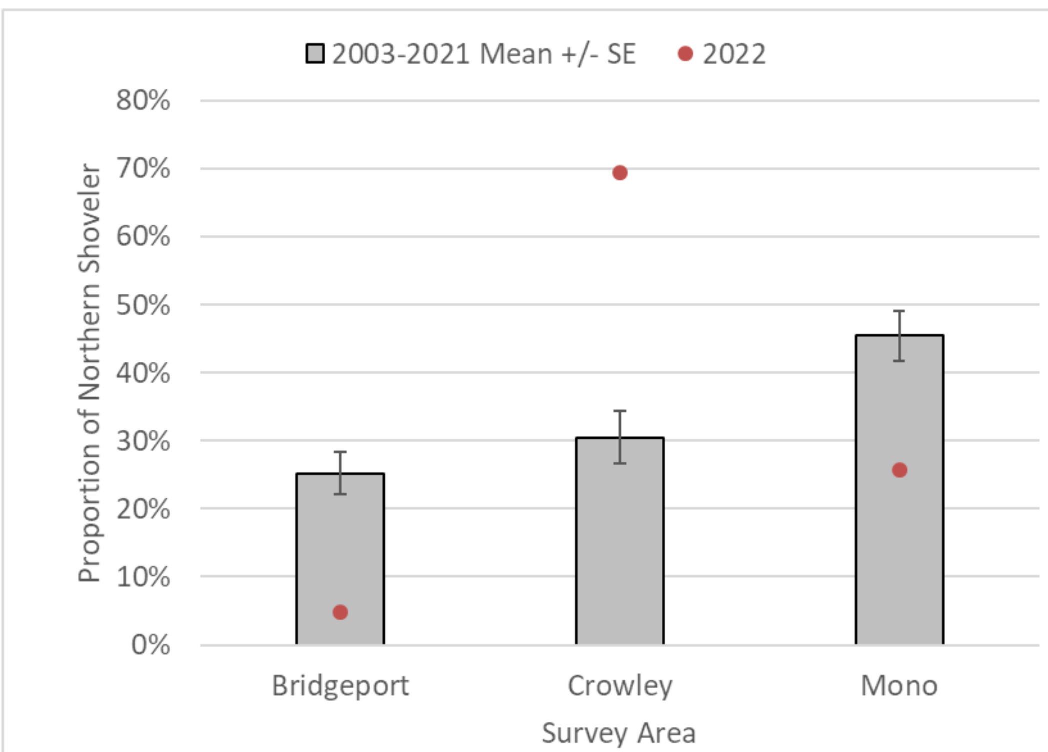

Figure 50. Proportional Abundance of Northern Shovelers by Survey Area

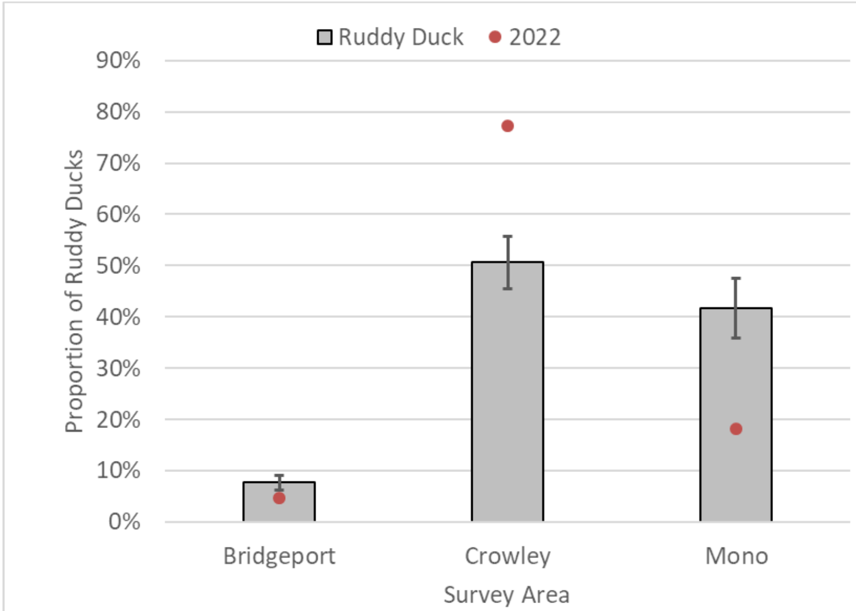

Figure 51. Proportional Abundance of Ruddy Ducks by Survey Area

Aerial Photography of Waterfowl Habitats

Please refer to Figure 36 for the location of each of the following shoreline subareas at Mono Lake.

Mono Lake Shoreline Subareas

Black Point (BLPO)

The Black Point (BLPO) shoreline area lies at the base of a volcanic hill on the northwest shore of Mono Lake. The shoreline in this area is composed of fairly dry, loose volcanic soils. At lower lake elevations, barren shoreline and alkali meadow predominate. In the western portion of BLPO, dry alkali meadow exists as a linear strip paralleling the shoreline. In the eastern portion of the shoreline area, unmapped springs exist, and alkali meadow generally extends to the shoreline creating improved foraging habitat for waterfowl. Based on a review of annual photos, brackish ponds become more numerous in the BLPO area at lake elevations above 6,382 feet, but relatively absent at lake elevations below this level. In 2022, the Black Point shoreline area was barren and dry (Figure 52, Figure 53), lacking any apparent brackish ponds.

Figure 52. Black Point Shoreline Area, Western Half

Figure 53. Black Point Shoreline Area, Eastern Half

Bridgeport Creek (BRCR)

This shoreline area is at the terminus of the Bridgeport Creek drainage, however there is currently no surface flow of water in the creek near the lakeshore. There are several springs in this area, most of which are slightly brackish and support small brackish ponds. The other wetland resources in the Bridgeport Creek shoreline area include alkaline wet meadow and small amounts of wet meadow and marsh. Waterbird use is often most concentrated at the western end of this area, where spring flow has consistently reached the shoreline at all elevations observed. At higher lake elevations, brackish ponds develop along much of the length of this shoreline area. With decreasing lake elevations, barren lakebed increases substantially without a subsequent expansion of vegetation, and brackish ponds disappear. In 2022, the eastern portion supported primarily meadow vegetation and extensive barren playa (Figure 54). The western portion of BRCR also had extensive barren playa due to the decline in lake level. There were a few locations where spring water entered the lake, such as at “Seeping Springs” (Figure 55), attracting shorebirds and small numbers of waterfowl.

Figure 54. Bridgeport Creek Shoreline Area, Eastern Portion

Figure 55. Bridgeport Creek Shoreline Area, Western Portion

"Seeping Springs" located in the center of the photo, supported a small stand of marsh and seepage of spring water to the lake.

DeChambeau Creek (DECR)

The DeChambeau Creek shoreline area is along the northwest shore of Mono Lake (see Figure 36). Flow in DeChambeau Creek is intermittent, and does not consistently reach the lakeshore. The DECR shoreline area has abundant freshwater resources, due to the presence of numerous springs that provide direct flow to the lake.

The freshwater springs at DeChambeau Creek support lush wet meadow and riparian scrub habitats. When the lake elevation is such that shoreline is exposed in this area, the extensive springflow can create freshwater mudflats. During periods of declining lake levels, wet meadow vegetation has been observed to expand onto exposed mudflats due to the abundance of freshwater spring flow. Increases in barren lake bed area with declining lake elevation have been much less apparent in the DECR area as compared to other shoreline subareas due to the slope of the shoreline and the vegetation expansion that occurs. However, some erosion and drying of the shore has occurred at the lowest lake levels. During periods of subsequent increasing lake elevations, this wet meadow vegetation, mudflats, and playa has been subsequently inundated, leaving little exposed shoreline.

Throughout the summer of 2022, the exposed playa was mudflats, but by fall, the shoreline was increasingly dry (Figure 56,
Figure 57). A small beaver dam near shore, first noted in this area in 2014, was still active. Spring flow continued to reach the lake shore in numerous places (Figure 56, Figure 57).

Figure 56. The DeChambeau Creek Area, Looking Northeast

Figure 57. The DeChambeau Creek Area, Looking North

DeChambeau Embayment (DEEM)

The DeChambeau Embayment area lies just east of the DeChambeau Ranch, and the DeChambeau and County restoration ponds (see Figure 36). Historically, Wilson Creek discharged to the lake in the DeChambeau Embayment area, although there was extensive upstream diversion for irrigation of the DeChambeau Ranch. Past diversions altered the discharge point of Wilson Creek to almost 5 miles west along the shoreline, near the Mill Creek delta.

The wetland resources in DeChambeau Embayment include alkaline wet meadow, small amounts of marsh, and several small brackish ponds. There are fresh, slightly brackish and moderately brackish springs in this area, the largest of which - Perseverance Spring (Figure 58) - is slightly brackish. Spring flow has reached the lake at all elevations observed.

The bathymetry of the shoreline and offshore area is more complex than other subareas. Very shallow sloping topography exists nearshore in the southern portion of the subarea, with a deeper bay just offshore. Pumice blocks litter the entire subarea, and are most often visible in the southern portion of this area due to the topography and shallow nearshore waters. At the higher lake elevations observed, the pumice blocks become partially to completely submerged and the shallow nearshore areas expand. As the lake level drops, this shoreline area experiences rapid increases in the acreage of barren lake bed and a land bridge forms with Gaines Island, as occurred by fall of 2022. At more extreme low lake levels, such as those observed in 2016, the geographic extent of the pumice blocks in the eastern portion of the subarea were revealed (LADWP 2018). The eastern portion of the shoreline in this subarea has a gradually sloping shoreline which extends offshore. In the eastern extent, small, isolated brackish ponds were present, and areas of spring flow to the lake shore (Figure 59).

Figure 58. DeChambeau Embayment, Perseverance Spring outflow area

Figure 59. DeChambeau Embayment, Eastern Extent

Lee Vining Creek (LVCR)

Lee Vining Creek, the second largest stream in the Mono Basin, has primarily a snowmelt-driven hydrologic regime, with peak stream flows occurring during the spring snowmelt season, and reduced flows during the remainder of the year. Peak flows typically occur in June or July in any given year, but may occur in April or May, particularly in dry years. Water diversion by LADWP began in 1941, resulting in a dry channel in the lower reaches of the creek in some years. Most of the impacts to the creek, as a result of LADWP diversions, occurred downstream of Highway 395 (SWRCB 1994). Under Decision 1631, LADWP was required to develop a stream restoration plan and undertake projects to rehabilitate Lee Vining Creek (LADWP 1996b). Channel maintenance and flushing flows, referred to as “stream restoration flows” were established in order to mimic seasonal snowmelt runoff, with the magnitude of the flow based on the hydrological conditions for the year (SWRCB 1994).

Lee Vining Creek is a woody riparian system. The lower reaches of Lee Vining Creek and its delta support small patches of wet meadow vegetation. The creek supplies abundant freshwater year-round, which remains confined to the main channel under low flow conditions, but inundates the lower floodplain under high flow conditions. At higher lake levels, the delta becomes flooded with lake water, inundating the willows and wet meadows close to shore, resulting in some dieback of willows and freshwater emergent vegetation from salt water stress. During periods of descending lake elevations, freshwater ponds may form behind littoral bars. At the most recent extreme low lake elevation observed in 2016, extensive drying of the delta meadows occurred. Ria extends offshore beyond the mapping boundary of Lee Vining Creek subarea, due to flows from Lee Vining Creek, however this waterfowl resource is not captured by landtype mapping (LADWP 2018).

Bathymetry of the area indicates limited shallow water areas near shore. Shallow sloping areas of water are limited to the delta and near the tufa grove, but depths rapidly increase offshore (LADWP 2018).

In 2022, the decline in lake level, as compared to 2021, resulted in an increase in exposed playa and drying of some delta soils (Figure 60). Only small shallow shoreline ponds existed in the northern portion of the delta. After South Tufa, the Lee Vining Creek delta is one of the busiest of the waterfowl survey sites, in terms of human activity. On two of the three summer surveys, several human and canine visitors were in the area during surveys. Waterfowl broods are frequently seen in the bay between Lee Vining Creek delta and the tufa groves to the south (Figure 61); however, little waterfowl activity was seen in the Lee Vining Creek area in 2022.

Figure 60. Lee Vining Creek Delta

Figure 61. Lee Vining Creek Delta, Western Portion

Mill Creek (MICR)

Mill Creek is Mono Lake's third largest tributary, and originates in Lundy Canyon. The Mill Creek delta is dominated by dense stands of shrub willow (Figure 62). Beaver activity in the delta since at least 2012 has resulted in fresh water ponds in amongst the willows. No springs have been identified in this area. However, freshwater often enters the lake at several points in the delta due to seepage through the loose volcanic soils. Previous bathymetry studies have indicated the creek mouth constitutes the only shallow areas in the Mill Creek delta area, and water depths increase rapidly off shore.

In 2022, beaver continued to be active, forming a series of fresh water ponds in the delta. A sheltered, brackish onshore pond received fresh water inflow from Mill Creek, yet was open to mixing with lake water due to the presence of openings in the sandbar (Figure 62). During the visit at the end of June, conditions in the delta were such that a shallow bench was present at the outflow, and many breeding waterfowl were observed foraging.

Figure 62. Mill Creek Delta

In 2022, an onshore sheltered, brackish pond was present. The beaver pond complex within the stand of shrub willows is visible on the left side of this photo.

Northeast Shore (NESH)

In the Northeast Shore area, extensive areas of barren playa dominate at most lake elevations as saline groundwater prevents the growth of vegetation. Barren playa comprises 99% of the Northeast Shore area, and only small amounts of alkali meadow are present.

At the higher lake elevations, extensive ponds have formed along the length of the shoreline segment. Although there are no known mapped springs in this reach, some are evident (D. House, pers. obs.) (Figure 63). Ephemeral ponds observed along Northeast Shore at elevated lake elevations are presumed to be brackish as flow from springs in adjacent subareas are likely contributing to creation of these ponds. Salinity of these ephemeral ponds may also be influenced by groundwater input. Historically, large perennial brackish ponds were present along the northeast shore. These historic ponds persisted in depressional areas above the high-water mark and above the target lake level for Mono Lake. In contrast to the perennial nature of these historic ponds, the ponds observed along the northeast shore in recent times have been more temporary in nature, persisting often a single season. Bathymetry studies indicates a very gradual sloped shoreline in this subarea. In 2022, the Northeast Shore area consisted primarily of dry playa, as is typical. However, the line of shallow water near shore indicates the drop in lake level has resulted in interception with the ground water (Figure 64).

Figure 63. An Unnamed Spring Along Northeast Shore

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Figure 64 Northeast Shore, Looking North

The salinity of the groundwater in this area prevents vegetative growth.

Ranch Cove (RACO)

The Ranch Cove shoreline area is a relatively small area located between Rush Creek and Lee Vining Creek. The shoreline area is narrow and generally dry, supporting primarily coyote willow (*Salix exigua*), rabbitbrush, upland scrub, and barren playa. This shoreline area has not shown significant changes with lake elevation. Waterfowl resources are limited in this area, and there is no direct spring flow evident.

Bathymetry shows essentially no shallow area in this shoreline subarea, and a steeply sloped shoreline. As is typical, in 2022 Ranch Cove showed a dry beach lacking onshore ponds or direct spring input (
Figure 65).

Figure 65. Ranch Cove Shoreline Area, Looking West

Rush Creek (RUCR)

Rush Creek (RUCR), the largest stream in the Mono Basin, has primarily a snowmelt-driven hydrologic regime with peak stream flows occurring during the spring snowmelt season, and reduced flows the remainder of the year. Peak flows typically occur in June or July in any one year, but may also occur in April or May, particularly in dry years (Beschta 1994). There is a long history dating back to the 1860s of diversion of Rush Creek flows for irrigation. Water diversion by LADWP for export began in 1941, resulting in a dry channel in the lower reaches of the creek in some years. Notable large runoff events occurring in 1967, 1969, and the early 1980s, caused substantial incision and scouring due to an absence of riparian vegetation to protect the banks and stabilize the soils. Floodplain incision then drained shallow groundwater tables and left former side channels stranded above the newly incised main stream channel (SWRCB 1994). Under Decision 1631, LADWP developed a stream restoration plan and has undertaken projects to rehabilitate Rush Creek (LADWP 1996b). Channel maintenance and flushing flows, referred to as "stream restoration flows" were established in order to mimic seasonal snowmelt runoff, with the magnitude based on the hydrological conditions for the year (SWRCB 1994).

The wetland resources available at Rush Creek are primarily meadow and woody riparian vegetation (*Salix* spp.) and the creek supplies abundant freshwater year-round. Just upstream of the delta, the floodplain is a broad meadow supporting scattered shrub willows. At higher lake levels or high creek flows, flooding has extended across the delta mouth. During periods of lake elevation recession, much channel braiding exists in the delta. From 2002 through 2014, side channels distributed water through the lower floodplain, creating saturated conditions, fresh water channels, and a stable fresh water pond along the eastern edge. In 2014, headcutting along the mainstem resulted in channel erosion, and side channel abandonment. By the following summer of 2015, pond and channels used by breeding waterfowl in the delta area disappeared as the lower floodplain experienced significant drying. Rush Creek flows create an area of ria that is expected to extend well beyond the delta.

Waterfowl habitat conditions in the delta showed signs of deterioration as compared to 2021, although increasing sinuosity along the creek from the County Road crossing downstream to the delta is resulting in more small backwater areas favorable for waterfowl foraging. Rush Creek flows were confined to the mainstem and the decreasing lake level resulted in an increased drying of deltaic soils (
Figure 66). Because the fresh water ponds in the delta were receiving less inflow from the mainstem, they were essentially dry by late summer. During late summer, there was still a low flow channel reaching the lake on the east edge of the delta (see

Figure 66). The point at which this channel was exiting had a very shallow bench and many *Artemia* and midge larvae were visible in the outflow area. The combination of low flow, shallow water, and some wind protection created excellent waterfowl foraging conditions, and in late summer, 22 waterfowl broods were seen foraging together at the outflow.

Figure 66. Rush Creek Delta

In 2022, flows were confined to the mainstem in the delta, and there was little ponding present. During summer, the small channel on the left-hand side of this photo was delivering a low flow to the shore. The combination of low flow and shallow water created excellent waterfowl foraging conditions.

Simons Spring (SASP)

The Simons Spring subarea includes the southeastern portion of the lakeshore (Figure 36). Located centrally in the subarea is the Simons Spring fault line, a conspicuous feature on the landscape. Several large springs arise from the fault, conducting groundwater to the surface (Rogers et al. 1992). Due to the shoreline gradient, small changes in lake elevation result in large changes in the degree of shoreline flooding.

The combination of high spring flow, shallow shoreline gradient, and the action of longshore currents, makes the Simons Spring shoreline area particularly dynamic, particularly west of Simons Spring fault line. Fresh water ponds are a prominent feature of the Simons Spring area due to the abundant spring flow in the area, however their presence and condition tends to be ephemeral, especially west of Simons Spring fault. Over the years, longshore currents have resulted in the development of several parallel littoral bars west of the Simons Springs fault line. These littoral bars retain upgradient spring flow and support the creation of ponds, wet meadow, and marsh behind the sandbars. During periods of increasing lake level, lake water inundates areas supporting wetland vegetation upgradient of littoral bars. The vegetation dies back due to salt stress, opening up areas previously grown over with marsh or meadow. During subsequent decreases in lake level, open fresh water ponds develop, supported by inflow from up gradient springs. If the lake level stabilizes, then wetland vegetation will recolonize the open water ponds, decreasing the amount of open water. Many of the freshwater springs in this area reach the lakeshore through breaks in littoral bars, creating extensive mudflats on exposed playa at certain lake levels. Although there may be a physical connection between the mudflats and lake water, the very shallow ponds that form on shore are typically fresh due to the high spring flow, and are colonized within 1-2 years by wetland vegetation.

Just east of the Simons Spring fault line, permanent to semi-permanent brackish water ponds are generally present year-round. The remainder of the subarea to the east lacks spring flow to the lake and supports alkali wet meadow up gradient and barren playa on shore.

Although not mapped as a landtype in this area, ria occurs wherever spring flow reaches the lake shore. The bathymetry indicates a more gradual offshore slope in the western half of the subarea, a steep offshore slope where the tufa towers of the fault line reach shore, and an increasing shallow slope to the east (LADWP 2018).

Waterfowl habitat conditions continued to be good at Simons Springs in 2022, at least in the area west of the fault line. Broad mudflats were present along much of the shoreline west of the Fault line (Figure 67). Although emergent vegetation continued to encroach on the mudflats and in upslope ponds, small open, fresh water ponds upgradient of a large berm persisted through the year. In addition, there were numerous places along the length of the shoreline where spring flow was creating small ponds and mudflats before exiting to the open water. The outflow of Goose Springs continued to exit to the lake at the extreme west end of Simons Spring shoreline area, further enhancing the resources for waterbirds (Figure 68).

The ponds east of Simons Spring Fault line persisted (
Figure 69). However, the area was heavily impacted by feral horse grazing. The emergent vegetation surrounding fresh and brackish open-water ponds had been grazed off, and there was heavy trampling throughout the wetland areas and spring heads (Figure 70). Feral horse herds were seen frequently summer through fall, primarily east of the Fault line.

Figure 67. Extreme Western Part of Simons Spring

Figure 68. Overview of the Simons Spring Area, West of the Faultline

Figure 69. Shoreline Ponds East of the Fault Line at Simons Springs

Figure 70. Heavy Feral Horse Grazing and Trampling of Gurgling Mound Spring, East of the Simons Spring Faultline

South Shore Lagoons (SSLA)

The South Shore Lagoons is a broad stretch of shoreline with scattered waterfowl habitat features. Waterfowl habitat features include permanent freshwater ponds supported by springs, seasonal to semi-permanent ponds supported by groundwater, and ephemeral brackish ponds. Like Simons Spring, the shoreline configuration in the South Shore Lagoons subarea is influenced by longshore currents.

At the western border of the subarea, a semi-permanent pond exists along a southwest-northeast trending fault line. The presence of this semi-permanent pond has been a function of lake elevation. At the higher lake elevations observed (approximately 6,383 feet), the pond has been full. Below approximately 6282.5 feet, the pond experiences notable contraction in size and, at elevations below 6,381.9 feet, has been absent.

Sandflat Spring is an isolated freshwater spring supporting two small freshwater ponds- an upper pond, and a lower pond, both partly surrounded by coyote willow. These were open water ponds until 2014, when water speedwell (*Veronica anagallis-aquatica*) and cattails (*Typha* sp.) encroached and enclosed the open water.

At the east end of the subarea is the Goose Springs complex. Goose Springs is a large spring complex that forms a series of interconnected freshwater ponds surrounded by wet meadow and marsh. In some years, the development of a littoral bar downgradient has captured spring flow, creating large onshore ponds that can be either fresh or brackish.

Away from the immediate shoreline in this subarea, the terrain is sandy hummocks with numerous small, depressions supporting alkali meadow in most years. Groundwater levels in this area have been found to be responsive to lake elevation changes (Rodgers et al. 1992) due to the high topographic gradient and very permeable soils. In 2006 and 2007 when the lake elevation was at its highest observed (above 6,385 feet), these depressions filled with groundwater, creating a series of scattered fresh water ponds in the South Shore Lagoons subarea that were quite attractive to waterfowl.

Waterfowl habitat conditions in the South Shore Lagoons area were poor in 2022. Very few ponds were present and the shoreline was dry along much of its length (Figure 71). The semi-permanent pond at the western extent of the subarea was dry (Figure 72). At Sand Flat Spring, there was no open water habitat, and no direct connection between spring flow and the open water of Mono Lake (Figure 73).

The Goose Springs area, which was excellent waterfowl habitat for many years, continued to degrade and at this point is poor and only attracting a few individuals. The outflow of Goose Springs is now fairly channelized and exiting to the Simons Spring area to the east (Figure 74). Water flow to the ponds immediately surrounding the springs appears reduced, and the ponds appear stagnant, algae-covered and reduced in size as emergent vegetation encroaches.

Figure 71. Overview of the South Shore Lagoons Area

Figure 72. South Shore Lagoons, West

The semi-permanent pond at the western extent of the subarea was dry 2022.

Figure 73. Sand Flat Spring

In 2022, there was no direct connection between spring flow from Sand Flat Spring and lake waters.

Figure 74. Goose Springs

The outflow of Goose Springs is channelized and open water ponds in the area have filled in with cattails, reducing habitat quality for waterfowl.

South Tufa (SOTU)

The South Tufa area is the primary visitor access point to the Mono Lake shoreline, notable for its large display of tufa towers. The western portion of the survey area, just east of the main tufa tower stand differs notably in terms of waterbird habitat from the eastern portion, just east of a small tufa prominence onshore between the South Tufa access point and Navy Beach. In the western portion, the shoreline is narrow, the offshore topography steep, and the brackish springs create wet mudflat conditions under most lake levels observed. East of the prominence, the shoreline is very gradually sloped onshore as well as offshore. The eastern portion supports an ephemeral brackish pond whose presence has varied as a function of lake elevation and season. At somewhat intermediate lake elevations, the shoreline pond in the eastern section has persisted from summer through fall. In periods of lower lake elevation, the brackish pond has been present in summer, but generally dried by fall.

In 2022, the western portion of this shoreline area, from Navy Beach to the tufa grove had a fairly dry beach and only small areas of mudflats (Figure 75). The eastern portion had a few small ponds early in summer, but was dry and sandy late summer to fall (Figure 76).

Figure 75. South Tufa, near Navy Beach

Figure 76. South Tufa, Eastern Extent

Warm Springs (WASP)

The Warm Springs area is located on the eastern shore of Mono Lake. The main feature of the Warm Springs area is a permanent brackish pond fed by the outflow of Pebble and Twin Warm Springs (referred to as "north pond"). These and other springs in the area support extensive wet meadow, alkali meadow, and marsh vegetation, primarily around the pond and springheads. The springs in the Warm Springs area are slightly to moderately brackish.

The north pond has been present at all lake elevations observed. Some expansion and contraction have occurred, with the pond at its largest extent in 2006. This pond is generally the most reliable place in the Warm Springs subarea to find waterfowl.

Due to the very gradual sloping shoreline in this area, small changes in lake elevation result in large differences in the amount of exposed playa on shore. Longshore action has also shaped this shoreline as evidenced by the prominent littoral bars creating the north pond and ponds downgradient. During periods of declining lake elevation, seepage of water from the north pond through the loose sandy soil results in the development of ephemeral brackish ponds downgradient of the north pond as was noted in 2010 (LADWP 2018).

Feral horse activity at Mono Lake continues to be highest in the Warm Springs area. Warm Springs was severely grazed again this year, as all of emergent vegetation along the spring channels and around the ponds had been consumed (Figure 79), and the meadows were grazed down to almost zero stubble, and bare patches of soil were appearing. Marsh-dwelling bird species formerly common at Warm Springs such as Marsh Wren, Red-winged Blackbird and Yellow-headed Blackbird are few in number now due to the loss of emergent vegetation from excessive feral horse grazing.

The intense grazing by the feral horses has had some interesting effects, at least in the short-term, on the conditions at Warm Springs, and the dynamics of waterbird use. Prior to the arrival of the horses to Mono Lake, the wetlands at Warm Springs supported extremely dense alkali meadow vegetation. The heavy grazing has removed much of the cover in the vicinity of the springs. In 2022, the area continued to be very wet, creating multiple shallow, open water ponds through areas formerly covered in dense meadow vegetation (Figure 77). Whether the observed flooding is a result of vegetation removal or changes in spring flow, is unknown. The openings and shallow flooding of the meadow has attracted waterfowl to feed and shorebirds to attempt nesting in places previously unavailable because of dense cover. Large numbers of Snowy Plover have been seen in the area, nesting and seeking cover in the hoof prints of horses. Grazing had removed all emergent vegetation surrounding the North Pond, and this pond was developing algae (Figure 78).

Figure 77. Overview of Warm Springs

The Warm Springs area was very wet in 2022.

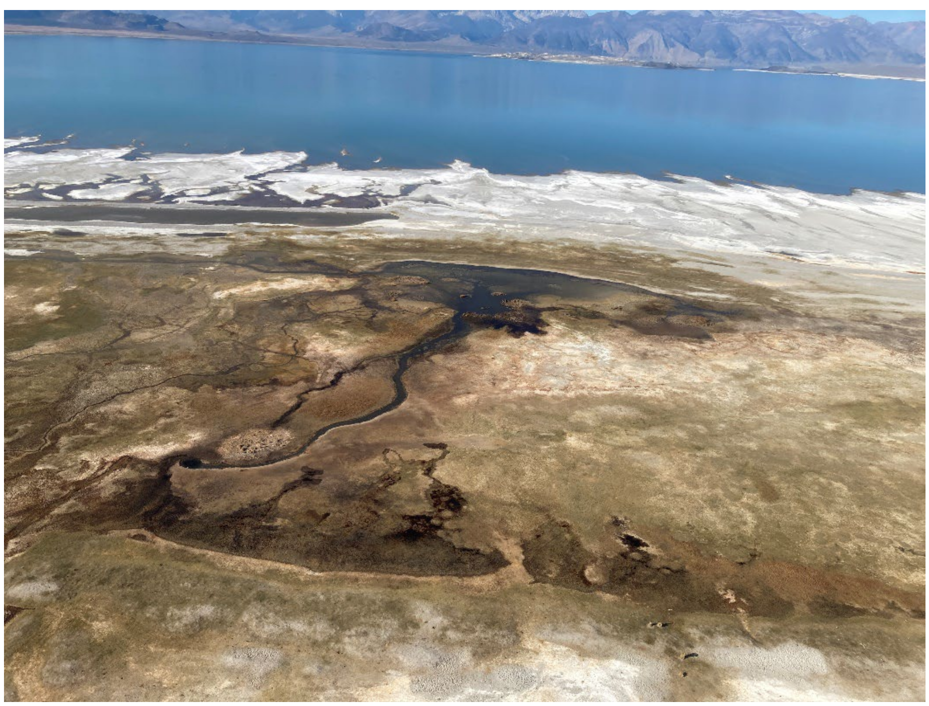

Figure 78. Warm Springs, North Pond, Looking East.

Feral horses had removed all of the emergent vegetation surrounding this pond in 2022.

Figure 79. Pebble Spring Channel at Warm Springs

Prior to feral horse use of the area, the Pebble Spring Channel, was filled with sedges. Heavy grazing has denuded the spring channels and ponds in the Warm Springs Area.

West Shore (WESH)

The majority of the West Shore subarea is located immediately east of Highway 395, along a steep fault scarp. While some shallow gradient areas exist along the southern boundary, most of this shoreline area is steeply sloping lakeward. Several fractured rock gravity springs (LADWP 1987) and two small drainages, Log Cabin Creek and Andy Thom Creek provide fresh water resources along the length of this shoreline subarea, although ponds are lacking. A very narrow beach exists along much of the length and becomes inundated at higher lake elevations. Significant changes have not been noted in the configuration of this shoreline subarea with lake elevation changes. The area supports lush wetland resources, but waterfowl use is limited (Figure 80).

Figure 80. Overview of the West Shore, Looking North/Northwest

Wilson Creek (WICR)

Wilson Creek, along the northwest shore, is one of the best and most important waterfowl habitat areas at Mono Lake. Wilson Creek supports a large expanse of wet meadow, multiple fresh water springs, and mudflats. The Wilson Creek subarea has the second highest median spring flow of the monitored springs (LADWP 2018). Due to the shoreline configuration and presence of large tufa towers, this subarea also has two protected bays. Submerged pumice blocks are present throughout the shallows of the eastern portion of the subarea. The bathymetry indicates a very gentle sloping topography throughout the protected bays and all along the shoreline (LADWP 2018). Due to the sheltered nature of the bay, the spring flow, and shallow waters near shore, the hypopycnal layer may be extensive in this area. The spring flow and shallow waters also lend toward the formation of mudflats, which have been present at most lake elevations observed. At the lowest elevation observed (2016), the retreat of shoreline resulted in some loss of the protection of the bays, however, mudflats were still prominent due to the high spring flow. The extreme low lake elevation observed in 2016 allowed an opportunity to visualize the near shore topography and the significance of spring flow to Wilson Creek bay (LADWP 2018). The topography is very gently sloping throughout the entire bay, extending out beyond the mouth of the bay and east of Tufa Mound spring. The high spring flow in this area combined with the sheltered nature of the bay is conducive to creating hypopycnal conditions. Even at higher lake elevations, such as in 2012, hypopycnal conditions would likely occur across the bay except under windy conditions, due to the high spring flow and contribution from Wilson Creek to the west in 2012. The shallow areas in the bay would make food more accessible to waterfowl. The high spring flow conditions combined with the sheltering of the bay and shallow waters support ideal feeding and loafing conditions for waterfowl at Mono Lake.

In 2022, the Wilson Creek area continued to support a fresh water pond along the west side of the bay. There was fresh water flowing into this pond early in summer, attracting waterfowl. By the end of June, however, there was no longer fresh water inflow to the pond, and little to no waterfowl activity occurred. (Figure 81). Despite the lowering lake level, the Wilson Creek area continued to support good waterfowl habitat in the form of a mix of mudflats, meadows, spring flow, and shallow water feeding areas (
Figure 82).

Figure 81. Wilson Creek Bay, as Viewed From the Southwest

The freshwater pond on the west side of the bay attracted waterfowl when receiving inflow, but little use when inflow ceased by the end of June.

Figure 82. Wilson Creek Bay, as Viewed From the North

Bridgeport Reservoir Shoreline

Please refer to Figure 38 for the location of each shoreline subareas at Bridgeport Reservoir. All three shoreline segments at Bridgeport Reservoir: North Arm, West Bay, and East Shore are shown in
Figure 83. The North Arm seen at the far end of the photo is in the narrowest part of the reservoir and includes primarily sandy beaches bordered by upland vegetation. The West Bay receives fresh water inflows from Buckeye and Robinson Creeks and the East Walker River, creating extensive mudflat areas adjacent to these creek inflow areas, especially when the water level in the reservoir is higher. The West Bay also receives extensive seepage and runoff from the adjacent irrigated pastures. The East Shore includes some mudflat and meadow areas in the vicinity of the East Walker River, but the majority of the East Shore area is bordered by Great Basin scrub or exposed reservoir bottom. The higher reservoir level in fall 2022, as compared to 2021, made the adjacent meadows and floodplain much wetter. Due to the higher flows in the East Walker River, surveyors were not able to cross the creek, and thus surveys were conducted primarily from Highway 182 or the eastern shoreline.

Figure 83. Bridgeport Reservoir, Looking Northwest

Crowley Reservoir Shoreline Subareas

Please refer to Figure 39 for the location of each of the following shoreline subareas at Crowley Reservoir. The major source of fresh water input to Crowley Reservoir is the Owens River. Other fresh water input includes flows from McGee and Convict Creeks, Layton Springs, and subsurface flow from other springs along the west shore. Vegetation communities immediately surrounding Crowley Reservoir include irrigated pasture, wet meadow, Great Basin scrub, alkali meadow, and mudflats.

Chalk Cliffs (CHCL)

The Chalk Cliffs subarea lacks fresh water inflow areas and wetland habitats, and is dominated by sandy beaches adjacent to steep, sagebrush-covered slopes (Figure 84).

Figure 84. Chalk Cliffs

Hilton Bay (HIBA)

Hilton Bay includes Big Hilton Bay to the north and Little Hilton Bay to the south (Figure 85). The Hilton Bay area, surrounded by meadow and sagebrush habitat, receives small amounts of fresh water input from Hilton Creek, Whiskey Creek, and area springs.

Figure 85. Hilton Bay

Layton Springs (LASP)

The Layton Springs shoreline area is bordered by upland vegetation and a sandy beach. Layton Springs provides fresh water input at the southern border of this lakeshore segment (Figure 86).

Figure 86. Layton Springs

McGee Bay (MCBA)

The McGee Bay shoreline area supports mudflat areas immediately adjacent to wet meadow habitats (Figure 87). McGee Creek and Convict Creek are tributaries to Crowley Reservoir in this shoreline area. Vast mudflats and wetlands occur along the west shore of Crowley Reservoir, as this area receives inflow from springs and subsurface flow from up-gradient irrigation. The slightly higher reservoir level in fall of 2022 as compared to 2021, resulted in a mix of mudflats and flooded wetland vegetation on shore in McGee Bay (Figure 87) (Figure 88). The growth of the aquatic plant widgeonweed (*Ruppia* sp.) was extremely heavy in McGee Bay in 2022, making passage with a boat difficult, and requiring multiple stops to clear the propeller.

Figure 87. McGee Bay Shoreline South of McGee and Convict Creek Outflow

Figure 88. McGee Creek Shoreline Area at Pelican Point

North Landing (NOLA)

The North Landing area is influenced by subsurface flows and supports meadow, wet meadow and mudflat habitats. In 2022, some mudflats were present, however much of the shoreline was dry and sandy (
Figure 89).

Figure 89. North Landing

Sandy Point (SAPO)

Most of the length of Sandy Point area is bordered by short cliffs or upland vegetation. Small areas of meadow habitat occur in this area, and limited freshwater input occurs at Green Banks Bay. In fall of 2022, the area primarily supported a narrow sandy beach (Figure 90).

Figure 90. Sandy Point

Upper Owens River (UPOW)

The Upper Owens River receives direct flow from the Owens River, the largest source of fresh water to Crowley Reservoir. Except at high reservoir levels, this subarea includes a large area of exposed reservoir bottom, and variable amounts of mudflats (Figure 91). The growth of the aquatic plant widgeonweed (*Ruppia* sp.) was extremely heavy in the Upper Owens River in 2022, making passage with a boat difficult, and requiring multiple stops to clear the propeller.

Figure 91. Upper Owens Delta

3.4.5 Waterfowl Survey Discussion

3.4.5.1 Summer Ground Surveys – Mono Lake Shoreline

Breeding Population Size and Composition

Following two above-average years of waterfowl breeding activity, the continuing drop in lake level resulted in reduced waterfowl breeding at Mono Lake in 2022. As was the case last year, the behavior of the breeding waterfowl community was altered from what has been typically observed. The breeding waterfowl appeared to be spending more time foraging away from appropriate nest and brooding sites, and commuting to specific areas around the lake where prey and foraging conditions were good at any particular point in time. If this is indeed what was occurring, it suggests that breeding waterfowl arrived at Mono Lake in poor condition, and may have been spending more time feeding in order improve or maintain their body condition, and less time on their nest or brooding. Hens also appeared more willing than usual to abandon their brood in the face of a disturbance.

Spatial distribution

Breeding waterfowl are concentrated into highly localized areas around the shoreline of Mono Lake, where fresh water resources occur for young ducklings. The breeding waterfowl community at Mono Lake is also responsive to annual, and sometime subtle changes in habitat conditions as suggested by the spatial distribution patterns observed.

In 2022, breeding conditions were good along the northwest shore (DeChambeau Creek, Mill Creek, Wilson Creek) and Simons Spring. The northwest shore supports an abundance of fresh water sources from springs and inflow from Mill Creek, and some of the most extensive wet meadow habitat at Mono Lake. Beaver activity in the northwest shoreline area has also led to the development of additional freshwater ponds.

Conditions at Simons Spring remained good despite the decrease in lake level. Small freshwater ponds remained uphill of old littoral bars, although encroachment by emergent vegetation is occurring. As lake level drops, breaches form in the littoral bars and release water onto mudflats. If the lake level continues to decline, narrow channels will form, and result in a drying of the playa, and draining of the ponds. In 2022, lake level was such that small ponds, extensive mudflats, and numerous areas of spring flow to the lake were still present. This often creates favorable conditions for waterfowl breeding and foraging.

Waterfowl conditions at Goose Springs continue to degrade. Although for many years, the Goose Springs area supported the highest number of broods, little waterfowl activity, and no broods were observed in 2022. The abandonment of the Goose

Springs area as a breeding location is likely due to vegetation encroachment, and a resulting change in spring channel flow patterns and a decrease in open water ponds.

Waterfowl habitat conditions in lower Rush Creek and Lee Vining Creek also appeared to degrade as fresh water ponds in the deltas continued to dry. Lower Rush Creek below the County Road continues to show an increase in sinuosity and the presence of backwater ponds, thus improving conditions for waterfowl foraging. As a testament to the responsiveness of waterfowl to local, or even temporary conditions, in late July, a large congregation of hen Gadwall and their broods gathered at the mouth of a small, low flow channel in the Rush Creek delta that provided a shallow and sheltered area for foraging. Based on waterfowl observations throughout the summer, most of the broods were not raised in Rush Creek, but were brought there by the hens to take advantage of good feeding conditions.

Habitat Use

Many studies have shown that waterfowl breeding productivity is linked to the abundance and quality of open water wetlands and ponds supporting high densities of aquatic invertebrates (Cox et al. 1998, Kaminski and Prince 1981, Krapu et al. 1983, Pietz et al. 2003). In addition, the abundance and availability of aquatic invertebrates limits the number of breeding waterfowl and waterfowl brood survival (Sjoberg et al. 2000). Habitat use patterns of the breeding waterfowl community at Mono Lake suggest that freshwater ponds, brackish ponds and ria are key habitat features that support the breeding waterfowl community at Mono Lake.

Young ducklings require fresh water in order to survive and gain weight (Swanson et al. 1984), and thus freshwater resources are a necessary component of the habitat of the breeding waterfowl community at Mono Lake. Freshwater resources at Mono Lake include freshwater ponds, freshwater streams, spring outflow and deltas, where a fresh water lens might occur depending on weather conditions, flow, and shoreline topography.

In 2022, breeding dabbling duck activity was concentrated in and around freshwater sources including ponds, spring and creek outflow areas of ria. Typically, brackish ponds are also heavily used, but few observations were in this habitat in 2022, possibly due to a reduction in brackish pond availability.

Freshwater ponds are an important component of the breeding waterfowl habitat at Mono Lake that was used by all dabbling duck species, but not Canada Goose. Freshwater outflow areas of creeks and springs (=”ria”) were used primarily by Gadwall for feeding, suggesting use of *Artemia*, however, other invertebrates have been observed in outflow areas including midge larvae. Canada Goose, an almost exclusively herbivorous species, was seen most frequently on mudflats, and alkali meadow where they feed on the fresh growth of roots, leaves, and tubers of emergent wetland plants and submerged aquatic plants. Mono Lake lacks submerged aquatic plants due to the salinity of the lake, and thus the sedges, grasses, and other herbaceous vegetation in shoreline meadow habitats at Mono Lake are the prime feeding areas for this species. Typically, the only species that has regularly used meadow habitats is Canada Goose. However, the habitat changes at Warm Spring due to heavy feral horse grazing is attracting Mallard to feed in the meadows and shallow ponds now accessible.

Factors Influencing Waterfowl Breeding Populations

Lake level has strongly influenced the breeding waterfowl population at Mono Lake. Spring lake levels, particularly March and April, have had the largest influence on the size of the breeding population. Spring conditions will influence whether waterfowl pairs chose to settle and breed at Mono Lake. Higher lake levels, at least within the range of lake levels observed, improves waterfowl habitat in general, by increasing shoreline ponds, and decreasing the distance between nesting areas, brooding ponds, and shoreline feeding areas.

Annual brood numbers are strongly influenced by lake level, particularly in the month of June, but the effect is nonlinear. Increases in brood numbers have been observed above a threshold of 6,382 feet. Below this lake level, the total number of broods has not only been significantly fewer, but the number of broods has not been influenced by further declines in lake level. In June 2022, the lake level was 6379.2 feet, thus below the 6,392-foot threshold, and brood numbers were significantly below the long-term mean.

Lake level-related changes to breeding habitat quality and quantity are believed to be the major factor influencing breeding waterfowl populations at Mono Lake. As lake level decreases, the number and size of the ponds- particularly along the south shore from South Shoreline to Simons Springs- decreases. Decreases in lake level also result in increased barren playa at most places around the lake, resulting in increased physical distance between nesting and brooding cover, and high productivity feeding areas near shore. The 6,382-foot threshold is being further investigated to determine critical habitat components that may be influencing this response.

Artemia population levels in early spring and summer were not found to influence the annual breeding population size or broods. It could be that food is super-abundant at Mono Lake during this time period, and not limiting in and of itself, or that the breeding species are not as reliant on *Artemia* as a food source as are fall waterfowl. Other factors such as access to food, which are influenced by lake level and bathymetry, could potentially influence waterfowl breeding, and are being further investigated.

Summer Ground Surveys - Restoration Ponds

The repairs completed in January 2021 restored the ability to deliver warm artesian water to DEPO1, DEPO2, and DEPO3. The waterfowl habitat at the Restoration Ponds continues to be impacted by ageing infrastructure and water delivery problems as the County Ponds remained dry in 2022. Waterfowl use of the restoration pond complex as a whole continued to be below the long-term average, since the County Ponds remain inactive. Repair work to the infrastructure of the DeChambeau ponds continues, with a goal of further improvements in habitat conditions.

3.4.5.2 Fall Counts

Mono Lake - Population Size and Species Composition

The early September count in 2022 was below average, however, waterfowl counts were comparable to the long-term mean for the End-of-September and Mid-October counts. A slight seasonal shift in use was observed in 2022, as was the case for 2021, in that a second pulse of Northern Shovelers at Mono Lake was evident on the End-of-October survey. Past monitoring has shown that waterfowl totals at Mono Lake have been highest during the month of September, with significantly reduced numbers on the Mid-October through Mid-November counts. In 2022, the seasonal peak at Mono Lake was in late-September, and numbers remained high through Mid-October. An obvious second pulse of Northern Shoveler arrived at the end of October, resulting in above-average numbers on the late fall counts. It is likely that early season Northern Shoveler flocks at Mono Lake are originating from a different source population than those arriving later in fall. The second pulse of birds may also be due to seasonal change in weather or drought conditions on the breeding grounds or along migration corridors, pushing birds farther south. Seasonal shifts such as this could also be an indication of waterfowl response to climate change. Waterfowl migration patterns have been observed to change over time (Lehikoinen and Jaatinen 2012, Reese and Weterings 2018), and the timing of waterfowl use may be useful for assessing waterfowl response to regional or local changes in conditions including those induced by climate change. In addition, as discussed in Section 3.2.3, the Artemia population has been trending higher in fall in recent years, and Northern Shoveler may be responding to the increase in food availability.

Waterfowl at Mono Lake appear to respond to local conditions, as spatial distribution patterns would indicate. The spatial distribution of waterfowl at shoreline sites in fall also suggests that waterfowl habitat at Mono Lake is highly localized, and in 2022, the main shoreline areas used were Wilson Creek and Simons Spring. Although the Wilson Creek area makes up <2% of the entire shoreline area, it supported 37% of all waterfowl in 2022. The combination of abundant spring flow, extensive wet meadow habitat upgradient, and shallow offshore gradient in the Wilson Creek bay and the configuration of the shoreline in this area providing protection from wind and wave action, contribute to creating a favorable shallow water feeding and loafing area for fall migrant waterfowl. As discussed previously, waterfowl habitat in the Simons Spring shoreline area was good in 2022, with extensive mudflats, several ponds, and numerous places of spring flow reaching the lake. In 2022, Ruddy Ducks occurred primarily in nearshore and offshore areas of the DeChambeau Embayment. The DeChambeau Embayment was not only the main area of Ruddy Duck concentration in 2022, but large numbers of phalaropes and Eared Grebes were also observed while conducting waterfowl surveys.

There are several things that contribute to the attractiveness of the DeChambeau Embayment to migrating waterbirds in fall. The DeChambeau Embayment is a large area that is wind-sheltered by neighboring hills. The area is also shallow with a complex bathymetry, including the presence of large number of pumice blocks. These pumice blocks provide excellent substrate for brine fly pupa to attach. The shallow, sheltered nature of the bay lends itself to improved foraging efficiency by waterbirds, and reduced energy use.

A time budget study of waterfowl use of shoreline areas and habitats during fall migration would document how fall migratory waterfowl use different shoreline subareas and habitats for feeding, drinking, roosting, or bathing. An understanding of how waterfowl use each subarea and habitat in fall would provide a greater understanding of the specific resources available for waterfowl around the lake, and how they support migratory waterfowl populations.

Waterfowl at Bridgeport and Crowley Reservoirs were similarly concentrated in around areas of fresh water inflow. Several creeks and potentially subsurface inputs from adjacent irrigated pastures exist along the West Bay and south part of the East Bay areas of Bridgeport Reservoir where waterfowl congregate. These delta areas also provide shallow feeding areas and protected bays ideal for dabbling ducks. At Crowley Reservoir, waterfowl concentrated in the McGee Bay and Upper Owens River delta areas. The McGee Bay subarea receives inflow from Convict and McGee Creeks, and spring flow and subsurface flows from irrigation upgradient. Wetland vegetation often extends to the shoreline, with small areas of mudflats present at all except the highest reservoir levels. The other area of waterfowl concentration is the Upper Owens River delta where flows from the Owens River enter the reservoir. Except at very high reservoir levels, this area has extensive mudflats for loafing, shallow feeding areas, and quiet backwater bays. The large amount of the submerged aquatic plant *Ruppia* present in Crowley Reservoir in 2022 likely contributed to record high numbers of waterfowl. *Ruppia* can be an important food source for waterfowl. *Ruppia* produces high-energy nutlets consumed by waterfowl (Euliss et al. 1991), and the floating masses of vegetation also support aquatic invertebrates consumed by waterfowl.

Waterfowl populations at Mono Lake are relatively small compared to Bridgeport and Crowley, likely due to a combination of salinity and water depth which limits feeding opportunities. Salinity and water depth influence not only the types and abundance of food items, but also accessibility. Mono Lake is deep, highly saline, with limited shallow shoreline areas. Despite the productivity of Mono Lake, access of these food resources to dabbling duck species like Northern Shoveler is somewhat limited due to its depth, as foods must not only be available, but accessible. The topography and bathymetry are such that shallow-water feeding areas, especially those near springs, are widely spaced and not extensive. The range of water depths for optimal foraging by dabbling ducks is 2-10 inches (Fredrickson 1982). Prey will generally be less accessible in water depths greater than about 10 inches, and thus foraging efficiency will decrease. At Mono Lake, dabbling ducks have been observed to feed almost exclusively near shore, and more specifically, where the bathymetry data suggests a greater extent of shallow water than areas where waterfowl use is lower or absent.

Lake level and the productivity of secondary producers have been a focus of the Waterfowl Restoration Plan. Unlike breeding waterfowl populations at Mono Lake, fall migratory waterfowl have not been directly influenced by lake level. However, the abundance of *Artemia*, one of the two primary food resources, is important. *Artemia* population cycles have been related to the mixing state of the lake with higher populations generally observed in years of monomixis, particularly in the first year after mixing. As has been observed with the California Gull population at Mono Lake (Burnett et al. 2021), fall waterfowl populations have been responsive to lake productivity and are generally higher during years of monomixis, and lower in periods of meromixus. Waterfowl using Mono Lake must balance the energetic costs of migration and molt and with food intake. If food resources at a migratory stopover location are of sufficient quantity, quality and accessible, fall migrating waterfowl may not be able to meet the energetic demands of migration, and thus will either overfly a location, or shorten their stay. An overall declining trend in *Artemia* discussed in section 3.2 of this report deserves further investigation as long-term productivity of the secondary producers at Mono Lake, including *Artemia* will be important to support several waterbird species reliant on Mono Lake, in addition to waterfowl.

The highly saline water of Mono Lake currently only support *Artemia* and *Ephydra*, however, other species may have occurred historically when the lake was no more than 50 gm/L salinity. The highly saline water also limits the availability of vegetable food sources favored by many dabbling duck species in fall, to isolated fresh water and brackish ponds since the salinity of the lake is above the tolerance of wetland plants.

Mono Lake is deep, highly saline, with limited shallow shoreline areas. These features limit the habitat quality for waterfowl and may ultimately limit recovery of waterfowl populations. In order for waterfowl to meet their energetic demands, food resources need to be accessible, abundant, and of sufficient quality.

3.4.5.3 Aerial Photography of Waterfowl Habitats

In fall of 2022, when photographs of lake-fringing wetlands were taken, the level of Mono Lake was 6379.4 feet, or 1.6 feet lower than at the same time in 2021. Shoreline conditions for waterfowl are dynamic at Mono Lake, and are influenced by lake level, wave and wind action, and spring and other fresh water inflow. Now that a feral horse herd is established in the Mono Basin, grazing is also influencing waterfowl habitat conditions.

As is expected, the continued decline in lake level resulted in an overall increase in barren playa on shore, and fewer shoreline ponds. Waterfowl habitat conditions do not necessarily change in a linear fashion across all shoreline areas, coincident with lake level, however. In 2022, waterfowl habitat conditions were still fairly good at Simons Springs, Wilson Creek, Mill Creek and DeChambeau Creeks due to factors such as the presence of spring flow to the lake shore, onshore ponds, and shallow feeding areas. Conditions were poor along the South Shore Lagoons due to vegetation encroachment and a lack of open water pond. Further drying of the deltas of Rush and Lee Vining Creeks was evident, however, transient conditions provided for excellent waterfowl foraging habitat at the delta of Rush Creek in mid-summer.

Grazing by feral horses was particularly heavy in the Warm Springs and Simons Spring areas. The intense grazing by the feral horses has had some interesting effects, at least in the short-term, on the conditions at Warm Springs, and the dynamics of waterbird use. The heavy grazing has removed much of the dense cover previously in this area. The Warm Springs area continued to be very wet, with multiple shallow, open water ponds, attracting waterbirds to feed and shorebirds to attempt nesting in places previously unavailable because of dense cover.

In contrast to Mono Lake, the levels of Bridgeport and Crowley Reservoirs were higher in fall of 2022 as compared to 2021. At Bridgeport Reservoir the higher reservoir level resulted in moister conditions of the adjacent meadows and floodplain. The level of Crowley Reservoir was slightly increased over fall of 2021, however, large amounts of reservoir bottom remained exposed in the Upper Owens area and other shoreline areas. In fall of 2022, the water appeared fairly clear as compared to other years. Fall of 2022 was notable due to the very heavy growth of widgeongrass in McGee Bay and the Upper Owens River delta areas.

4.0 SUMMARY AND RECOMMENDATIONS

The Mono Basin Waterfowl Habitat Restoration Program was developed to evaluate the effect of changes in the Mono Lake area relative to the restoration objectives, and to provide information to guide future restoration activities. The program has included a number of restoration projects, objectives, and monitoring projects. Restoration has included establishing a target lake elevation, reestablishing perennial flow in tributaries, channel openings, providing financial assistance for the restoration of waterfowl habitat, and exotic species control. Ecological conditions in the Mono Basin have improved considerably as a result of the restoration program.

The implementation of Decision 1631 appears to have resulted in the lake level stabilization, although Mono Lake is still well below the target lake level 27 years later. Climatic factors may be influencing Mono Lake and its recovery. Current trends indicate seasonal increases in salinity and water temperature, a finding aligned with regional climatic trends.

Within the range of lake elevations observed since 2002, shoreline waterfowl habitat in general shows improvement at higher lake level. These improvements include increased shoreline pond acreage and increased connectivity of shoreline ponds with the shoreline and spring outflow areas. Breeding waterfowl have been very responsive to lake level increases, however, fall migratory populations have not, instead responding to the productivity of secondary producers.

Mono Lake is deep, highly saline, with limited shallow shoreline areas. These features limit the habitat quality for waterfowl and may ultimately limit recovery of waterfowl populations. In order for waterfowl to meet their energetic demands, food resources need to be accessible, abundant, and of sufficient quality. The current trends seen in the data with regard to salinity, water temperature and *Artemia* populations, if continued, will also influence waterfowl habitat conditions at Mono Lake.

1) **Continue to implement measures to support lake level recovery.** Of the restoration measure outlined in Order 98-05, lake level recovery remains the single most important measure for improving and maintaining waterfowl habitat. Higher lake levels - at least within the range of levels observed – appear to improve breeding habitat conditions, and increase brood production. Increased lake levels have resulted in more shoreline ponds and greater connectivity between nesting habitats and preferred foraging areas. As shown by the fall waterfowl data, lake level alone may not enhance use by migrating waterfowl, as the biomass of secondary producers such as *Artemia* is a key variable,

potentially swamping out any effect of lake level change. Mono Lake must continue to remain productive in order to continue to support waterfowl populations and improving our understanding of how to support lake productivity should be considered.

2) **Enhance and restore the functioning of the Restoration Ponds.** Although the total number of waterfowl that could be supported by the Restoration Ponds is just a fraction of that occurring on Mono Lake, there are management strategies, repairs, and improvements that would increase waterfowl use. The most basic of improvements would be to restore water delivery to the County Ponds, which have been dry for several years. The second would be to implement seasonal or rotational flooding regime to enhance forage production for waterfowl, while continuing to provide waterfowl habitat year-round at the ponds. In addition, we also recommend the Mono Basin Waterfowl Director work with partners restoring the functioning of the DeChambeau Ponds on ensuring that monitoring efforts are not being duplicated.

3) **Investigate algal community dynamics**. Consideration should be given to conducting more detailed studies and monitoring of the algal community and its dynamics. Notable changes in Mono Lake limnology include a sustained high abundance of phytoplankton starting in 2015, significant decreases in transparency during the summer months, and a weak, downward trend in the *Artemia* population. The extensive limnological dataset has limited information on specifics of the algal community that may help interpret these more recent trends in Mono Lake ecology.

4) **Conduct a second waterfowl time budget study.** Order 98-05 required a time budget study to be conducted during each of the first two fall migration periods after the plan was approved, and again when Mono Lake reaches its target lake elevation. A single time budget study of Ruddy Ducks was completed in fall of 2000 by Jehl. We recommend the Mono Basin Waterfowl Program Director develop a study plan for the second required time budget study focusing on shoreline use by waterfowl. A time budget study allows for the determination of the relative importance of different shoreline sites for migratory waterfowl, and would provide insight into the importance of the various habitat types for feeding, resting, or drinking.

5) **Reinstate the vegetation monitoring programs in lake-fringing wetlands and riparian areas.** The vegetation monitoring conducted at the lake-fringing wetland sites in 2021 (LADWP 2022) documented impacts from feral horse grazing at Warm Springs. In early 2022, horses were first observed in the Rush

Creek delta area. It is recommended that the riparian monitoring program be reinstated, including the transects in Rush Creek and Lee Vining Creek.

6) **Reinstate annual restoration meetings.** In prior years, the Mono Basin Parties met to hear the reports from the scientists and their findings during their annual monitoring. These meetings are an excellent way to foster communication and knowledge sharing among the parties, and I recommend reinstating an annual meeting, perhaps separate from the Annual Operations Plan meeting so that more time is available for the discussion of scientific findings. The advent of the "virtual meeting" makes it easier for all to participate, especially if travel is an issue.

5.0 LITERATURE CITED

Beschta, R. L. 1994. Rush Creek – Flows, channels, and riparian conditions: Pre-1941 and Today. Report for the Rush Creek Restoration Technical Committee.

Burnett, R.D., K. Nelson, and A.E. Schmidt. 2021. Population size and reproductive success of California Gulls at Mono Lake. Contribution No. 2347, Point Blue Conservation Science, Petaluma, CA.

California Invasive Plant Council (Cal-IPC). 2006. California Invasive Plant Inventory. Published February 2006.

Corvallis Environmental Research Laboratory and Environmental Monitoring and Support Laboratory. 1978. U.S. Environmental Protection Agency National eutrophication survey. Working Paper Series. No. 743. Report on Lake Crowley, Mono County, California, EPA Region IX.

Cox, R.R., M.A. Hanson, C.C. Roy, D.H. Johnson, and M.G. Butler. 1998. Mallard duckling growth and survival in relation to aquatic invertebrate. The Journal of Wildlife Management, Vol. 62(1): 124–133.

Dana, G. L., and P. H. Lenz. 1986. Effects of increasing salinity on *Artemia* population from Mono Lake, California. Oecologia 68: 428–436.

Dana, G. L., R. Jellison, J.M. Melack, and G.L. Starrett. 1983. Relationships between *Artemia monica* life history of characteristics and salinity. Hydrobiolgia263: 129-143.

Euliss, N.H., Jr., R.L. Jarvis, and D.S. Gilmer. 1991. Feeding ecology of waterfowl wintering on evaporation ponds in California. The Condor 93:582-590.

Fredrickson, L.H. 1982. Strategies for water manipulations in moist-soil systems. Waterfowl Management Handbook. 26.

Gollop, J. B, and W. H. Marshall. 1954. A guide for aging duck broods in the field. Mississippi Flyway Council. Tech. Sect. Report.

Heath, H. 1924. The external development of certain phyllopods. Journal of Morphology Vol 38(4): 453–483.

Herbst, D. 1986. Comparative studies of the population ecology and life history patterns of an alkaline salt lake insects: *Ephydra (hyropyrus) hians* Say (Diptera: Ephydridae). A thesis submitted to Oregon State University. March 1986.

Herbst, D. 1988. Comparative population ecology of *Ephydra hians* Say (Diptera: Ephydridae) at Mono Lake (California) and Abert Lake (Oregon). Hydrobiologia. 158: 145-166.

Herbst, D. and D. W. Blinn. 1998. Experimental mesocosm studies of salinity effects on the benthic algal community of a saline lake. Journal of Phycology. 34: 772-778.

Horne, A. J. 2003. Report on Bridgeport Reservoir beneficial use impairment: limnology in the summer-fall 2000 and comparisons with 1989. Report prepared for Lahontan Regional Water Quality Control Board. South Lake Tahoe.

House, D. 2021. Fall waterfowl use of Bridgeport Reservoir, Mono County, California. Western Birds 52(4): 278-295.

Jehl, J. R. 1986. Biology of the Red-necked Phalarope (*Phalaropus lobatus*) at the western edge of the Great Basin in fall migration. Great Basin Naturalist Vol. 46(2): Article 1.

Jehl, J. R. 1988. Biology of the Eared Grebe and Wilson's Phalarope in the nonbreeding season: a study of adaptations to saline lakes. Studies in Avian Biology No. 12. Cooper Ornithological Society.

Jehl, J. R. 2002. Waterfowl populations at Mono Lake, California 2001. Hubbs Sea World Institute. Technical Report 2002-330. Prepared for Los Angeles Department of Water and Power.

Jellison, R., H. Adams, and J.M. Melack. 2001. Re-appearance of rotifers in hypersaline Mono Lake, California, during a period of rising lake levels and decreasing salinity. Hydrobiologia 466: 39-43

Jellison, R. and J.M. Melack. 1993. Meromixis in hypersaline Mono Lake, California. 1 Stratification and vertical mixing during the onset, persistence, and breakdown of meromixis. Limnol. Oceanogr. 38(5): 1008–1019.

Jellison, R., J. Romero, and J.M. Melack. 1998. The onset of meromixis during restoration of Mono Lake, California: Unintended consequences of reducing water diversions. Limnol. Oceanogr. 43(4): 706–711.

Jellison, R., K. Rose and J. M. Melack. 2003. Assessment of internal nutrient loading to Crowley Lake, Mono County. Final Report to the State Water Resources Control Board.

Jellison, R. 2011. Field and laboratory protocols for Mono Lake limnological monitoring. Prepared for Los Angeles Department of Water and Power. June 21, 2011.

Jellison, R. and K. Rose. 2011. Mixing and plankton dynamics in Mono Lake, California. Marine Sciences Institute. University of California, Santa Barbara.

Kaminski, R. M., and H. H. Prince. 1981. Dabbling duck activity and foraging responses to aquatic macroinvertebrates. The Auk 98(1): 115-126.

Kelts, L.J. 1979. Ecology of a tidal marsh corixid, *Trichoorixa verticalis* (Insect, Hemiptera). Hydrobiologia 64: 37-57.

Kooloos, J.G.M., Kraaijeveld, A.R., Langenbach, G.E.J., and Zweers, G.A. 1989. Comparative mechanics of filter feeding in *Anas platyrhynchos*, *Anas clypeata*, and *Aythya fuligula* (Aves, Anseriformes). Zoomorphology 108: 269-290. https://doi.org10.1007/BF00312160.

Krapu, G., and A. Klett, and D. Jorde. 1983. The effect of variable spring water conditions on Mallard reproduction. The Auk 100: 689–698.

Lehikoinen, A., and K. Jaatinen. 2012. Delayed autumn migration in northern European waterfowl. J. Ornithol. 153:563-570.

Los Angeles Department of Water and Power (LADWP). 1987. Mono Basin geology and hydrology. Prepared by Aqueduct Division Hydrology Section. March 1987.

Los Angeles Department of Water and Power (LADWP). 1996a. Mono Basin waterfowl habitat restoration plan. Prepared for the State Water Resources Control Board. In response to Mono Lake Basin Water Right Decision 1631.

Los Angeles Department of Water and Power (LADWP). 1996b. Mono Basin stream and stream channel restoration plan. Prepared for the State Water Resources Control Board. In response to Mono Lake Basin Water Right Decision 1631.

Los Angeles Department of Water and Power (LADWP). 2004. Mono Basin waterfowl habitat restoration compliance report. Prepared for the State Water Resources Control Board. In response to State Water Resources Control Board Order Nos. 98-05 and 98-07.

Los Angeles Department of Water and Power (LADWP). 2018. Mono Basin Waterfowl Habitat Restoration Program Periodic Overview Report. Prepared by Deborah House and Motoshi Honda for the State Water Resources Control Board. May 2018.

Los Angeles Department of Water and Power (LADWP). 2022. Mono Basin waterfowl habitat restoration compliance report. Prepared for the State Water Resources Control Board. In response to State Water Resources Control Board Order Nos. 98-05 and 98-07.

McBain and Trush, Inc, and R. Taylor and Associates. 2010. Mono Basin stream restoration and monitoring program: synthesis of instream flow recommendations to the State Water Resources Control Board and the Los Angeles Department of Water and Power. Final Report. April 30, 2010.

Melack, J. M. 1983. Large, deep salt lakes: a comparative limnological analysis. Hydrobiologia 105: 223–230.

Pelagos. 1987. A bathymetric and geological survey of Mono Lake, California, unpublished report to the Los Angeles Department of Water and Power. Pelagos Corp. San Diego, Calif.

Pietz, P., G. Krapu, D. Brandt, and R. Cox, Jr. 2003. Factors affecting Gadwall brood and duckling survival in prairie pothole landscapes. The Journal of Wildlife Management. Vol 67(3): 564–575.

Raumann, C. G., S. Stine, A. Evans, and J. Wilson. 2002. Digital bathymetric model of Mono Lake, California. Miscellaneous Field Studies Map MF 2393.

Reese, J.G, and R. Weterings. 2018. Waterfowl migration chronologies in central Chesapeake Bay during 2002-2013.

Reheis, M. C., S. Stine, and A. M. Sarna-Wojcicki. 2002. Drainage reversals in Mono Basin during the late Pliocene and Pleistocene. Geological Society of America Bulletin. V:114(8): 991-1006. August 2002.

Rogers, D.B., S.J. Dreiss and D.P. Groeneveld. 1992. Near-Shore Groundwater and Salt-Flat Processes at Mono Lake, White Mountain Research Station Symposium, 4, 367–395.

Russell, I.C. 1889. Quaternary history of Mono Valley, California. University of Michigan Library.

Sharpe, S. E., M. E. Cablk, and J. M. Thomas. 2007. The Walker Basin, Nevada and California: physical environment, hydrology, and biology. Desert Research Institute Publication No. 41231. May 2007. Revision 01 May 2008.

Sjoberg, K., H. Poysa, J. Elmberg, and P. Nummi. 2000. Response of Mallard ducklings to variation in habitat quality: an experiment of food limitation. Ecology 81(2): 329–335.

Starrett, G. L., and W. M. Perry. 1985. Multiple generation salinity acclimation experiment. Report to the Los Angeles Department of Water and Power.

State Water Resources Control Board (SWRCB). 1994. Mono Lake Basin water right decision 1631.

State Water Resources Control Board (SWRCB). 1998. Order WR 98-05. Order requiring stream and waterfowl habitat restoration measures. September 2, 1998.

Strickland, J.D.H., and T.R. Parsons. 1972. A practical handbook of seawater analysis. Bulletin 167 (2nd ed.). Fisheries Research Board of Canada, Ottawa, Canada.

Swanson, G. A., V.A. Adomaitis, F.B. Lee, J.R. Serie, and J.A. Shoesmith. 1984. Limnological conditions influencing duckling use of saline lake in South-Central North Dakota. The Journal of Wildlife Management. Vol. 48(2): 340-349.

University of California. 2010. Saltcedar. A nonnative invasive plant in the western U.S. WRIC leaflet #02-2.

U.S.D.A Forest Service. 2005. Decision Memo. Pond 3-Barrow Pit Pond Wildlife Habitat Improvement. Inyo National Forest. Mono County, CA. July 18, 2005.

Verschuren, D., J. Tibby, K. Sabbe, and N. Roberts. 2000. Effects of depth, salinity, and substrate on the invertebrate community of a fluctuating tropical lake. Ecology 81(1): 164-182.

Vorster, P. 1985. A water balance forecast model for Mono Lake, California. Monograph 10. USDA Forest Service Region 5.

Williams, W. D. 2002. Environmental threats to salt lakes and the likely status of inland saline ecosystems in 2025. Environmental Conservation 29(2): 154–167.

Wurtsbaugh, W.A., C. Miller, S. Null, R. Justin DeRose, P. Wilcock, M. Hahenberger, F. Howe, and J. Moore. 2017. Decline of the worlds saline lakes. Nature Geoscience 10: 816–821.

Zellmer, J. T. 1977. Environmental and engineering geology of Bridgeport Valley, Mono County California. Thesis. University of Nevada, Reno.