Los Angeles Department of Water & Power

BUILDING A STRONGER L.A.

May 12, 2022

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: Compliance with State Water Resources Control Board Order Nos. 98-05 and 98-07

Pursuant to the State Water Resources Control Board Decision No. 1631 and Order Nos. 98-05 and 98-07 (Orders), and in accordance with the terms and conditions of the Los Angeles Department of Water and Power (LADWP) Mono Basin Water Rights License Nos. 10191 and 10192, enclosed is a compact disc (CD) containing a submittal, "Compliance Reporting May 2022", which includes the following four reports required by the Orders. The reports are as follows:

- Section 1: Mono Basin Annual Operations Plan: Planned Operations for Runoff Year (RY) 2022-23.
- Section 2: Mono Basin Fisheries Monitoring Report Rush, Lee Vining, and Walker Creeks 2021
- Section 3: Stream Monitoring Report RY 2021-22
- Section 4: Mono Basin Waterfowl Habitat Restoration Program 2021 Monitoring Report

The filing of these reports, along with the restoration and monitoring performed by LADWP in the Mono Basin, fulfills LADWP's requirements for RY 2021-22 as outlined in Decision 1631 and the Orders, as well as the renewed Temporary Urgency Change Petition.

Electronic copies of the submittal on CD will be provided to the interested parties listed on the enclosed distribution list. Hard copies of the submittal will be provided upon request.

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

Sincerely,

Adam Perez
Manager of Aqueduct

JC:mt
Enclosures
c/enc: Distribution List
Mr. Mark Y. Ching

In Response to the
State Water Resources Control Board
Order Nos. 98-05 and 98-07

COMPLIANCE REPORTING

Mono Basin Operations
Fisheries Monitoring
Stream Monitoring
Waterfowl Habitat & Population Monitoring

May 2022
Los Angeles Department of Water and Power

Section 1

Mono Basin Operations

Compliance with State Water Resources Control Board Decision 1631 and Order Nos. 98-05 and 98-07

May 2022

Los Angeles Department of Water and Power

Mono Basin Runoff Year 2022-23
Annual Operations Plan

Licenses 10191 and 10192
Order WR 2021-0086 EXEC – October 1, 2021

May 2022

Los Angeles Department of Water and Power

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I. Abbreviations, Definitions, Memberships Table

amsl	above mean sea level
AF	acre-feet
AFA	acre-feet per annum
AOP	Annual Operations Plan
CDFW	California Department of Fish and Wildlife
CEQA	California Environmental Quality Act
cfs	cubic feet per second
DSOD	California Department of Water Resources, Division of Safety of Dams
Deputy Director	Deputy Director for the Division of Water Rights
Division	Division of Water Rights
GLOMP	Grant Lake Operations and Management Plan
GLR	Grant Lake Reservoir
Grant Outlet	Grant Lake Outlet
LADWP	Los Angeles Department of Water and Power (Licensee)
MAT	Mono Basin Monitoring Administration Team
MBOP	Mono Basin Operations Plan
MGORD	Mono Gate One Return Ditch
Monitoring Directors	Stream Monitoring Team, Limnology Director, and Waterfowl Director
Parties	California Department of Fish and Wildlife, Mono Lake Committee, and California Trout
RCTE	riffle crest thalweg elevation
RY	runoff year
SCE	Southern California Edison
SEFs	Stream Ecosystem Flows
SMT	Stream Monitoring Team
State Water Board	California State Water Resources Control Board
TUCP	Temporary Urgency Change Petition
USFS	United States Forest Service
USGS	United States Geological Service
Teams and Directors as of Current Runoff Year:	
MAT	The Parties and the Licensee
SMT	Bill Trush, Ross Taylor
Waterfowl Director	Debbie House (Interim)
Limnology Director	Dr. John Melack

II. Introduction

The purpose of the AOP is to describe how operations will work for the current year-type to accomplish exports and stream releases in accordance with the water license. The AOP will provide specific information about the flow schedule, export, and facility operations for the year ahead. The AOP will also review the prior year's plan and compare it to actual runoff and operations.

The timeline for AOP development and submittal is as follows:

- By March 31: convene a meeting to prepare for developing the AOP. Meeting attendees to include the SMT, the Waterfowl Director, the Limnology Director, and the Parties.
- By April 15: distribute a draft AOP to the Waterfowl Director, Limnology Director, and the Parties.
- By May 5: convene a meeting to resolve any unresolved issues.
- By May 15: submit AOP to the SWRCB Deputy Director for a 30-day review, modification, and approval if necessary. No Division approval will be necessary if the terms of the AOP are entirely within the parameters of the MBOP then in effect.

III. Summary of Mono Basin RY 2021-22 Operations

For RY 2021-22, Mono Basin was operated under renewed TUCPs approved by the SWRCB, pursuant to Water Code Section No. 1435, and the amended licenses (after October 1, 2021).

RY 2021-22 was classified as a Dry Year; thus, the specific flow tables were Table 1G for Rush Creek, and Tables 2B (spring and summer) and 2C (fall and winter) for Lee Vining Creek. No water diversions occurred on Walker or Parker Creeks. The April 1, 2021 runoff forecast was 68,800 AF of runoff, and the actual runoff was 53,314 AF.

The Five Siphons spillway was utilized during Summer 2021, based on GLR elevations as of June 30. For employee and general public safety, due to construction work taking place on the Lee Vining Conduit outlet at GLR, the bulkhead at Five Siphons was left in place through November 4 instead of October 1. After October 1, while the work was taking place, all water in Lee Vining was released down the creek and no diversions occurred. However, occasional leaked water entered Lee Vining Conduit, totaling approximately 65 AF during the October 1 to November 4 time frame.

GLR did not spill.

See the following table for total volume and peak flow rate information for various Mono Basin waterways. The flow tables were generally followed; however, from September 1 to 17, flows above 23 cfs in Lee Vining Creek were diverted to Lee Vining Conduit, for the safety of staff and stream scientists performing biological studies downstream.

Lee Vining Creek operations were based on upstream flows according to Table 2B and adjusted on an hourly basis. Diversions may have occurred throughout the day when flows exceeded 30 cfs and stopped when flows decreased according to Table 2B. This hourly-based method of operation maintained compliance with the SEF requirements, but may not be apparent when viewing average daily flow data.

Location	Total Volume (AF)	Peak Daily Avg. (cfs)	Peak Flow Date
Rush Creek at Dam Site	19501	99	6/3/2021
Lee Vining Creek Above	22593	114	5/13/2021
Parker Creek	5354	31	6/5/2021
Walker Creek	2405	15	6/5/2021
Return Ditch	27329	74	5/19/2021
5 Siphons	600	(-)	(-)
Grant Lake Spill	0	(-)	(-)

See the following table for water level elevations in GLR and Mono Lake during RY 2021-22.

Month	Grant Lake Elevation	Grant Lake Storage	Mono Lake Elevation USGS reference
4/1/2021	7104.7	22,603	6381.29
5/1/2021	7105.8	23,563	6381.19
6/1/2021	7108.0	25,388	6381.01
7/1/2021	7107.2	24,740	6380.86
8/1/2021	7105.1	22,976	6380.58
9/1/2021	7103.6	21,713	6380.22
10/1/2021	7102.9	21,168	6379.89
11/1/2021	7098.6	17,860	6379.75
12/1/2021	7094.9	15,229	6379.79
1/1/2022	7093.5	14,320	6379.86
2/1/2022	7092.4	13,572	6379.92
3/1/2022	7091.1	12,680	6379.92

A total of 13,480 AF of water was exported from Mono Basin, under the 16,000 AF limit, based on the April 1, 2021, Mono Lake elevation.

IV. Proposed Mono Basin Operations Plan for RY 2022-23

A. Forecast for RY 2022-23

The runoff forecast for RY 2022-23 is 60% of normal, which is classified as a "Dry" year and calls for Rush Creek and Lee Vining Creek operations to be based on Tables 1G, 2B, and 2C, respectively. These tables are included in the Attachments**.** The Mono Basin's April 1 forecast for RY 2022-23 for April to March period is 70,900 AF (see Attachments).

B. Adaptive Management

LADWP has not received any adaptive management recommendations for RY 2022-23 at the time of this report. The SMT can provide adaptive management recommendations for flow requirements (such as ramping rates, durations, timing, and/or start and end dates) for SEF Tables 1 and 2, per amended license 11.a.1, 20.f.3 and 20.f.4. The SMT will produce an Annual Monitoring Report to document monitoring observations and discuss possible adaptive management recommendations.

Real-time adaptive management in response to unforeseen circumstances may also be proposed by the SMT, per amended license 20.f . Such recommendations will be made by written notice to the Division, and they shall be developed in consultation with the Licensee and Parties.

Adaptive management recommendations are subject to review, modification, and approval of the Deputy Director.

C. Planned Operations

Operations for RY 2022-23 will be based on the Mono Basin runoff forecast, SEF tables, Mono Lake elevation, and any events that may arise during the course of the year. Events such as stream monitoring or other work may require flow reductions for safety purposes, during the monitoring/work period. RY operations were modeled in eSTREAM using April 1, 2022 elevations of GLR and Mono Lake, as well as the SEF tables for Rush Creek (1G) and Lee Vining Creek (2B and 2C). Given the dry year and forecasted runoff, license condition 11.b.2.i (20,000 AF GLR storage July 1 – September 30) may not be met depending on actual runoff volume and timing. LADWP will delay exports until October in response to the projected GLR storage level during summer.

LADWP does not plan to add or replace staff gages at Mono Lake this RY. The current staff gage in use is "1S" with a zero elevation of 6377.79 feet (USGS reference).

Additionally, due to possible construction at GLR spillway, GLR elevations may be operated at lower than a typical year without construction activities. The maximum GLR elevation during spillway construction is 7,106 feet (23,700 AF storage). To accomplish this, export timing may be shifted to match the construction period, and/or flows above

Rush Creek SEFs during spring and summer may be released from GLR. High baseflows in fall and winter in Rush Creek may impact trout survivability, so releases above SEFs earlier in the year will help minimize SEF deviations in fall and winter. If construction does begin this RY that requires changes in GLR operations, LADWP will contact Parties ahead of time to discuss changes to planned operations.

The starting conditions for RY 2022-23 include GLR elevation at 7,089.5 feet (storage = 11,610 AF), and Mono Lake elevation at 6,379.9 feet. Daily flow rates downstream of GLR and Lee Vining Intake will be based on SEF tables, with possible spillway construction impacts as discussed above. Planned exports will be 4,500 AF or less, based on the April 1, 2022, Mono Lake elevation. Modeled export flow in eSTREAM was approximately 25 cfs from October through December; actual export may vary from the model run but will be at a steady rate starting in October, likely to be 25 cfs +/- 10 cfs. Based on projected GLR elevations Five Siphons operation will be required from July 1 to September 30. Lee Vining Creek and Conduit daily flows depend on both hydrology and SCE operations, and therefore may differ from eSTREAM model flows.

Per amended license 11.b.1, if GLR storage is at or below 11,500 AF, outflows will be matched to inflows unless SEF requirements are less than inflows, in which case SEFs will be released. After the initial setting of outflow to match the prior day's average flow at Rush Creek at Dam Site, LADWP plans to then change outflow every Tuesday, based on the average flow of the preceding Tuesday to Monday time period, but only if the outflow must change by at least 5 cfs from the prior setting. If Parties disagree with weekly averaging, LADWP will match outflow to inflow daily, based on the prior day's average flow and only if the outflow needs to change by at least 5 cfs from the prior setting. No exports will take place while GLR storage is at or below 11,500 AF.

Each year the GLR outlet valve must be cycled per DSOD requirements. The planned cycling period will depend on the particular SEF tables for Rush Creek; in general, the cycling will take place during periods of higher SEF flows in the summer months. The downstream effects will include a reduction and then an increase in flows, followed by a return to the SEF flow rate at the completion of the cycling exercise. The cycling procedure occurs over a 2 or 3 hour period and the reduction and increase in flows is attenuated downstream due to the relatively short duration of flow variation. SEF flow values will likely be met during the cycling exercise based on past experience.

Using representative historical inflow data (1992 runoff year at 60.5 percent of normal), and above-specified flows, projected GLR elevations and flows, along with projected Mono Lake elevations are shown in the Attachments. Forecasted scenarios are predictive based on historical statistically significant data and assume similar conditions moving forward to the comparative year. Operations are subject to change due to actual hydrology, and SCE operations, during the upcoming RY. LADWP will notify the Parties of adjustments in operations via electronic communication within 3 days, if changes are in conflict with amended license requirements. Otherwise, monthly reports will document adjustments in operations.

ATTACHMENTS

Attachment 1 – SEF Tables

TABLE 1G: RUSH CREEK STREAM ECOSYSTEM FLOWS FOR DRY YEARS

Hydrograph Component	Timing	Flow Requirement	Ramping Rate
Spring Baseflow	April 1 – April 30	30 cfs	Maximum: 10% or 10 cfs*
Spring Ascension	May 1 – May 18	30 cfs ascending to 70 cfs	Target: 5% Maximum: 25%
Snowmelt Bench	May 19 – July 6	70 cfs	Maximum Ascending: 20% Maximum Descending: 10% or 10 cfs*
Medium Recession (Node)	July 7 – July 12	70 cfs descending to 48 cfs	Target: 6% Maximum: 10% or 10 cfs*
Slow Recession	July 13 – July 28	48 cfs descending to 30 cfs	Target: 3% Maximum: 10% or 10 cfs*
Summer Baseflow	July 29 – September 30	30 cfs target 28 cfs minimum	Maximum: 10% or 10 cfs*
Fall and Winter Baseflow	October 1 – March 31	27 cfs target 25 cfs minimum and 29 cfs maximum	Maximum: 10% or 10 cfs*
			* whichever is greater

TABLE 2B: LEE VINING CREEK STREAM ECOSYSTEM FLOWS

Timing: April 1 – September 30						Year-type: Dry/Normal I, Dry				
Maximum ramping at the beginning and end of this period is 20%.										
Inflow	**Flow Requirement**									
30 cfs or less	Licensee shall bypass inflow.									
31 – 250 cfs	Licensee shall bypass flow in the amount corresponding to inflow which is displayed as blocks of 10 cfs (left-hand vertical column) and 1 cfs increments within such blocks (top horizontal row).									
	0	1	2	3	4	5	6	7	8	9
30		30	30	30	30	30	30	30	30	30
40	30	30	30	30	30	30	30	30	30	30
50	30	30	30	30	30	30	30	30	31	32
60	32	33	34	34	35	36	36	37	38	38
70	39	40	41	41	42	43	43	44	45	45
80	46	47	47	48	49	49	50	51	52	52
90	53	54	54	55	56	56	57	58	59	59
100	60	61	61	62	63	64	64	65	66	66
110	67	68	69	69	70	71	72	72	73	74
120	74	75	76	77	77	78	79	80	80	81
130	82	82	83	84	85	85	86	87	88	88
140	89	90	91	91	92	93	94	94	95	96
150	97	97	98	99	100	100	101	102	103	103
160	104	105	106	106	107	108	109	109	110	111
170	112	112	113	114	115	115	116	117	118	118
180	119	120	121	121	122	123	124	124	125	126
190	127	128	128	129	130	131	131	132	133	134
200	134	135	136	137	138	138	139	140	141	141
210	142	143	144	144	145	146	147	148	148	149
220	150	151	151	152	153	154	155	155	156	157
230	158	158	159	160	161	162	162	163	164	165
240	165	166	167	168	169	169	170	171	172	172
250	173									
251 cfs and greater	Licensee shall bypass inflow.									

TABLE 2C: LEE VINING CREEK STREAM ECOSYSTEM FLOWS

<table>
<tr><td colspan="2">Timing: October 1 – March 31</td><td colspan="3">Year-type: All</td></tr>
<tr><td colspan="5">Maximum ramping at the beginning and end of this period and at all times is 20%.</td></tr>
<tr><td>Timing</td><td colspan="4">Flow Requirement</td></tr>
<tr><td></td><td>Extreme/Wet, Wet</td><td>Wet/Normal</td><td>Normal</td><td>Dry/Normal II, Dry/Normal I, Dry</td></tr>
<tr><td>October 1 – October 15</td><td>30 cfs</td><td>28 cfs</td><td>20 cfs</td><td rowspan="4">16 cfs</td></tr>
<tr><td>October 16 – October 31</td><td>28 cfs</td><td>24 cfs</td><td rowspan="3">18 cfs</td></tr>
<tr><td>November 1 – November 15</td><td>24 cfs</td><td>22 cfs</td></tr>
<tr><td>November 16 – March 31</td><td>20 cfs</td><td>20 cfs</td></tr>
</table>

Attachment 2 – Mono Basin Runoff Forecast

2022 EASTERN SIERRA
RUNOFF FORECAST
April 1, 2022

APRIL THROUGH SEPTEMBER RUNOFF

	MOST PROBABLE VALUE (Acre-feet)	MOST PROBABLE VALUE (% of Avg.)	REASONABLE MAXIMUM (% of Avg.)	REASONABLE MINIMUM (% of Avg.)	LONG-TERM MEAN (1971 - 2020) (Acre-feet)
MONO BASIN:	**56,200**	**56%**	69%	43%	100,307
OWENS RIVER BASIN:	**117,200**	**39%**	52%	26%	300,298

APRIL THROUGH MARCH RUNOFF

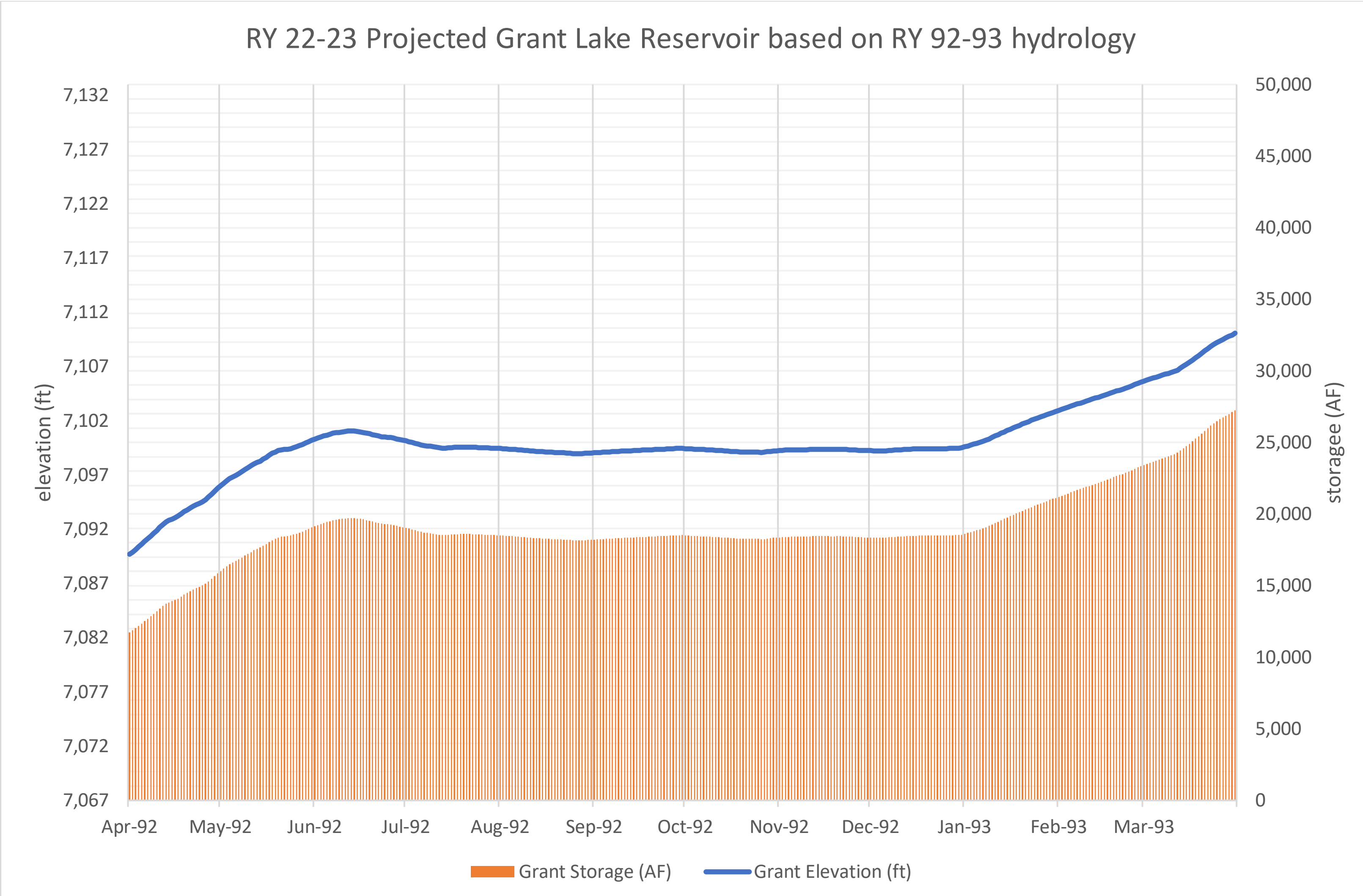
	MOST PROBABLE VALUE (Acre-feet)	MOST PROBABLE VALUE (% of Avg.)	REASONABLE MAXIMUM (% of Avg.)	REASONABLE MINIMUM (% of Avg.)	LONG-TERM MEAN (1971 - 2020) (Acre-feet)
MONO BASIN:	**70,900**	**60%**	74%	46%	118,170
OWENS RIVER BASIN:	**194,300**	**47%**	60%	35%	409,364

MOST PROBABLE - That runoff which is expected if median precipitation occurs after the forecast date.

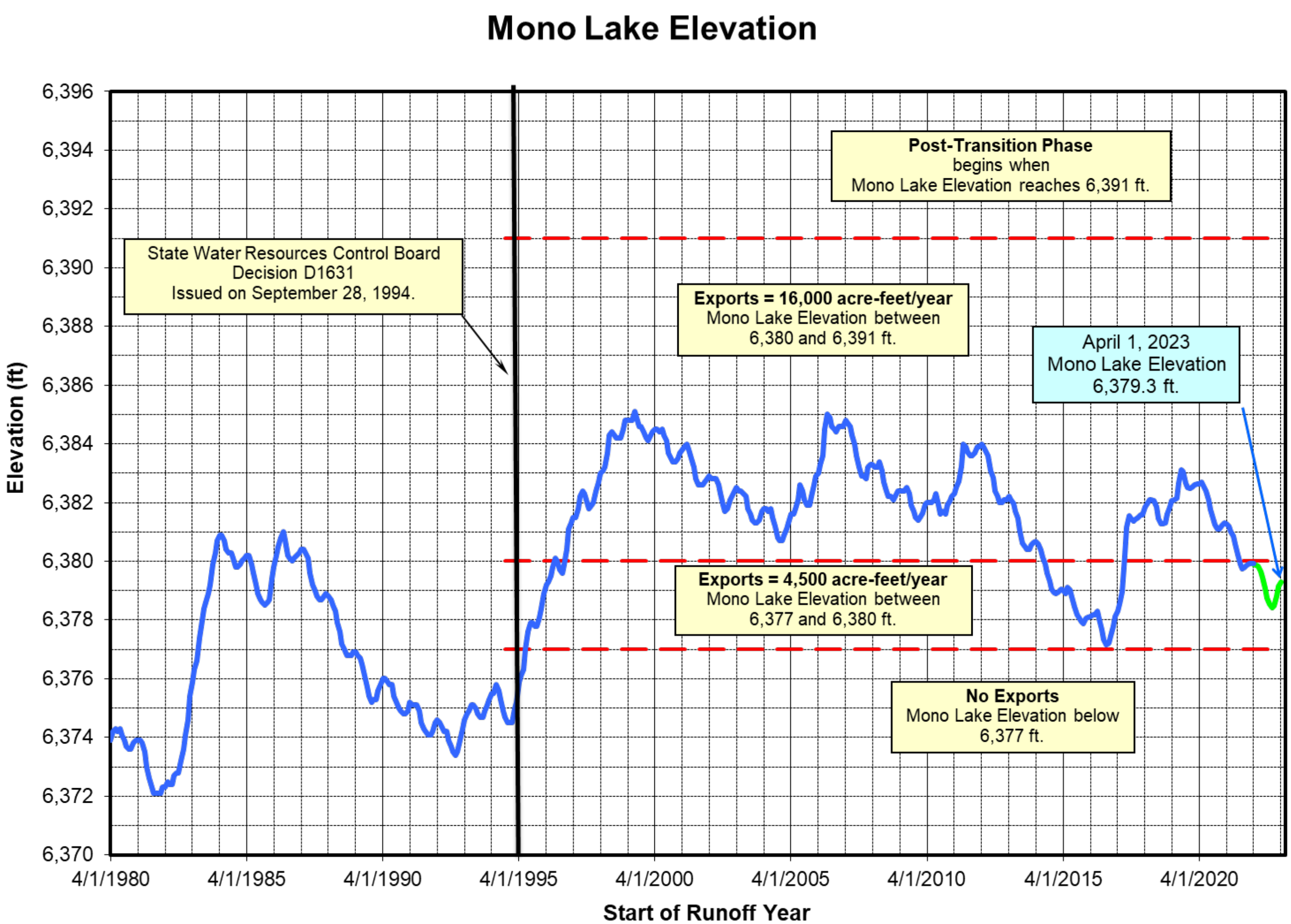
REASONABLE MAXIMUM - That runoff which is expected to occur if precipitation subsequent to the forecast is equal to the amount which is exceeded on the average once in 10 years.

REASONABLE MINIMUM - That runoff which is expected to occur if precipitation subsequent to the forecast is equal to the amount which is exceeded on the average 9 out of 10 years.

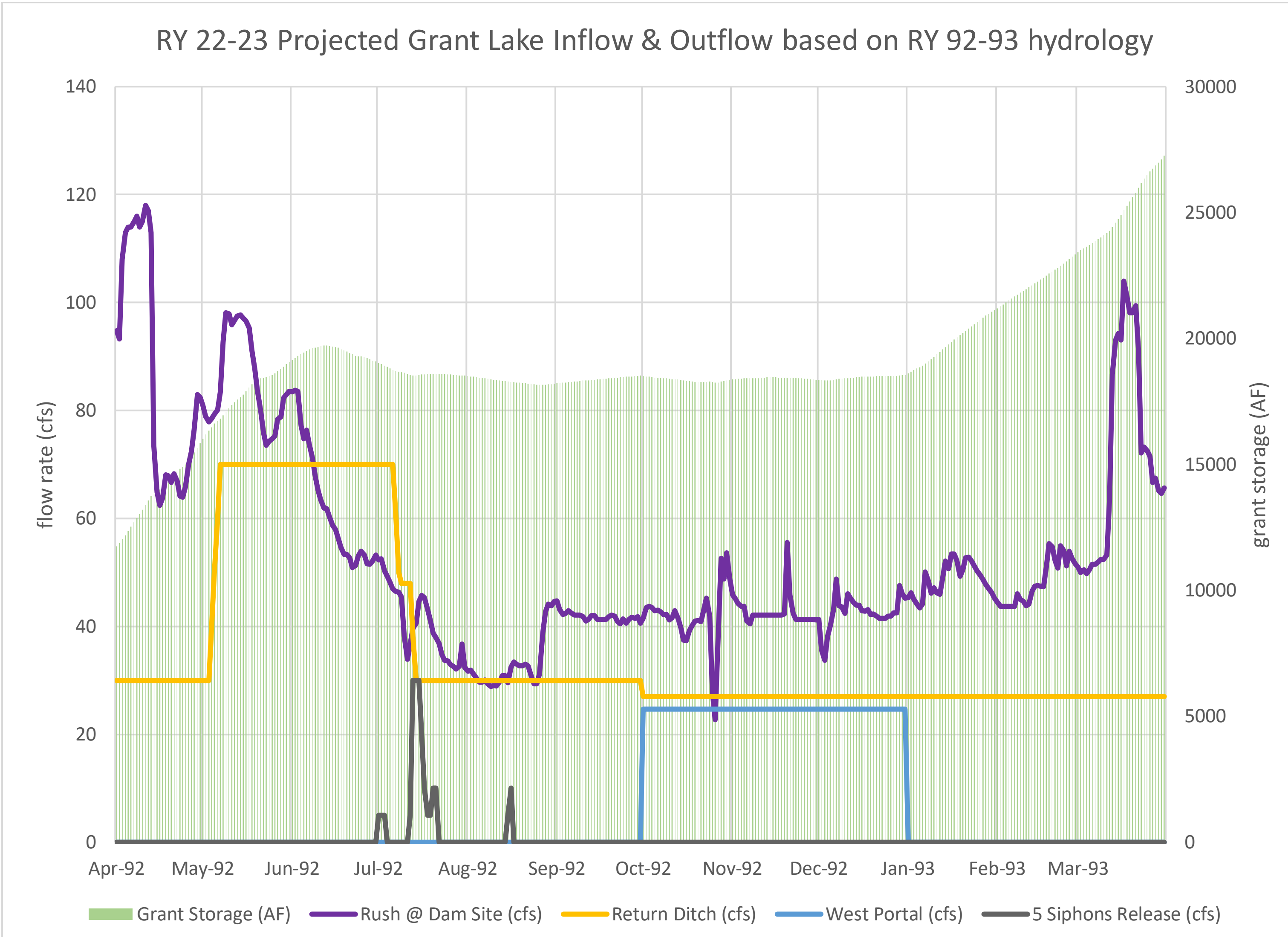
Attachment 3 – Projected GLR Elevations



Attachment 4 – Projected Mono Lake Elevations



Attachment 5 – Projected GLR Flows



Section 2

Mono Basin Fisheries Monitoring Report: Rush, Lee Vining, Parker, and Walker Creeks 2021

Mono Basin Fisheries Monitoring Report
Rush, Lee Vining, and Walker Creeks
2021

Prepared by Ross Taylor and Associates for

Los Angeles Department of Water and Power's Annual Compliance Report to the State Water Resources Control Board

April 15, 2022

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Executive Summary

This report presents results of the 25th year of trout population monitoring for Rush, Lee Vining, and Walker Creeks pursuant to SWRCB's Water Right Decision 1631 (D1631) and the 23rd year following SWRCB Orders #98-05 and #98-07. Order #98-07 stated that the monitoring team would develop and implement a means for counting or evaluating the number, weights, lengths and ages of trout present in various reaches of Rush Creek, Lee Vining Creek, Parker Creek and Walker Creek. This report provides trout population and demographic data collected in 2021 as mandated by the Orders and the Settlement Agreement.

The 2021 runoff year (RY) was 58% of normal and classified a Dry RY type, as measured on April 1, 2021. The range of runoff that defines a Dry RY is ≤ 68.5% (80% - 100% exceedence). The preceding nine years included a Dry-Normal 1 RY of 71% in 2020, 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 years (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 the Lee Vining Creek side channel and in Walker Creek. Although single-pass electrofishing has typically been performed in the MGORD during odd-years, we conducted two passes in 2021 because we only conducted a single pass in 2020 when wildfires prevented us from completing the recapture runs at all mark-recapture study sections.

Population Estimates

The Upper Rush section supported an estimated 467 age-0 Brown Trout in 2021 compared to 1,868 age-0 Brown Trout in 2020 and 2,647 age-0 fish in 2019. This section supported an estimated 586 Brown Trout 125-199 mm in length in 2021 compared to 859 fish in 2020 and 616 fish in 2019. In 2021, Upper Rush supported an estimated 63 Brown Trout ≥200 mm in length compared to an estimate of 93 fish in 2020 and 203 fish in 2019. In 2021, the Upper Rush section supported an estimated 77 Rainbow Trout <125 mm in length (253 fish in 2020), an estimated 25 Rainbow Trout 125-199 mm in length (119 fish in 2020), and an estimated nine Rainbow Trout ≥200 mm in length.

The Bottomlands section supported an estimated 677 age-0 Brown Trout in 2021 compared to 662 fish in 2020 and 638 age-0 fish in 2019. This section supported an estimated 345 Brown Trout 125-199 mm in length in 2021 compared to 364 fish in 2020 and 433 fish in 2019. The Bottomlands section supported an estimated 41 Brown Trout ≥200 mm in 2021 compared to 67 fish in 2020.

In 2021, the MGORD section of Rush Creek supported an estimated 677 age-0 Brown. For this section, the 2021 population estimate for Brown Trout in the 125-199 mm size class equaled 154 fish compared to 446 fish in 2020. The 2021 population estimate of Brown Trout ≥200 mm in length in the MGORD was 625 fish, versus 583 fish in 2020.

Lee Vining Creek's main channel section supported an estimated 57 age-0 Brown Trout in 2021, compared to an estimated 449 fish in 2020 and 414 age-0 fish in 2019. This section supported an estimated 402 Brown Trout 125-199 mm in length in 2021 compared to 171 fish in 2020 and 118 fish in 2019. Lee Vining Creek's main channel supported an estimated 51 Brown Trout ≥200 mm in 2021 versus 24 fish in 2020 and 48 fish in 2019.

A total of four Rainbow Trout were captured in Lee Vining Creek's main channel in 2021. These fish were 65, 81, 212 and 332 mm in length.

The 2021 age-0 Brown Trout estimate for Walker Creek was 227 fish, compared to 180 fish in 2020 and 179 fish in 2019. The 2021 population estimate for Brown Trout in the 125-199 mm size class equaled 119 fish, compared to 139 fish in 2020 and 70 fish in 2019. The 2021 population estimate of Brown Trout ≥200 mm in length was 16 fish, compared to 45 fish in 2020 and 34 fish in 2019.

In the Lee Vining Creek side channel, 33 Brown Trout were captured in two electrofishing passes during the 2021 sampling (16 fish in three passes during the 2020 sampling and 21 fish in two passes during the 2019 sampling). The estimates for each size class were: <125 mm = 21 fish; 125-199 mm = eight fish; and ≥200 mm= four fish. No Rainbow Trout were captured in the side channel in 2021. This was the 13th consecutive year that no age-0 Rainbow Trout were captured in the Lee Vining Creek side channel and the 11th consecutive year that no age-1 and older Rainbow Trout were captured.

Densities of Age-0 Brown Trout

In 2021, the Upper Rush section's estimated density of age-0 Brown Trout was 1,657 fish/ha and the Bottomlands section's estimated density of age-0 Brown Trout equaled 2,347 fish/ha. In Walker Creek, the 2021 density estimate of age-0 Brown Trout was 5,147 fish/ha. In the MGORD, the estimated density of age-0 Brown Trout was 357 fish/ha in 2021.

The 2021 age-0 Brown Trout density estimate in the main channel of Lee Vining Creek was 419 fish/ha. In 2021, the age-0 Brown Trout density estimate in the Lee Vining Creek side channel equaled 625 fish/ha.

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

In 2021, the Upper Rush section's estimated density of age-1+ Brown Trout was 2,302 fish/ha and the Bottomlands section's estimated density of age-1+ Brown Trout equaled 1,338 fish/ha.

In Walker Creek, the 2021 density estimate of age-1+ Brown Trout was 3,061 fish/ha. In the MGORD, the 2021 density estimate of age-1+ Brown Trout was 411 fish/ha.

The 2021 age-1+ Brown Trout density estimate in the main channel of Lee Vining Creek was 3,350 fish/ha. In 2021, the Lee Vining Creek side channel's density estimate of age-1 and older Brown Trout was 357 fish/ha.

Standing Crop Estimates

In 2021, the estimated standing crop for Brown Trout in the Upper Rush section was 127.2 kg/ha and the estimated standing crop for Rainbow Trout was 10.8 kg/ha, thus the total standing crop equaled 138 kg/ha. The estimated standing crop for Brown Trout in the Bottomlands section of Rush Creek was 78 kg/ha in 2021. The estimated standing crop for Brown Trout in Walker Creek was 158 kg/ha in 2021. The MGORD's estimated standing crop of Brown Trout equaled 67 kg/ha in 2021.

In 2021, the Lee Vining Creek main channel's estimated standing crop for Brown Trout equaled 146 kg/ha. The Lee Vining Creek side channel's total Brown Trout standing crop estimate was 22 kg/ha in 2021.

Condition Factors

In 2021, no sample sections had condition factors of Brown Trout 150 to 250 mm in length that exceeded 1.00 (considered fish in average condition). In 2021, the condition factor of Brown Trout 150 to 250 mm in length equaled 0.98 in the MGORD section, 0.96 in the Upper Rush section, 0.93 in the Bottomlands section and 0.94 in Walker Creek. In 2021, the condition factors of Brown Trout 150 to 250 mm in length were 0.94 in the Lee Vining Creek main channel and 0.86 in the Lee Vining Creek side channel.

Relative Stock Densities (RSD)

In the Upper Rush section, the RSD-225 equaled 6 for 2021, the fourth consecutive drop from the record RSD-225 value of 78 in 2017. This decrease was most likely influenced by greater numbers of fish smaller than 225 mm. The RSD-300 value was 1 in 2021. This low RSD-300 value in 2021 was influenced by the higher numbers of fish ≤225 mm caught and also a drop in the numbers of Brown Trout ≥300 mm.

In the Bottomlands section of Rush Creek, the RSD-225 for 2021 equaled 9, the same value as in 2020. As in the Upper Rush section, the Bottomlands 2021 RSD-225 value was influenced by greater numbers of fish smaller than 225 mm. The RSD-300 value was 1 in 2021.

In the MGORD, the RSD-225 value in 2021 was 53, an increase from the previous two years. In 2021, the RSD-300 value was 11, a decrease from the value of 13 in 2020. The RSD-375 value in

2021 was 3, a slight increase from the previous two years. In 2021, a total of 47 Brown Trout ≥300 mm in length were caught, including 12 fish ≥375 mm.

In 2021, RSD values in Lee Vining Creek were generated for the main channel only. The RSD-225 value equaled 3 for 2021, the lowest value recorded for this section. Of the fish ≥150 mm, 97% of them were <225 mm in length. In 2021, no Brown Trout greater than 300 mm in length were captured in the Lee Vining Creek main channel, which resulted in a RSD-300 value of 0.

Introduction

Study Area

Between September 7th and 17th 2021, Los Angeles Department of Water and Power (LADWP) staff and Ross Taylor (the SWRCB fisheries scientist) conducted the annual fisheries monitoring surveys in six reaches along Rush, Lee Vining, and Walker creeks in the Mono Lake Basin. The six reaches were similar in length to those sampled between 2009 and 2020 (Figure 1). Aerial photographs of the sampling reaches are provided in Appendix A.

Hydrology

The 2021 RY was 58% of normal and classified 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 nine years included a Dry-Normal 1 RY 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). Under existing SWRCB orders and the Stream Restoration Flows (SRF), a Dry RY prescribes a Rush Creek summer baseflow of 31 cfs followed by baseflows of 36 cfs from October 1 through March 31. No snowmelt peak flow is required under the SRF for a Dry RY type. However, prior to April 1, 2021, LADWP submitted a Temporary Urgency Change Petition (TUCP) to the SWRCB to implement the Stream Ecosystem Flows (SEF) Dry RY flow regime instead of the SRF. The SEF Dry RY flow regime has a 30 cfs baseflow for the month of April, followed by a spring ascension from 30 to 70 cfs over a 15-day period, followed by snowmelt bench of 70 cfs for 51 days, followed by a slow recession from 70 to 45 cfs and 45 to 27 cfs in July, and then a baseflow of 27 cfs (Figure 2). In Lee Vining Creek, the existing SWRCB orders (SRF) require that the primary peak flow is passed downstream. However, in 2021 LADWP included the Lee Vining Creek in their TUCP to the SWRCB and implemented the diversion rate table and fall/winter baseflows consistent with the recommended SEF (Figure 3).

The 2021 Rush Creek hydrograph at the MGORD generally followed the SEF flows for a Dry RY, with the required spring ascension, followed by a ramp up to approximately a 70 cfs snowmelt bench for 50 days, from May 18th through July 6th (red line on Figure 2). After the snowmelt bench, flows receded down to a summer baseflow by July 24th (red line on Figure 2). The flows upstream of GLR (At Damsite) depicted a range of peaks and drops in Rush Creek flows due to snowmelt runoff, SCE operations and possibly rain-storm peaks (blue line on Figure 2). In past annual reports, a third line is usually included on the Rush Creek hydrograph that depicts the accretions from Parker and Walker creeks added to the MGORD flows, for a “below Narrows” total flow in lower Rush Creek. However, an error was discovered (during the review period of the draft report) in the reporting of the Walker Creek discharge, thus no accurate “below Narrows” flow data were available for this report.

In 2021, multiple, small peaks occurred in Lee Vining Creek above the intake, with a peak of 114 cfs on May 13th (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 22 cfs for the duration of the fisheries sampling (Figure 3). By mid-November, a winter base flow of approximately 16-18 cfs was established in Lee Vining Creek below the intake (Figure 3). A late October rainstorm resulted in a peak flow >100 cfs (Figure 3).

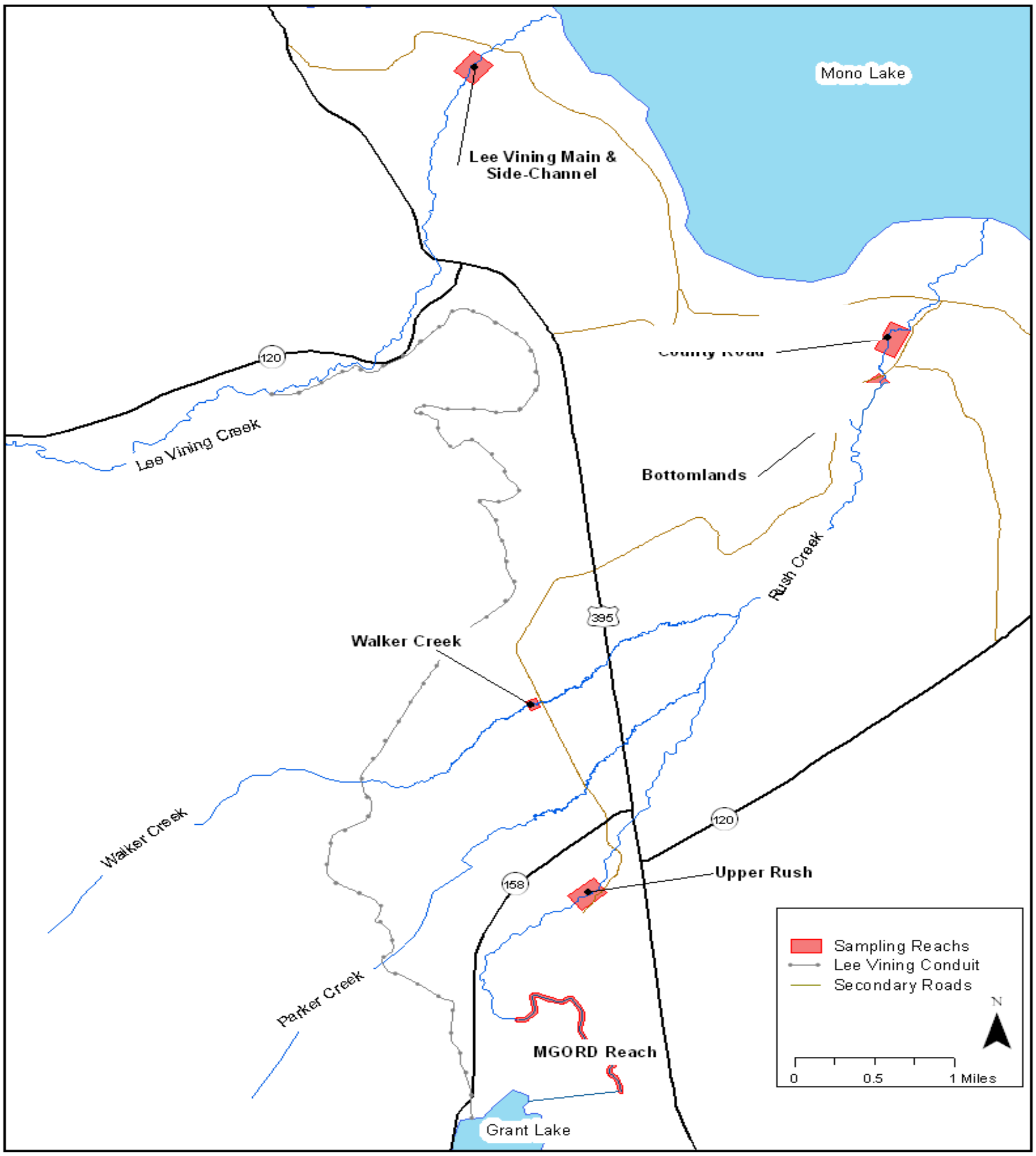

Figure 1. Annual fisheries sampling sites within Mono Basin study area, September 2021.

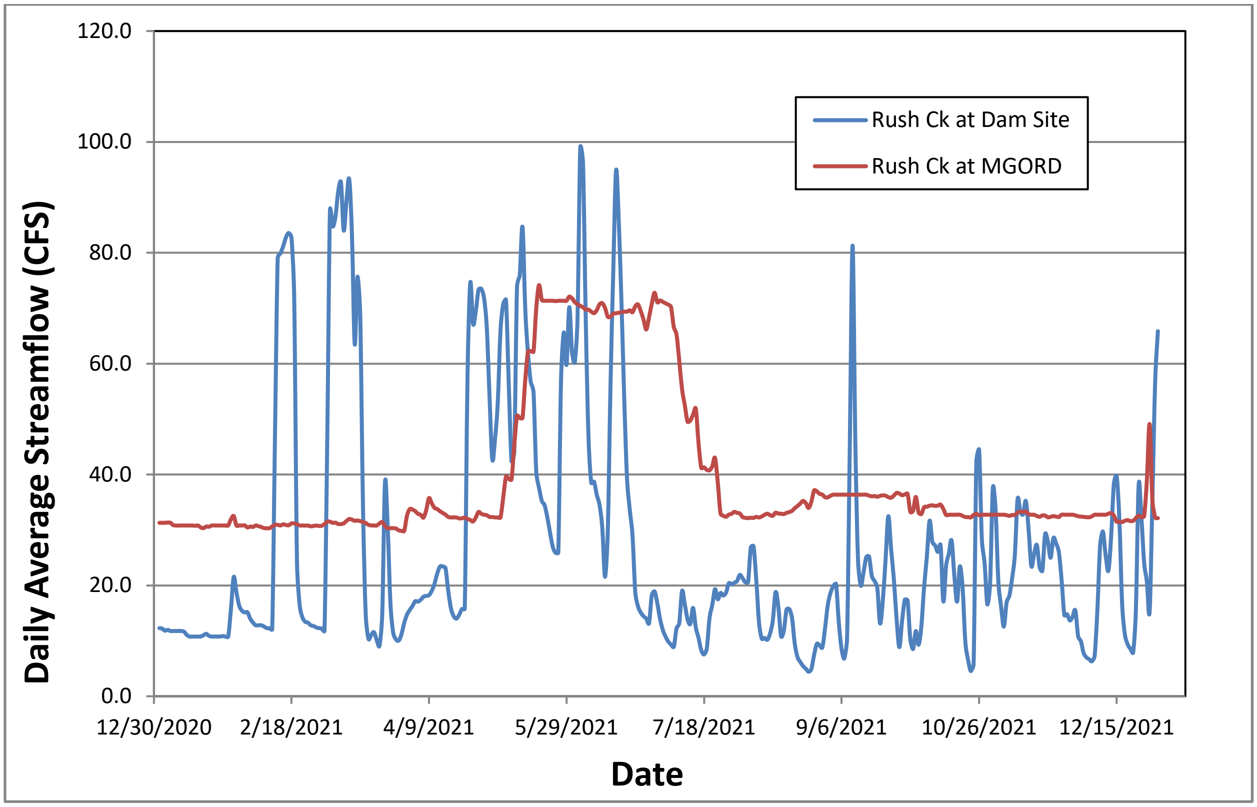

Figure 2. Rush Creek hydrographs between January 1st and December 31st of 2021.

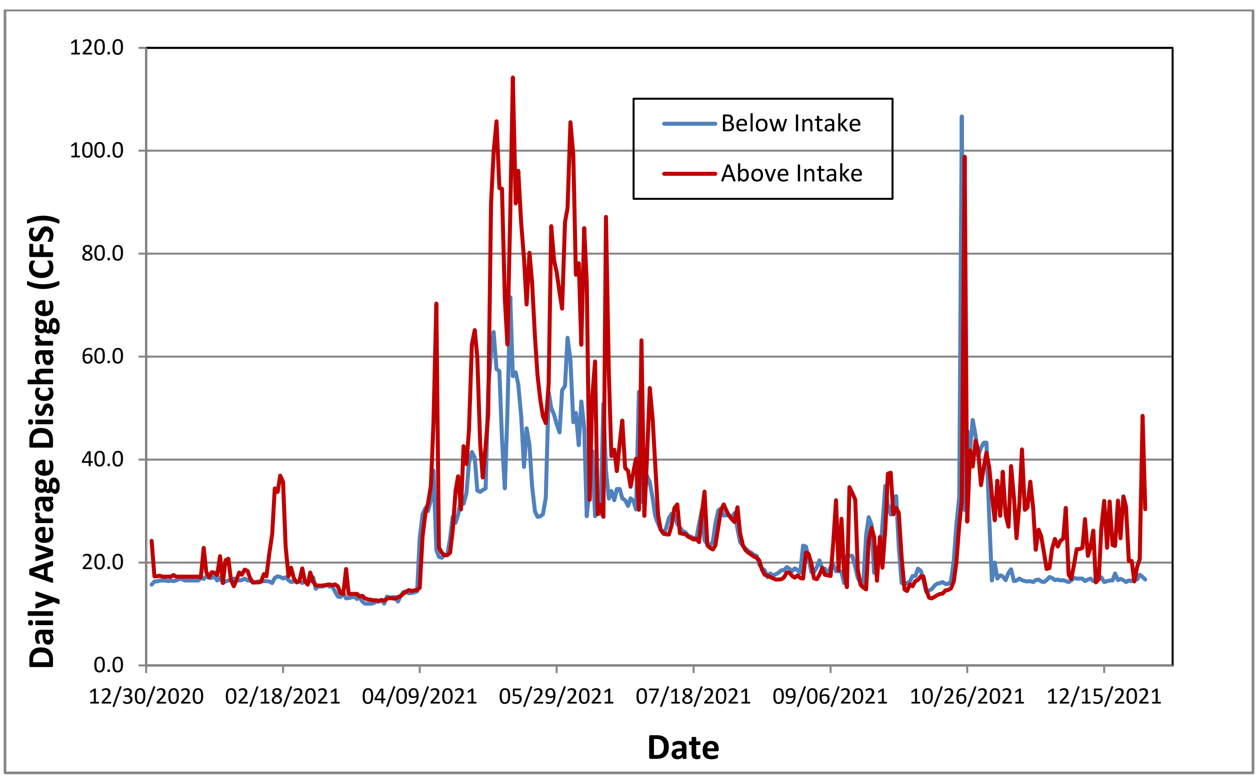

Figure 3. Lee Vining Creek hydrograph between January 1st and December 31st of 2021.

Grant Lake Reservoir

In 2021, storage elevation levels in GLR fluctuated from a high of 7,110.7 ft on January 1st to a low of 7,092.9 ft in late-December (Figure 4). In 2021, GLR dropped from January until late April, filled until early June, and then dropped throughout the remainder of the year (Figure 4).

During the summer months of RY2021, GLR's elevation was 3.0 ft to 7.2 ft above the "low" GLR level 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). However, the 2021 summer water temperature monitoring documented concerningly warm water temperatures with sometimes large diurnal fluctuations, leading to less than favorable conditions for Brown Trout, at all Rush Creek locations downstream of GLR for variable lengths of the summer period, defined as July through September.

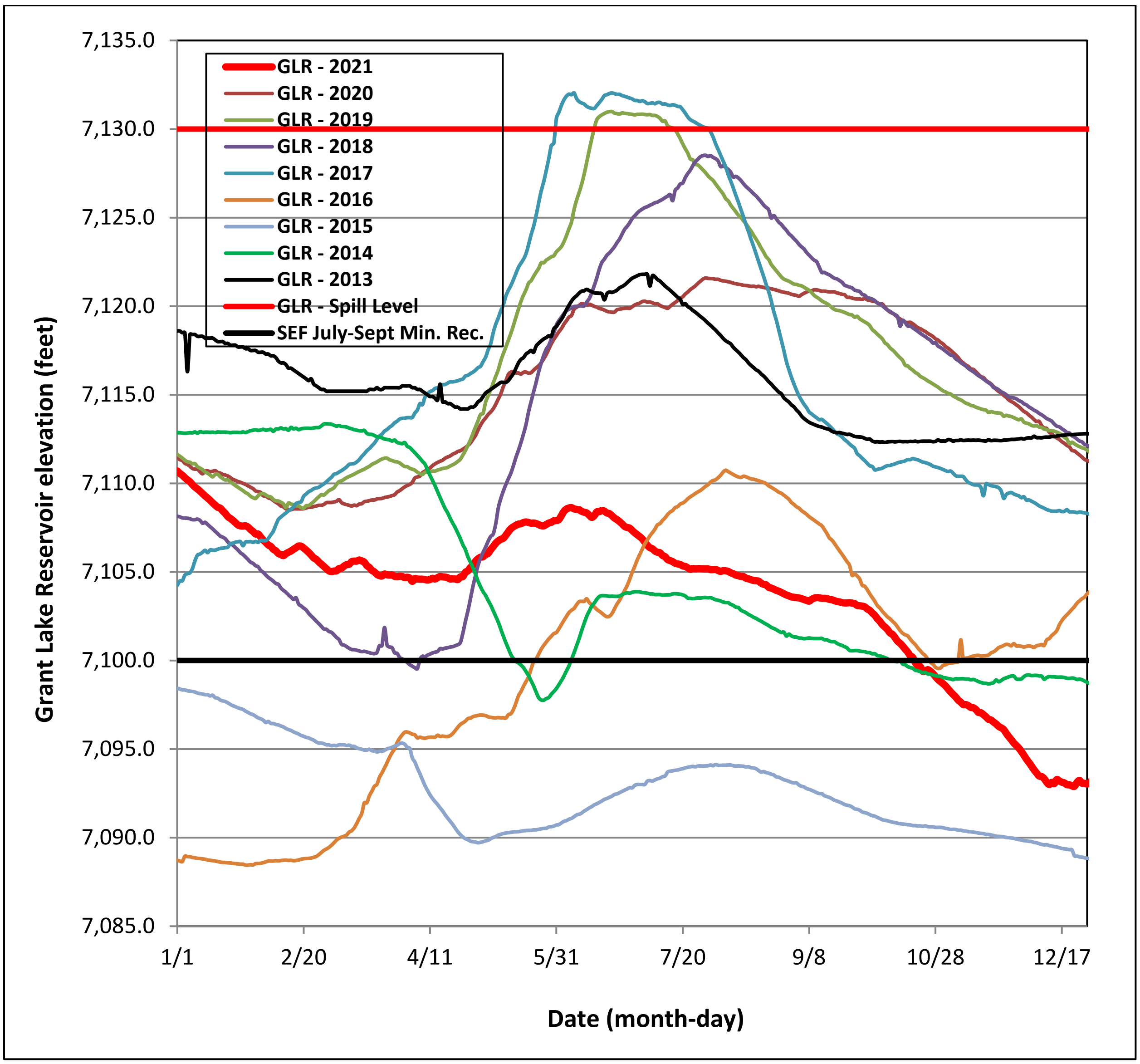

Figure 4. Grant Lake Reservoir's elevation between January 1st and December 31st 2013 - 2021.

Methods

The annual fisheries monitoring was conducted between September 7th and 17th of 2021. 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 Lee Vining Creek side channel and Walker Creek sections.

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. Sticks were used to keep the top of the seine above the water surface. Both ends of the seine net were then tied to bank vegetation to hold it in place.

Equipment used to conduct mark-run electrofishing on Rush Creek included a six-foot plastic barge that contained the Smith-Root© 2.5 GPP electro-fishing system, an insulated cooler, and battery powered aerators. The Smith-Root© 2.5 GPP electro-fishing 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 2021, 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 operator’s 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, 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-24 backpack electrofishers, two to three netters, and one bucket carrier who transported the captured trout. One less netter was available on the recapture-run due to ill and injured crew members.

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. A second safety officer walked the streambank and observed the in-stream operations. 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.

For the Walker Creek and Lee Vining Creek side channel depletions, 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-24 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 a suitable estimate, 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 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 Bottomlands 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 2014-2021 field seasons. Starting in 2017, PIT tags implanted in trout caught in the MGORD were focused primarily on fish up to 250 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 2011). 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.

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 average (Reimers 1963; Blackwell et al. 2000).

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). 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 LADWP personnel from the Bishop Office 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 2021:

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

For the fisheries monitoring program, the year-long data sets were edited to focus on the 2021 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 2020 were provided for comparisons (Table 1).

Table 1. Total length, average wetted width, and total surface area of sample sections in Rush, Lee Vining, and Walker creeks sampled between September 7-17, 2021. Values from 2020 provided for comparisons.

Sample Section	Length (m) 2020	Width (m) 2020	Area (m^2) 2020	Length (m) 2021	Width (m) 2021	Area (m^2) 2021	Area (ha) 2021
Rush – Upper	381	7.8	2,971.8	381	7.4	2,819.4	0.2819
Rush - Bottomlands	437	6.6	2,884.2	437	6.6	2,884.2	0.2884
Rush – MGORD	2,230	7.9	17,617.0	2,230	8.5	18,955.0	1.8955
Lee Vining – Main	255	5.1	1,300.5	255	5.3	1,351.5	0.1352
Lee Vining - Side	175	2.1	367.5	168	2.0	336.0	0.0336
Walker Creek	195	2.4	468.0	210	2.1	441.0	0.0441

Trout Population Abundance

In 2021, a total of 577 Brown Trout ranging in size from 67 mm to 400 mm were captured on the two mark-recapture electrofishing passes in the Upper Rush section; 319 of these fish were caught on the mark-run (Figure 5). For comparison, in 2020 a total of 835 Brown Trout were caught on the mark-run. In 2021, age-0 Brown Trout comprised 28% of the total catch (compared to 56% in 2020 and 62% in 2019). The Upper Rush section supported an estimated 467 age-0 Brown Trout in 2021 compared to 1,868 age-0 Brown Trout in 2020 (a 75% decrease) (Table 2).

In 2021, the 367 Brown Trout captured in the 125-199 mm size class comprised 64% of the total catch in the Upper Rush section (compared to 38% in 2020). The Upper Rush section supported an estimated 586 Brown Trout in the 125-199 mm size class in 2021, compared to 859 fish in 2020 (a 32% decrease) (Table 2).

Brown Trout ≥200 mm in length comprised 8% of the Upper Rush total catch in 2021 (compared to 6% in 2020). In 2021, Upper Rush supported an estimated 63 Brown Trout ≥200 mm in length compared to an estimate of 93 fish in 2020 (a 32% decrease) (Table 2). In 2021, four Brown Trout ≥300 mm in length were captured in the Upper Rush section (Figure 5).

A total of 69 Rainbow Trout were captured in the Upper Rush section comprising 11% of the section's total catch in 2021; Rainbow Trout also comprised 11% of the total catch in 2020. The 69 Rainbow Trout ranged in length from 66 mm to 250 mm and 40 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 population estimates (Table 2). In 2021, the Upper Rush section supported an estimated 77 Rainbow Trout <125 mm in length (253 in 2020 and 418 in 2019), an estimated 25 Rainbow Trout 125-199 mm in length (119 in 2020), and an estimated nine Rainbow Trout ≥200 in length (also nine in 2020) (Table 2).

In 2021, a total of 308 Brown Trout ranging in size from 64 mm to 440 mm were captured on the two mark-recapture electrofishing passes in the Bottomlands section of Rush Creek; 167 of these fish were caught on the mark-run (Figure 7). For comparison, in 2020 a total of 384 Brown Trout were caught on the mark-run. Brown Trout <125 mm in length comprised 34% of the total catch in 2021 versus 45% of the total catch in 2020. The Bottomlands section supported an estimated 677 Brown Trout <125 mm in length in 2021 versus 662 fish in 2020 (an 8% increase). Although trout in the <125 mm size category have typically been considered age-0 fish; in 2021 there was a strong break in the length-frequency data that suggests age-0 fish topped out at 100 to 110 mm and age-1 fish were as small as 120 mm (Figure 7).

Brown Trout 125-199 mm in length comprised 55% of the total catch in the Bottomlands section in 2021 versus 45% of the total catch in 2020. This section supported an estimated 345 Brown Trout 125-199 mm in length in 2021 compared to 364 fish in 2020 (a 5% decrease).

Brown Trout ≥200 mm in length comprised of 11% of the total catch in 2021 (10% in 2020) with the largest trout 440 mm in length (Figure 7). The Bottomlands section supported an estimated 41 Brown Trout ≥200 mm in 2021 compared to 67 trout in 2020 (a 39% decrease).

In 2021, one Rainbow Trout was caught in the Bottomlands section of Rush Creek. In comparison, five Rainbow Trout were caught in 2020, 10 Rainbow Trout were caught in 2019 and no Rainbow Trout were caught in 2018 within the Bottomlands section.

Within the MGORD section of Rush Creek a total of 556 Brown Trout were captured in 2021, with 267 fish caught on the mark-run. In comparison, 431 Brown Trout caught in one pass in 2020. In 2021, these Brown Trout ranged in size from 76 mm to 542 mm (Figure 8). A total of 123 Brown Trout <125 mm in length were captured in 2021, which comprised 22% of the total catch of Brown Trout (105 age-0 fish were caught in 2020) (Figure 8). The MGORD section supported an estimated 677 Brown Trout <125 mm in length in 2021 (Table 2).

In 2021, a total of 82 Brown Trout 125-199 mm in length were caught during the mark-recapture sampling and comprised 15% of the total Brown Trout catch in the MGORD section (116 fish were caught in 2020's single pass). The MGORD supported an estimated 154 Brown Trout in the 125-199 mm size class in 2021, compared to an estimate of 446 fish in 2020, a decrease of 66% (Table 2).

In 2021, a total of 351 Brown Trout ≥200 mm in length were caught during the mark-recapture sampling and comprised of 63% of the total catch in the MGORD section (210 fish were caught in 2020's single pass). The MGORD supported an estimated 625 Brown Trout in the ≥200 mm size class in 2021, compared to 583 fish in 2020, an increase of 7% (Table 2).

In 2021, 47 Brown Trout ≥300 mm were captured in the MGORD (43 fish ≥300 mm were captured during the single pass made in 2020). Twelve Brown Trout ≥375 mm in length were captured in 2021 (compared to six fish in 2020, four fish in 2019, 15 fish in 2018, 11 fish in 2017 and 20 fish in 2016), seven of these fish were >400 mm in length and two of these fish were >500 mm in length (Figure 8).

In 2021, 15 Rainbow Trout were captured in the MGORD section (Figure 9). In the previous eight years, the Rainbow Trout catch in the MGORD has ranged from zero to 40 fish. Most of the Rainbow Trout captured in 2021 appeared to be of natural origin, with several larger fish exhibiting signs of hatchery origin.

For the past 16 sampling years, electrofishing passes through the MGORD have produced the following total catch values (all size classes of Brown and Rainbow Trout):

- 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 2. Rush Creek mark-recapture estimates for 2021 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

Stream / Section / Species / Date	Size Class (mm)	M	C	R	Morts	**Estimate**	S.E.
		Mark - Recapture Estimate					
Rush Creek							
Upper Rush - BNT							
9/7/2021 & 9/14/2021							
	0 - 124 mm	79	100	15	7	**467**	91
	125 - 199 mm	212	241	86	3	**586**	38
	≥200 mm	28	32	14	0	**63**	8
Upper Rush - **RBT**							
9/7/2021 & 9/14/2021							
	0 - 124 mm	24	22	6	2	**77**	19
	125 - 199 mm	16	13	8	0	**25**	12
	≥200 mm	5	7	4	0	**9**	1
Bottomlands - BNT							
9/8/2021 & 9/15/2021							
	0 - 124 mm	52	63	4	0	**677**	253
	125 - 199 mm	96	99	27	0	**345**	46
	≥200 mm	19	18	8	0	**41**	7
MGORD - BNT							
9/9/2021 & 9/16/2021							
	0 - 124 mm	40	81	4	6	**677**	249
	125 - 199 mm	39	57	14	0	**154**	26
	≥200 mm	180	237	68	2	**625**	50
Lee Vining Creek							
Main Channel - BNT							
9/10/2021 & 9/17/2021							
	0 - 124 mm	23	23	9	0	**57**	10
	125 - 199 mm	144	149	53	0	**402**	34
	≥200 mm	25	25	12	0	**51**	7

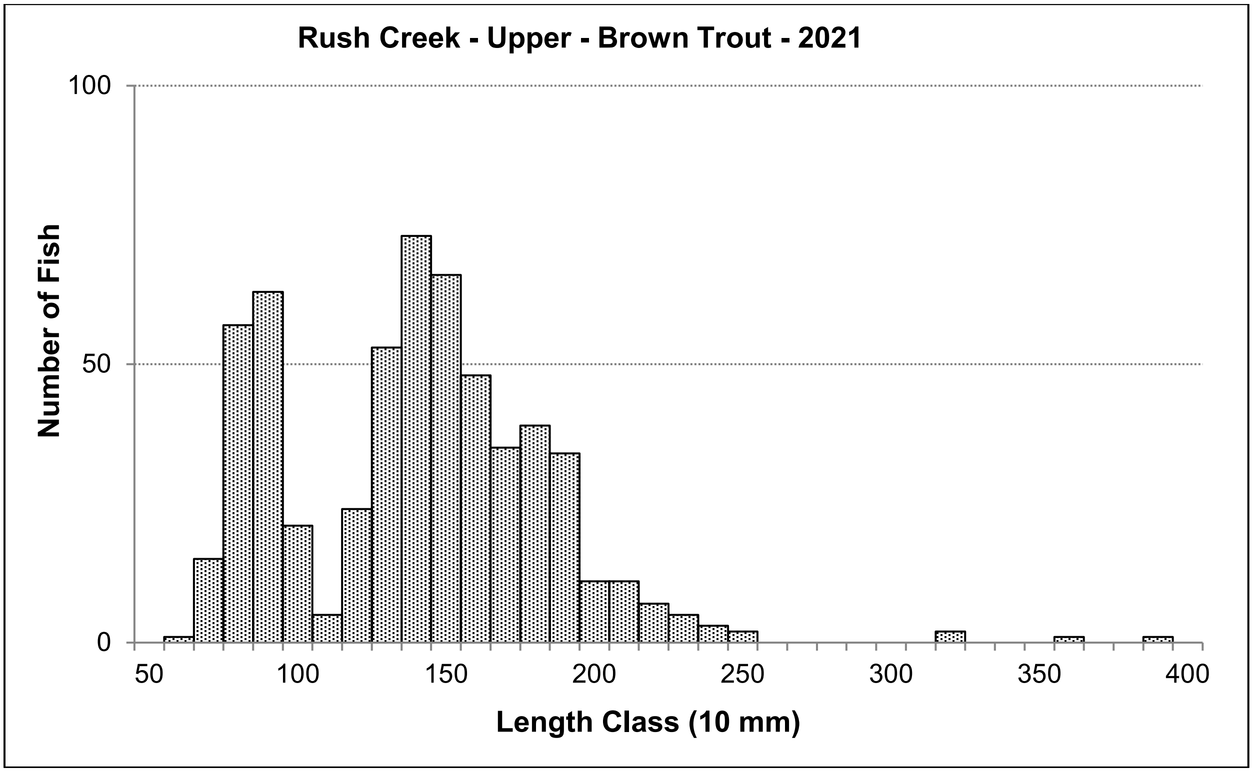

Figure 5. Length-frequency histogram of Brown Trout captured in Upper Rush, September 7th and 14th, 2021.

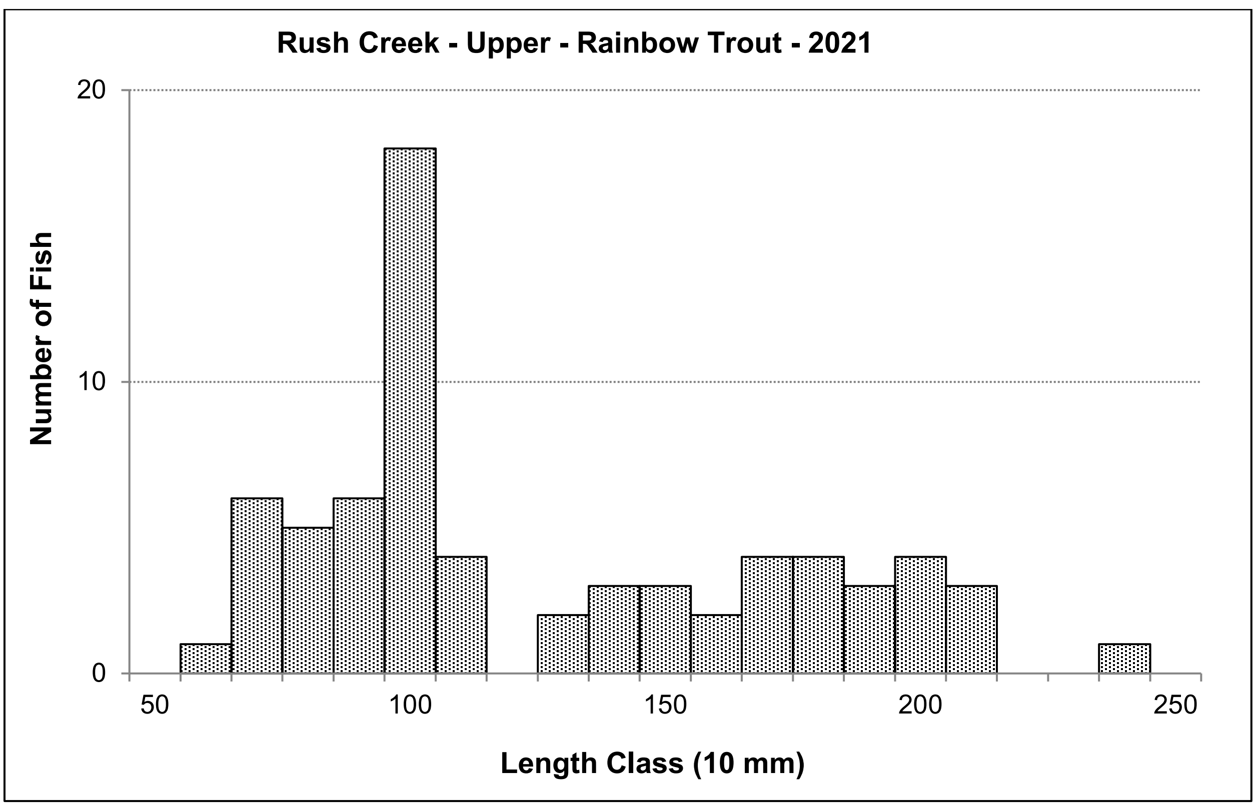

Figure 6. Length-frequency histogram of Brown Trout captured in Upper Rush, September 7th and 14th, 2021.

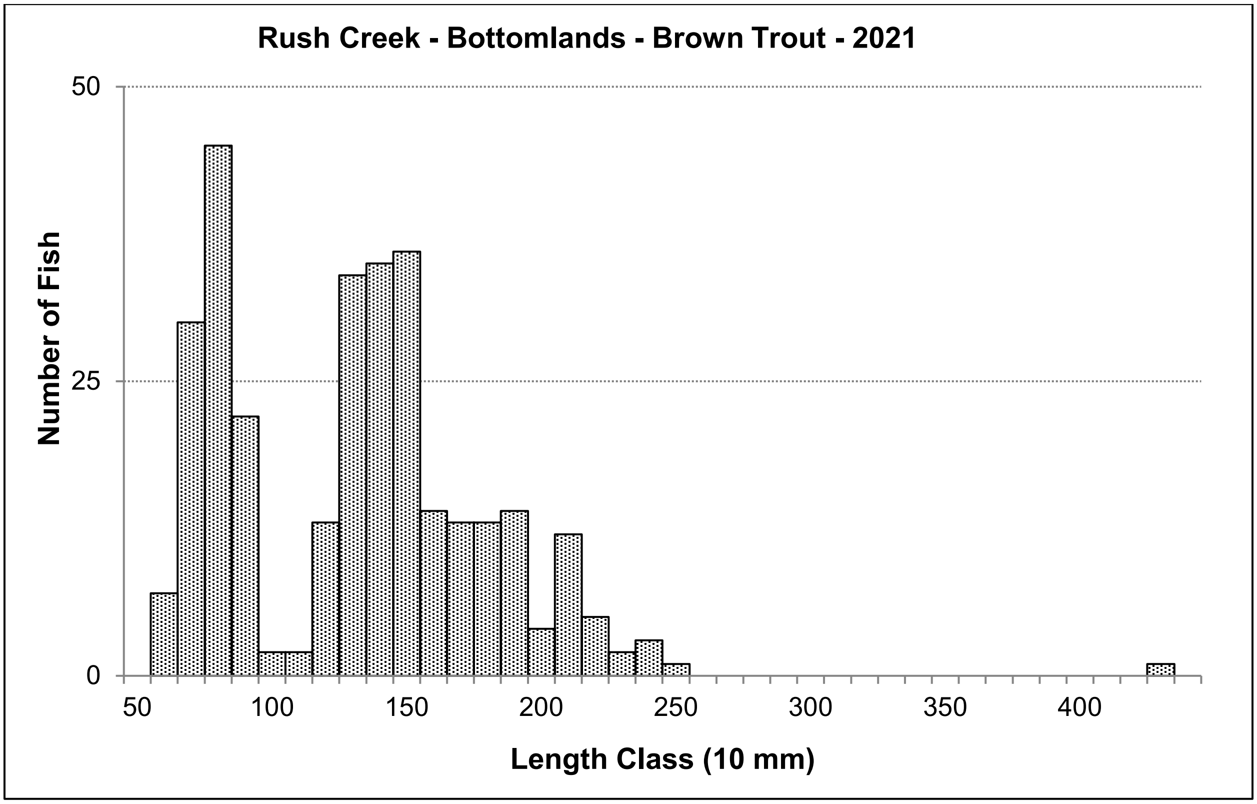

Figure 7. Length-frequency histogram of Brown Trout captured in the Bottomlands section of Rush Creek, September 8th and 15th, 2021.

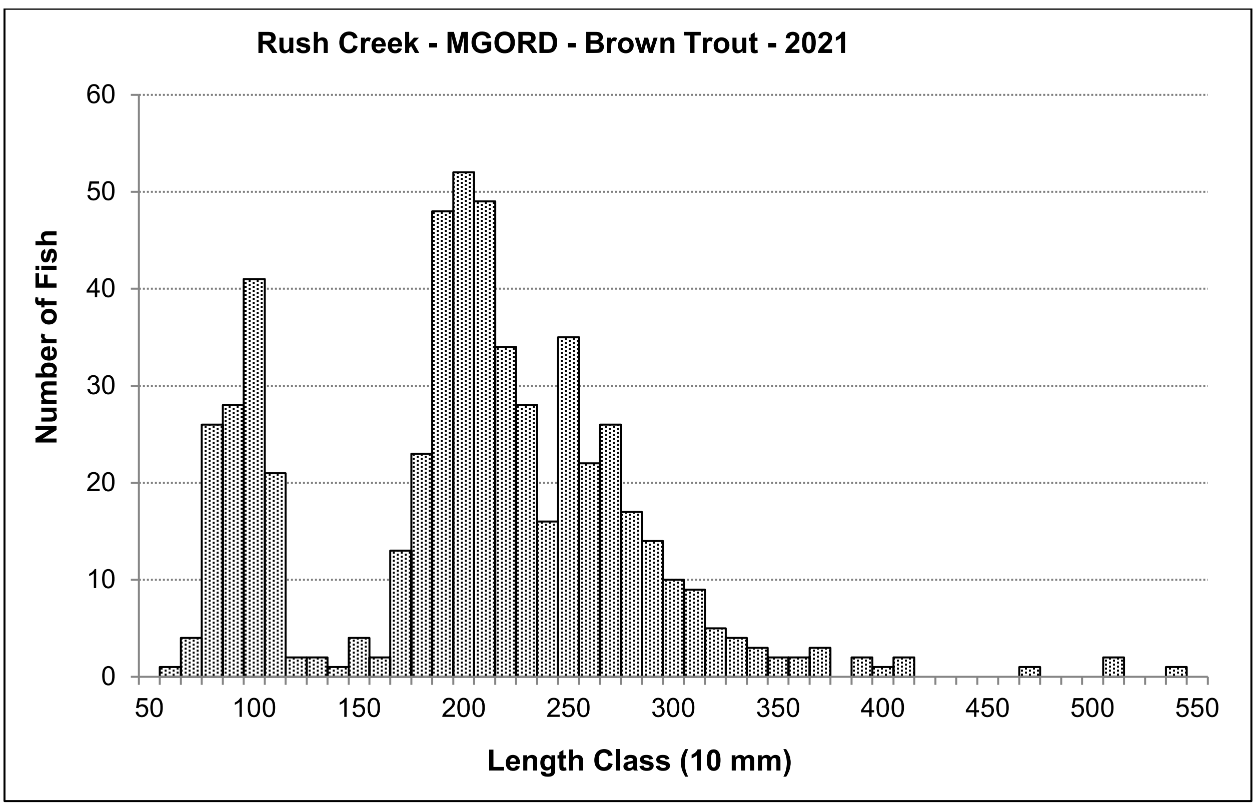

Figure 8. Length-frequency histogram of Brown Trout captured in the MGORD section of Rush Creek, September 9th and 16th, 2021.

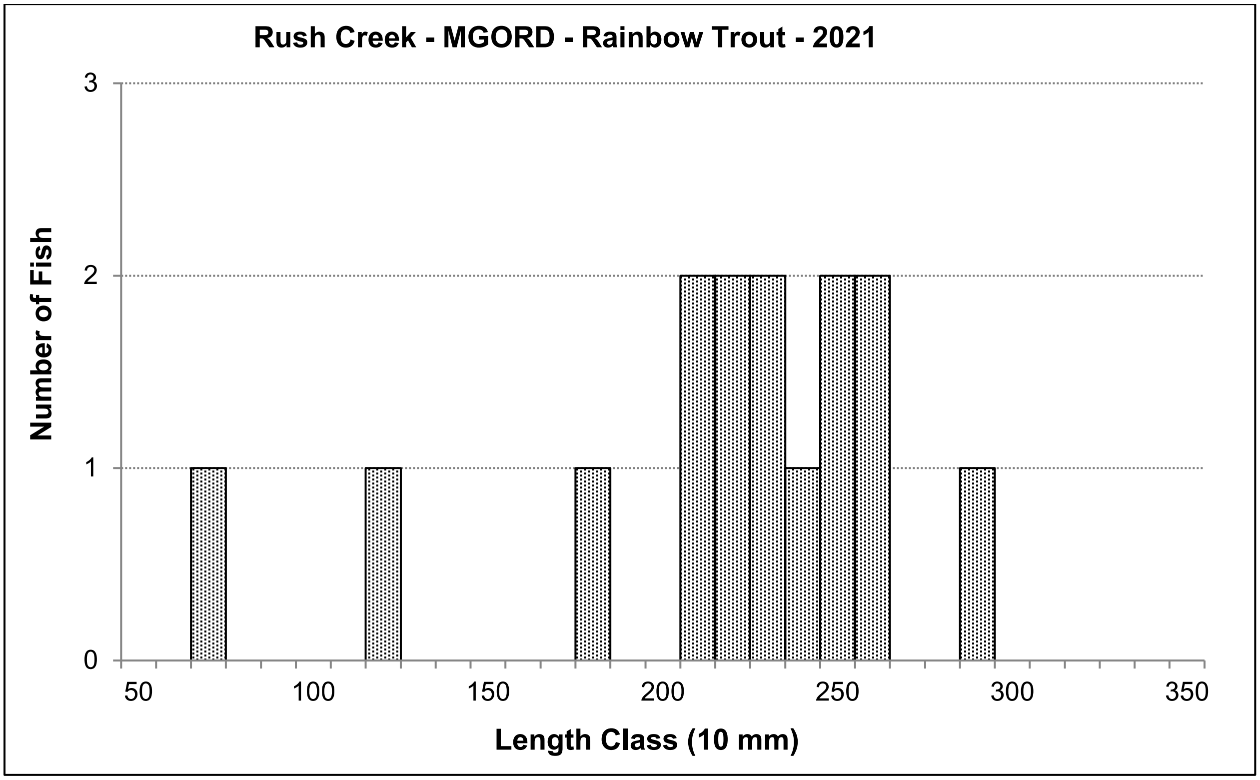

Figure 9. Length-frequency histogram of Rainbow Trout captured in the MGORD section of Rush Creek, September 9th and 16th, 2021.

Lee Vining Creek

In 2021, a total of 319 trout were captured on the mark-recapture electrofishing passes made in the Lee Vining Creek main channel section versus 263 trout in 2020 on a single pass (Table 2). Most (315 fish) of the trout captured in 2021 were Brown Trout. In 2021, Brown Trout ranged in size from 68 mm to 251 mm in length (Figure 10). Fish <125 mm in length comprised 12% of the total Brown Trout catch in 2021, compared to 60% in 2020, 63% in 2019 and 62% in 2018. In 2021, the Lee Vining Creek's main channel section supported an estimated 57 Brown Trout in the <125 mm size class, compared to an estimated 449 Brown Trout in 2020, an 87% decrease (Table 2).

In 2021, Brown Trout 125-199 mm in length comprised 76% of the total Brown Trout catch in Lee Vining Creek's main channel section (versus 33% in 2020). This section supported an estimated 402 Brown Trout 125-199 mm in length in 2021 (Table 2) compared to 171 fish in 2020 (a 135% increase).

In 2021, the population estimate of Brown Trout ≥200 mm in Lee Vining Creek's main channel was 51 fish (versus 24 fish in 2020 and 48 fish in 2019) (Table 2). No Brown Trout captured in 2021 were >300 mm in length (Figure 10).

No population estimate was generated for Rainbow Trout due to insufficient numbers of fish, with only four captured during the mark-recapture electrofishing passes made in 2021. These fish were 65, 81, 212 and 332 mm in length.

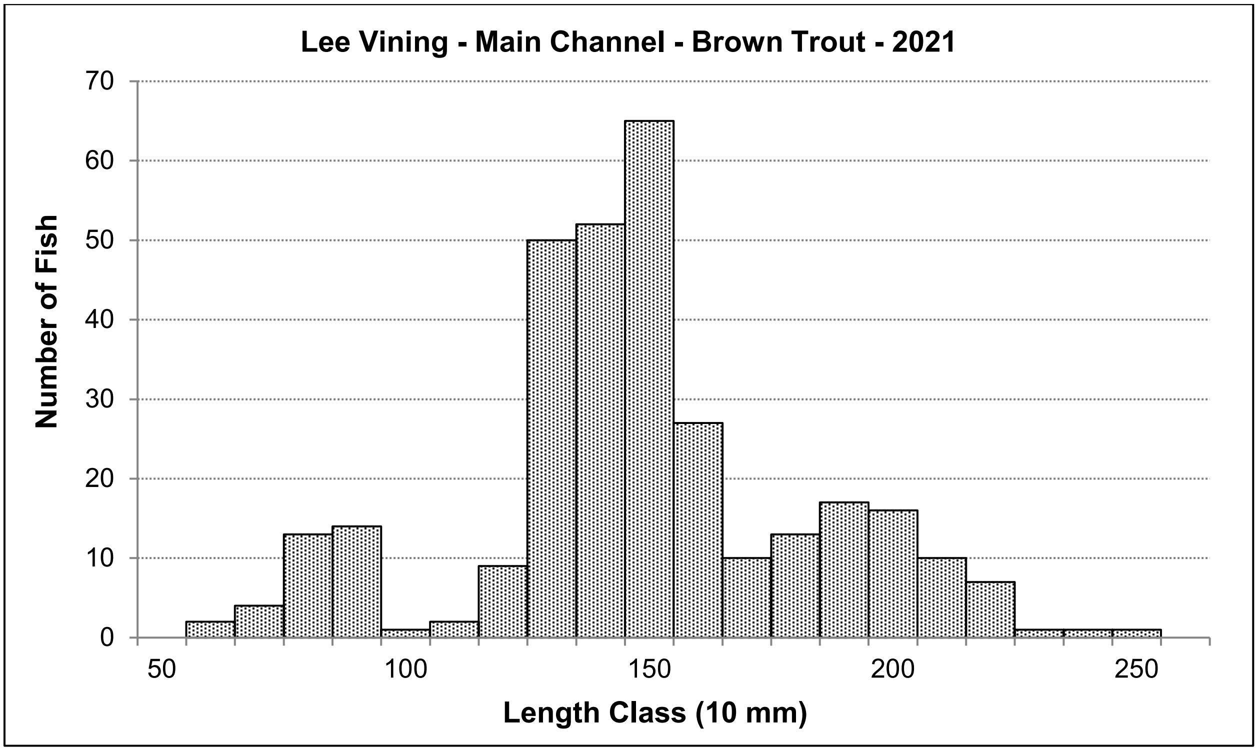

Figure 10. Length-frequency histogram of Brown Trout captured in the main channel section of Lee Vining Creek, September 10th and 17th, 2021.

In the Lee Vining Creek side channel, 33 Brown Trout were captured in two electrofishing passes made during the 2021 sampling (Table 3). Twenty-one age-0 fish were captured (<125 mm) in 2021 (Figure 11). The estimates for the three size classes equaled the catch numbers and the probability of captures ranged from 0.80 to 1.00 (Table 3). No Rainbow Trout were captured in the side channel in 2021. This was the 13th consecutive year that no age-0 Rainbow Trout were captured in the Lee Vining Creek side channel and the 11th consecutive year that no age-1 and older Rainbow Trout were captured in the side channel.

Walker Creek

In 2021, 356 Brown Trout were captured in two electrofishing passes in the Walker Creek section (362 caught in 2020, 278 caught in 2019 and 175 caught in 2018) (Table 3). Two hundred eight of these captured fish, or 49%, were age-0 fish ranging in size from 56 mm to 107 mm in length (Figure 12). The break in the length-frequency histogram (Figure 12) and one PIT tag recapture suggest that age-1 trout were as small as 115 mm in 2021. The 2021 estimated population of Brown Trout <125 mm in length was 227 fish, compared to 180 fish in 2020 (Table 3). For trout <125 mm in length, the estimated probability of capture during 2021 was 0.83 (Table 3).

Brown Trout in the 125-199 mm size class (119 fish) accounted for 33% of Walker Creek's total catch in 2021. The 2021 population estimate for Brown Trout in the 125-199 mm size class was 119 trout (a 14% decrease from the 2020 estimate) with an estimated probability of capture of 0.94 (Table 8).

Brown Trout ≥200 mm in length (16 fish caught) accounted for 5% of the total catch in 2021. The 2021 population estimate for this size class was 16 Brown Trout with a probability of capture of 1.00 because all 16 fish were caught on the first pass (Table 3). The largest Brown Trout captured in Walker Creek in 2021 was 230 mm in length (Figure 12).

In 2021, two Rainbow Trout were captured in Walker Creek (183 and 186 mm in length).

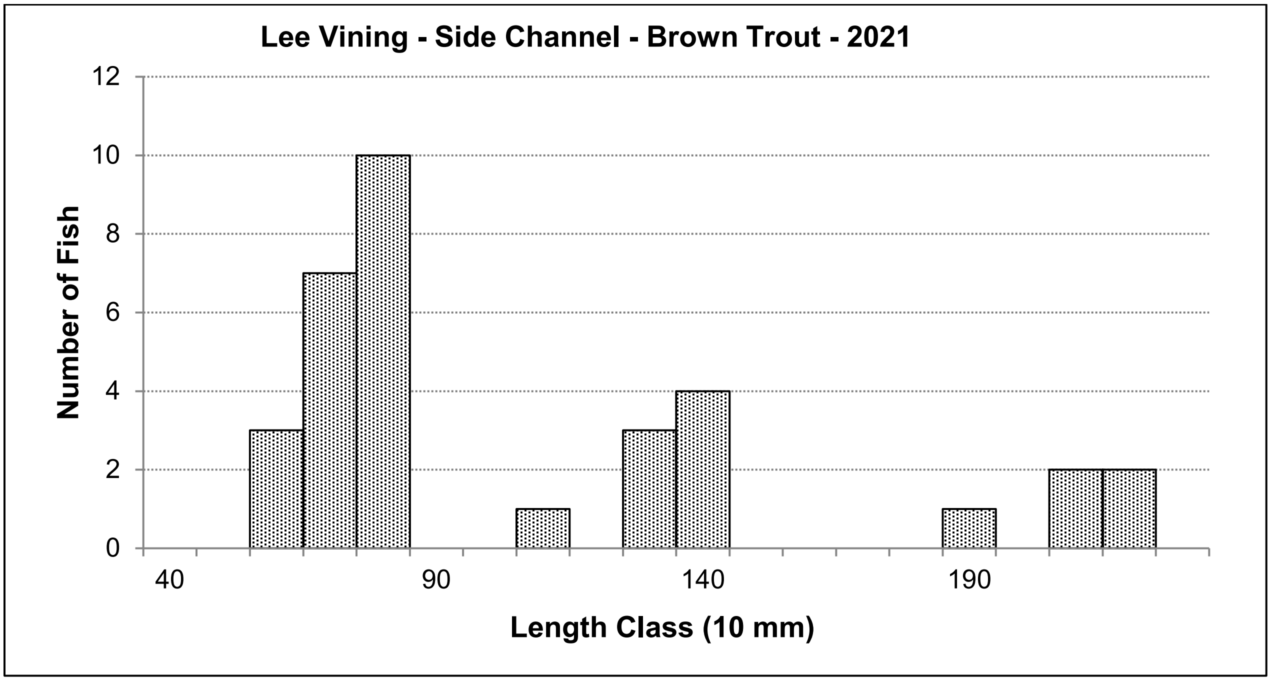

Figure 11. Length-frequency histogram of Brown Trout captured in the side channel section of Lee Vining Creek, September 10th, 2021.

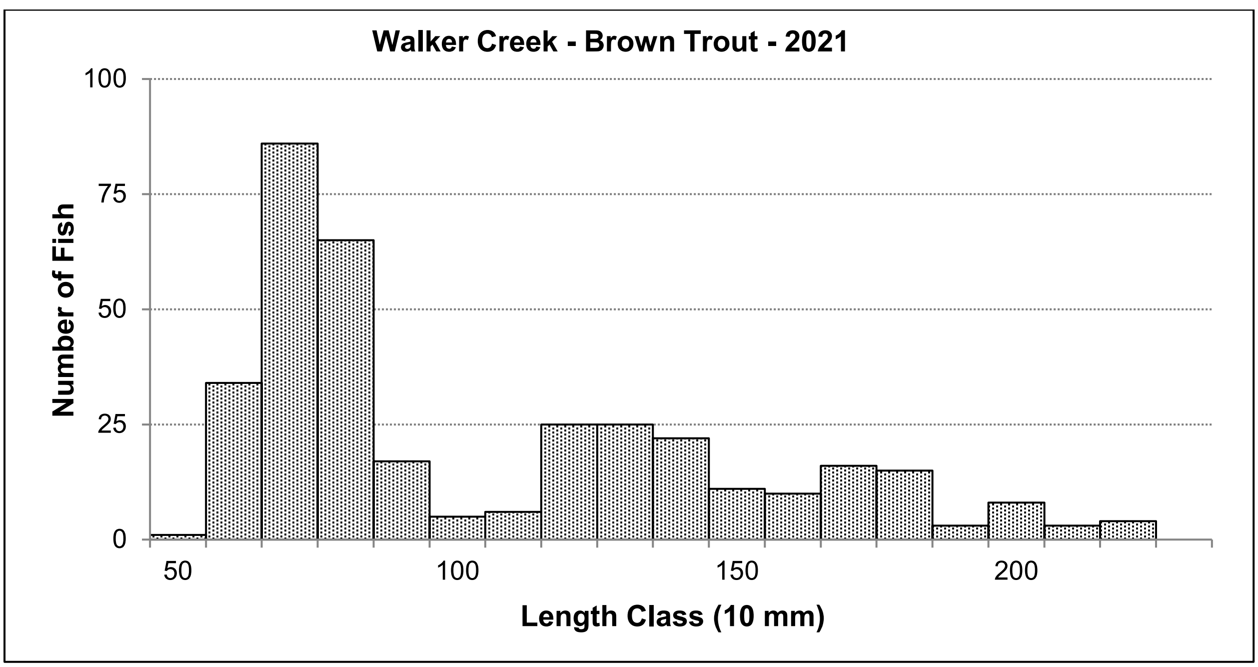

Figure 12. Length-frequency histogram of Brown Trout captured in Walker Creek, September 11th, 2021.

Table 3. Depletion estimates made in the side channel section of Lee Vining Creek and Walker Creek during September 2021 showing number of trout captured in each pass, estimated number, probability of capture (P.C.) by species and size class.

Stream - Section Date **Species**	Size Class (mm)	Removals	Removal Pattern	**Estimate**	P.C.
Lee Vining Creek- Side Channel - 9/10/2021					
Brown Trout					
	0 - 124 mm	3	17 4	**21**	0.84
	125 - 199 mm	3	6 2	**8**	0.80
	200 + mm	3	4 0	**4**	1.00
Walker Creek - above old Hwy 395 - 9/11/2021					
Brown Trout					
	0 - 124 mm	2	189 32	**227**	0.83
	125 - 199 mm	2	112 7	**119**	0.94
	200 + mm	2	16 0	**16**	1.00

Catch of Rainbow Trout in Rush and Lee Vining Creeks

Beginning with the 2008 annual report through the 2016 annual report, we only reported catch summaries for Rainbow Trout in Rush Creek and did not attempt to estimate their populations. This decision was made because Rainbow Trout usually accounted for less than 5% of Rush Creek's total catch. However, since the 2017 sampling season, Rainbow Trout have comprised 10% to 18% of the total catch in Rush Creek, with sufficient numbers recaptured to generate population estimates for most of the size classes in most of the past four sampling seasons.

For the 2018 sampling, Rainbow Trout comprised 17.8% of the total catch in the Upper Rush section (168 Rainbow Trout/944 total trout). Nearly 85% of these Rainbow Trout were age-0 fish and most of the larger fish appeared to be naturally-produced, thus for 2018, Rainbow Trout were included in generating biomass estimates for the Upper Rush section. This substantial increase in age-0 Rainbow Trout may have occurred due to the recent, record low numbers of Brown Trout. In 2019, numerous Rainbow Trout were captured in the Upper Rush section and comprised 15% of the total catch. Age-0 fish comprised 66% of the Rainbow Trout caught and age-1 fish comprised another 30% of the Rainbow Trout caught in 2019 and sufficient numbers were caught on both the mark and recapture runs to generate unbiased population estimates. In 2020, Rainbow Trout comprised 10.7% of the total catch in the Upper Rush section and catch efficiencies from the previous two years were used to generate 2020 population estimates. In 2021, Rainbow Trout comprised 11% of the total catch in the Upper Rush section and sufficient numbers were caught on both the mark and recapture runs to generate population estimates.

Between 1999 and 2012 Rainbow Trout numbers in Lee Vining Creek were variable, generally increasing during drier RY types and decreasing during wetter years. However, since 2012 the annual catch of Rainbow Trout in Lee Vining Creek has dropped steadily and dramatically. In 2012, a total of 235 Rainbow Trout were captured, including 226 age-0 fish. In 2013, 127 Rainbow Trout were captured (26 were age-0 fish), followed by 57 rainbows in 2014 (six were age-0 fish), 20 rainbows in 2015 (no age-0 fish), seven rainbows in 2016 (no age-0 fish), no rainbows in 2017, nine rainbows in 2018, four rainbows in 2019, two rainbows in 2020 and four rainbows in 2021. This large drop in Rainbow Trout numbers has occurred during the time period when CDFW shifted to stocking sterile catchable Rainbow Trout. We suggested that in years prior to 2012, supplementation of the Rainbow Trout population with reproductively viable hatchery Rainbow Trout originating from CDFW stocking (upstream of LADWP's point of diversion), and their successful spawning, probably, to a large degree, supported the Lee Vining Creek Rainbow Trout population (Taylor 2019).

Prior to 2012, Rainbow Trout historically encompassing a large portion (10-40%) of the Lee Vining Creek trout population and an effort was made to generate density and biomass values using the available data. In years when adequate numbers of Rainbow Trout have been captured, statistically valid density and biomass estimates have been generated. In years when less than adequate numbers of Rainbow Trout have been captured, catch numbers have been used to generate density and biomass estimates. An unbiased estimate of age-0 Rainbow Trout in Lee Vining Creek was last made in 2013 and 2015 was the last year that sufficient numbers of age-1+ Rainbow Trout were caught to generate an unbiased estimate of fish in the 125-199 mm size class. Since 2016, we have resorted to only reporting the low numbers of Rainbow Trout captured in Lee Vining Creek.

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 4). Slopes of these relationships were near 3.0 indicating isometric growth, which was assumed to compute fish condition factors, was reasonable.

Table 4. 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 2021 regression equations are in ***bold*** type.

Section	Year	N	Equation	r^2	P
Bottomlands	***2021***	***205***	***$Log_{10}(WT) = 3.0091*Log_{10}(L) – 5.05262$***	***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	$Log_{10}(WT) = 2.7997*Log_{10}(L) – 4.591$	0.98	<0.01
	2012	495	$Log_{10}(WT) = 2.8149*Log_{10}(L) – 4.6206$	0.98	<0.01
	2011	361	$Log_{10}(WT) = 2.926*Log_{10}(L) – 4.858$	0.99	<0.01
	2010	425	$Log_{10}(WT) = 2.999*Log_{10}(L) – 5.005$	0.99	<0.01
	2009	511	$Log_{10}(WT) = 2.920*Log_{10}(L) – 4.821$	0.99	<0.01
	2008	611	$Log_{10}(WT) = 2.773*Log_{10}(L) – 4.524$	0.99	<0.01
Upper Rush	***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	$Log_{10}(WT) = 2.9114*Log_{10}(L) – 4.816$	0.99	<0.01

Table 4 (continued).

Section	Year	N	Equation	r^2	P
Upper Rush	2012	554	$Log_{10}(WT) = 2.8693*Log_{10}(L) – 4.721$	0.99	<0.01
	2011	547	$Log_{10}(WT) = 3.006*Log_{10}(L) – 5.014$	0.99	<0.01
	2010	420	$Log_{10}(WT) = 2.995*Log_{10}(L) – 4.994$	0.99	<0.01
	2009	612	$Log_{10}(WT) = 2.941*Log_{10}(L) – 4.855$	0.99	<0.01
	2008	594	$Log_{10}(WT) = 2.967*Log_{10}(L) – 4.937$	0.99	<0.01
	2007	436	$Log_{10}(WT) = 2.867*Log_{10}(L) – 4.715$	0.99	<0.01
	2006	485	$Log_{10}(WT) = 2.99*Log_{10}(L) – 4.98$	0.99	<0.01
	2005	261	$Log_{10}(WT) = 3.02*Log_{10}(L) – 5.02$	0.99	<0.01
	2004	400	$Log_{10}(WT) = 2.97*Log_{10}(L) – 4.94$	0.99	<0.01
	2003	569	$Log_{10}(WT) = 2.96*Log_{10}(L) – 4.89$	0.99	<0.01
	2002	373	$Log_{10}(WT) = 2.94*Log_{10}(L) – 4.86$	0.99	< 0.01
	2001	335	$Log_{10}(WT) = 2.99*Log_{10}(L) – 4.96$	0.99	< 0.01
	2000	309	$Log_{10}(WT) = 3.00*Log_{10}(L) – 4.96$	0.98	< 0.01
	1999	317	$Log_{10}(WT) = 2.93*Log_{10}(L) – 4.84$	0.98	< 0.01
MGORD	***2021***	***498***	$\boldsymbol{Log_{10}(WT) = 2.9447*Log_{10}(L) – 4.8871}$	***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	$Log_{10}(WT) = 2.8567*Log_{10}(L) – 4.692$	0.98	<0.01
	2012	795	$Log_{10}(WT) = 2.9048*Log_{10}(L) – 4.808$	0.99	<0.01
	2011	218	$Log_{10}(WT) = 2.917*Log_{10}(L) – 4.823$	0.98	<0.01
	2010	694	$Log_{10}(WT) = 2.892*Log_{10}(L) – 4.756$	0.98	<0.01
	2009	689	$Log_{10}(WT) = 2.974*Log_{10}(L) – 4.933$	0.99	<0.01
	2008	862	$Log_{10}(WT) = 2.827*Log_{10}(L) – 4.602$	0.98	<0.01
	2007	643	$Log_{10}(WT) = 2.914*Log_{10}(L) – 4.825$	0.98	<0.01
	2006	593	$Log_{10}(WT) = 2.956*Log_{10}(L) – 4.872$	0.98	<0.01
	2004	449	$Log_{10}(WT) = 2.984*Log_{10}(L) – 4.973$	0.99	<0.01
	2001	769	$Log_{10}(WT) = 2.873*Log_{10}(L) – 4.719$	0.99	<0.01
	2000	82	$Log_{10}(WT) = 2.909*Log_{10}(L) – 4.733$	0.98	<0.01

Condition factors of Brown Trout 150 to 250 mm in length in 2021 decreased from 2020 values in three sections, increased in two sections and remained the same in one section (Figures 13 and 14). In 2021, no sections had Brown Trout condition factors ≥1.00, thus all condition factors were less than average (Figures 13 and 14). This was the second consecutive year that no sections supported Brown Trout with condition factors ≥1.00.

Brown Trout in the Upper Rush section had a condition factor of 0.96 in 2021 a slight increase from 0.95 in 2020 (Figure 13). The Upper Rush section has had Brown Trout condition factors ≥1.00 in 10 of 22 sampling seasons (Figure 13).

Brown Trout in the Bottomlands section of Rush Creek had a condition factor of 0.93 in 2021, a decrease from 0.95 in 2020 (Figure 13). In 14 years of sampling, the Bottomlands section has failed to generate a Brown Trout condition factor ≥1.00 (Figure 13).

The MGORD's 2021 Brown Trout condition factor was 0.98, the same value as in 2020 (Figure 13). In 2021, condition factors for larger Brown Trout in the MGORD were also computed: fish ≥300 mm had a condition factor of 0.94 (0.93 in 2020) and fish ≥375 mm had a condition factor of 0.97 (1.04 in 2020).

In 2021, the condition factor for Brown Trout in Lee Vining Creek's main channel was 0.94 and in the side channel the condition factor was 0.86 (Figure 14). The main channel's 2021 value was the third straight year that Brown Trout condition factors were less than 1.00 (Figure 14). For the eleventh year in a row, no age-1+ Rainbow Trout were captured in the Lee Vining Creek side channel. In 2021, 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 0.94 in 2021, a decrease from 0.96 in 2020 and 0.98 in 2019 (Figure 13). Brown Trout condition factors in Walker Creek have been ≥1.00 in 12 of the 22 sampling years (Figure 13).

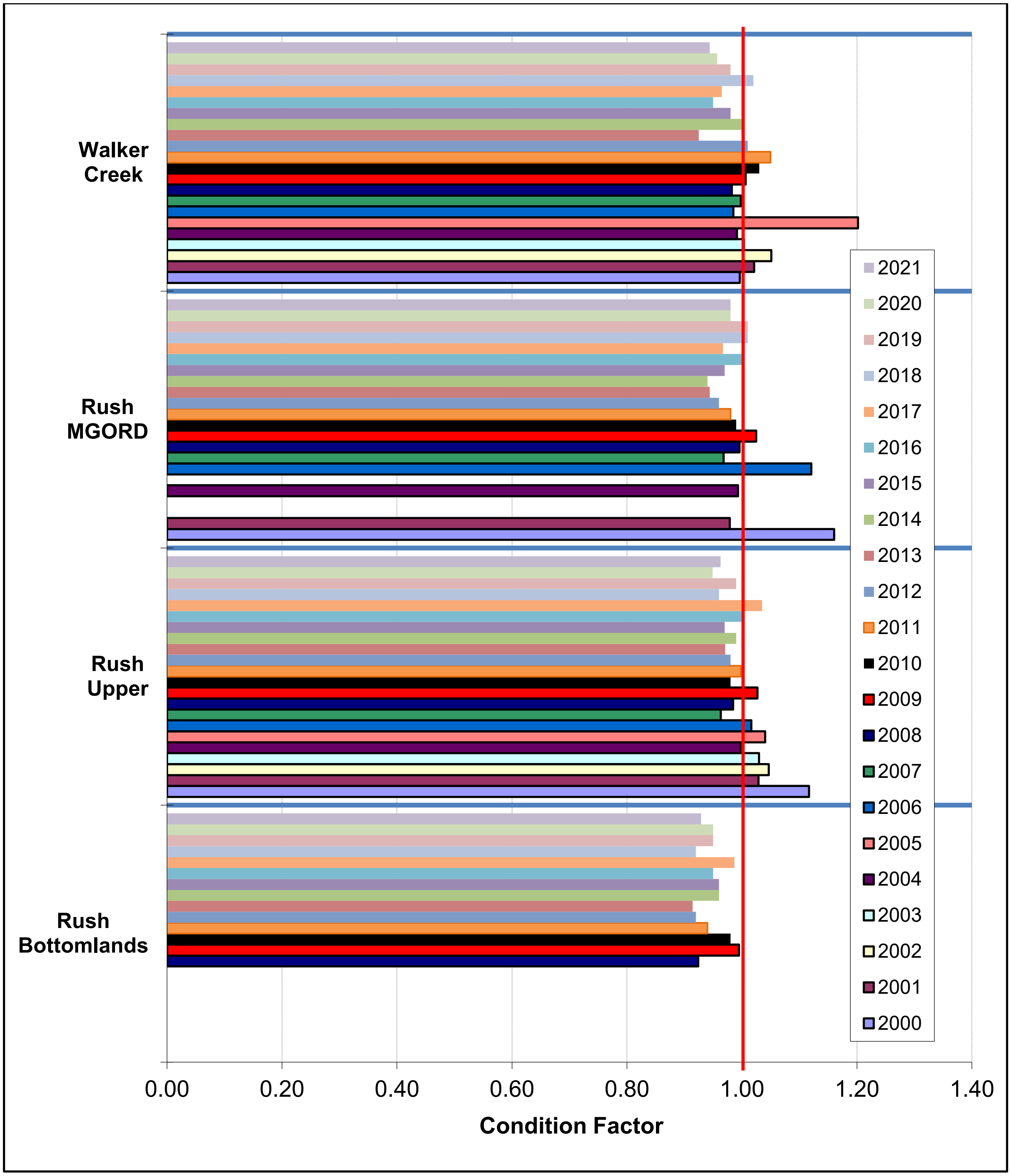

Figure 13. Condition factors for Brown Trout 150 mm to 250 mm in length from sample sections of Rush Creek and Walker Creeks from 2000 to 2021.

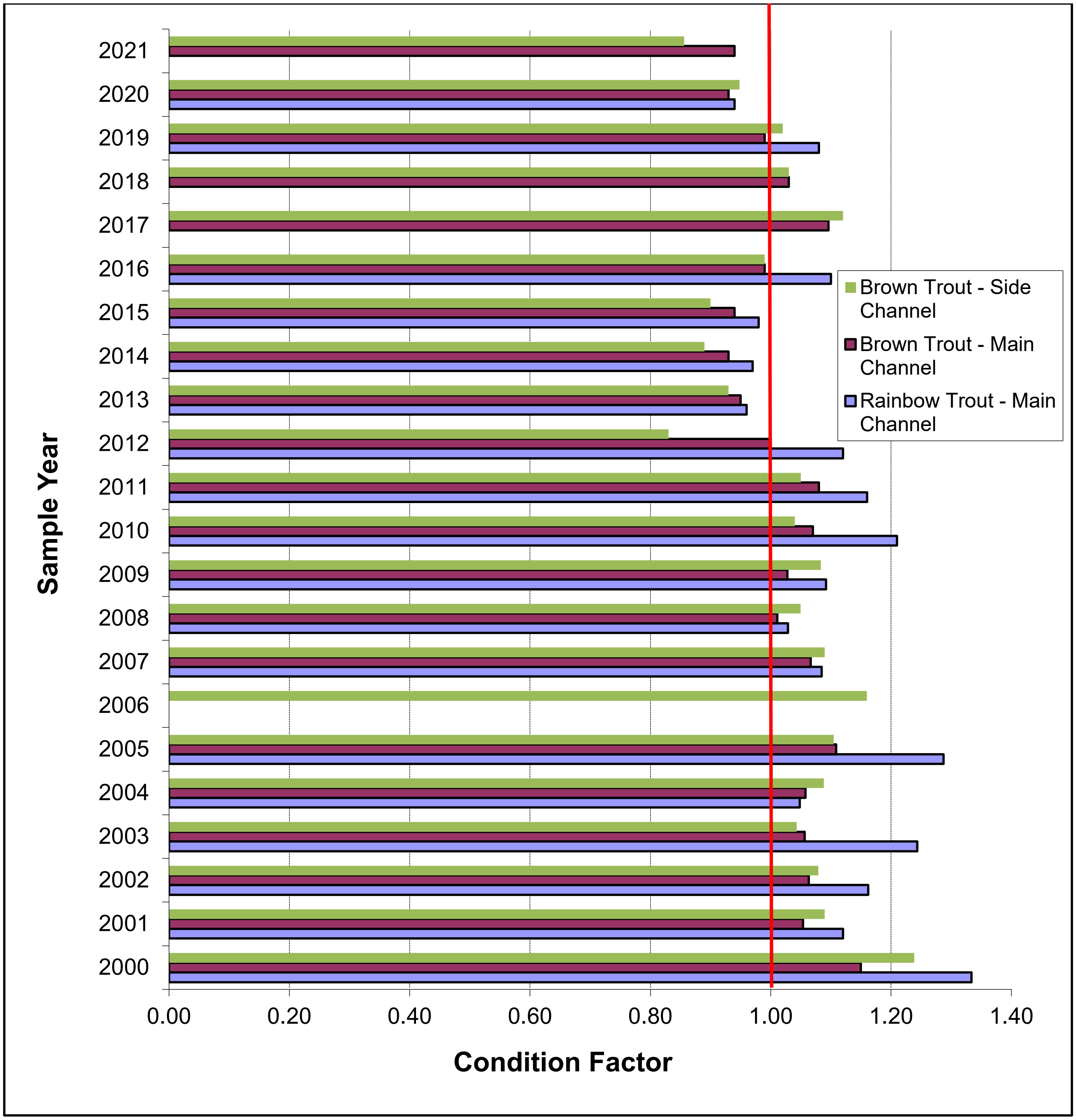

Figure 14. Comparison of condition factors for Rainbow Trout and Brown Trout 150 to 250 mm in length from the main channel and side channel sections of Lee Vining Creek from 2000 to 2021. Main channel was not sampled in 2006 due to high flows.

Estimated Trout Densities Expressed in Numbers per Hectare

Age-0 Brown Trout

The Upper Rush section had an estimated density of 1,657 age-0 Brown Trout/ha in 2021, a decrease of 74% from 2020's estimate of 6,285 age-0 Brown Trout/ha (Figure 15). The 2021 density estimate in the Upper Rush section was 71% lower than the 22-year average of 5,647 age-0 Brown Trout/ha.

The Bottomlands section of Rush Creek had a density estimate of 2,347 age-0 Brown Trout/ha in 2021, a 2% increase from 2020's estimate of 2,295 age-0 trout/ha (Figure 15). When compared to the 14-year average of 2,119 age-0 Brown Trout/ha, the 2021 estimate was 11% higher.

In Walker Creek, the 2021 density estimate of 5,147 age-0 Brown Trout/ha was a 34% increase from the 2020 estimate of 3,846 age-0 trout/ha (Figure 15). The 2021 density estimate was 44% higher than the 23-year average of 3,574 age-0 trout/ha (Figure 15).

In 2021, the estimated density of age-0 Brown Trout in the main channel section of Lee Vining Creek was 419 age-0 trout/ha, which was an 88% decrease from the 2020 density estimate of 3,451 age-0 trout/ha (Figure 16). The 23-year average density estimate for the main channel section of Lee Vining Creek equaled 1,717 age-0 Brown Trout/ha (Figure 16).

In 2021, the estimated density of age-0 Brown Trout in the side channel section of Lee Vining Creek equaled 625 age-0 fish/ha, a 109% increase from the 2020 estimated density of 299 age-0 fish/ha (Figure 16). The 22-year average density estimate for the side channel section of Lee Vining Creeks equaled 332 age-0 Brown Trout/ha (Figure 16).

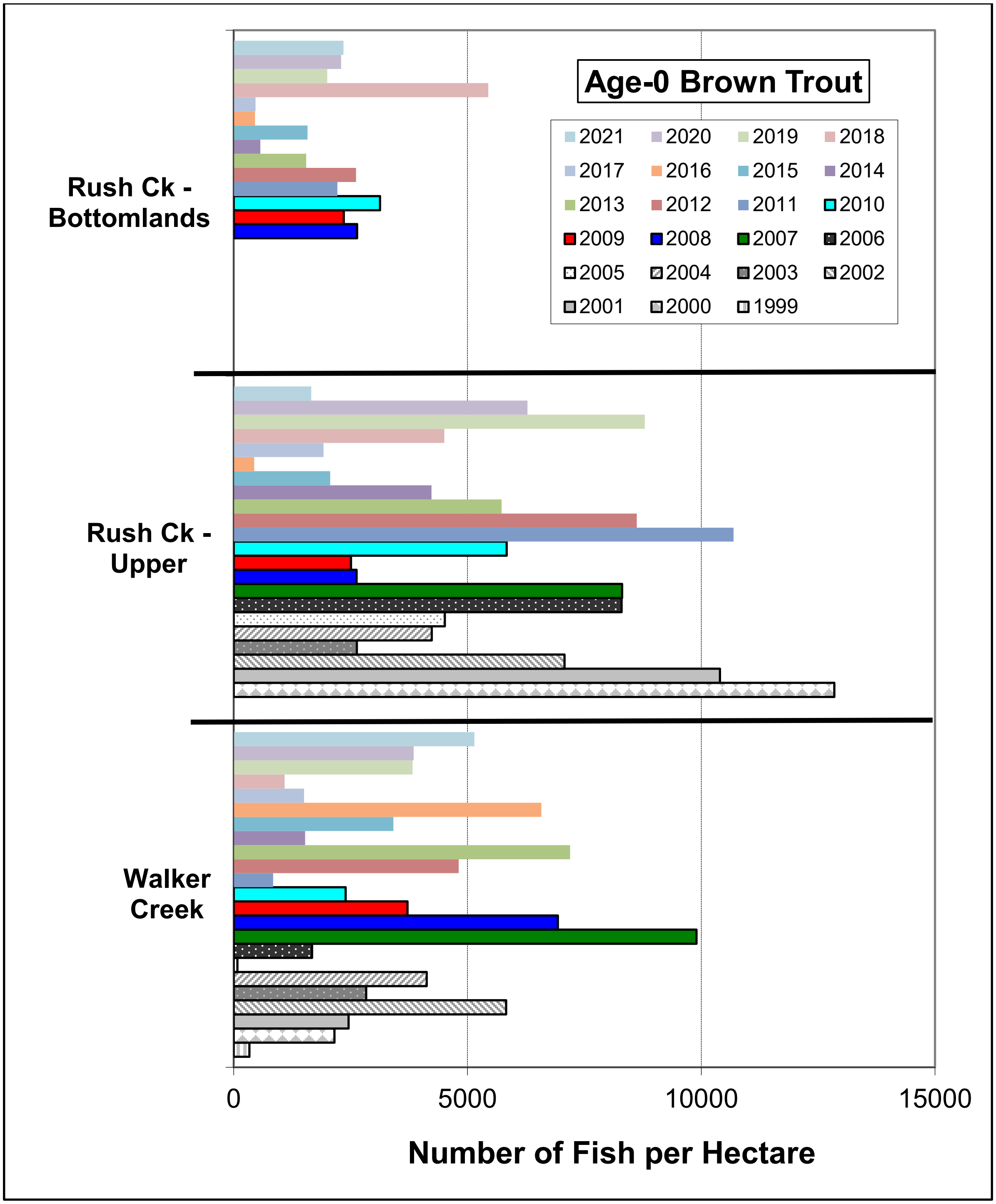

Figure 15. Estimated number of age-0 Brown Trout per hectare in Rush Creek and Walker Creek from 1999 to 2021.

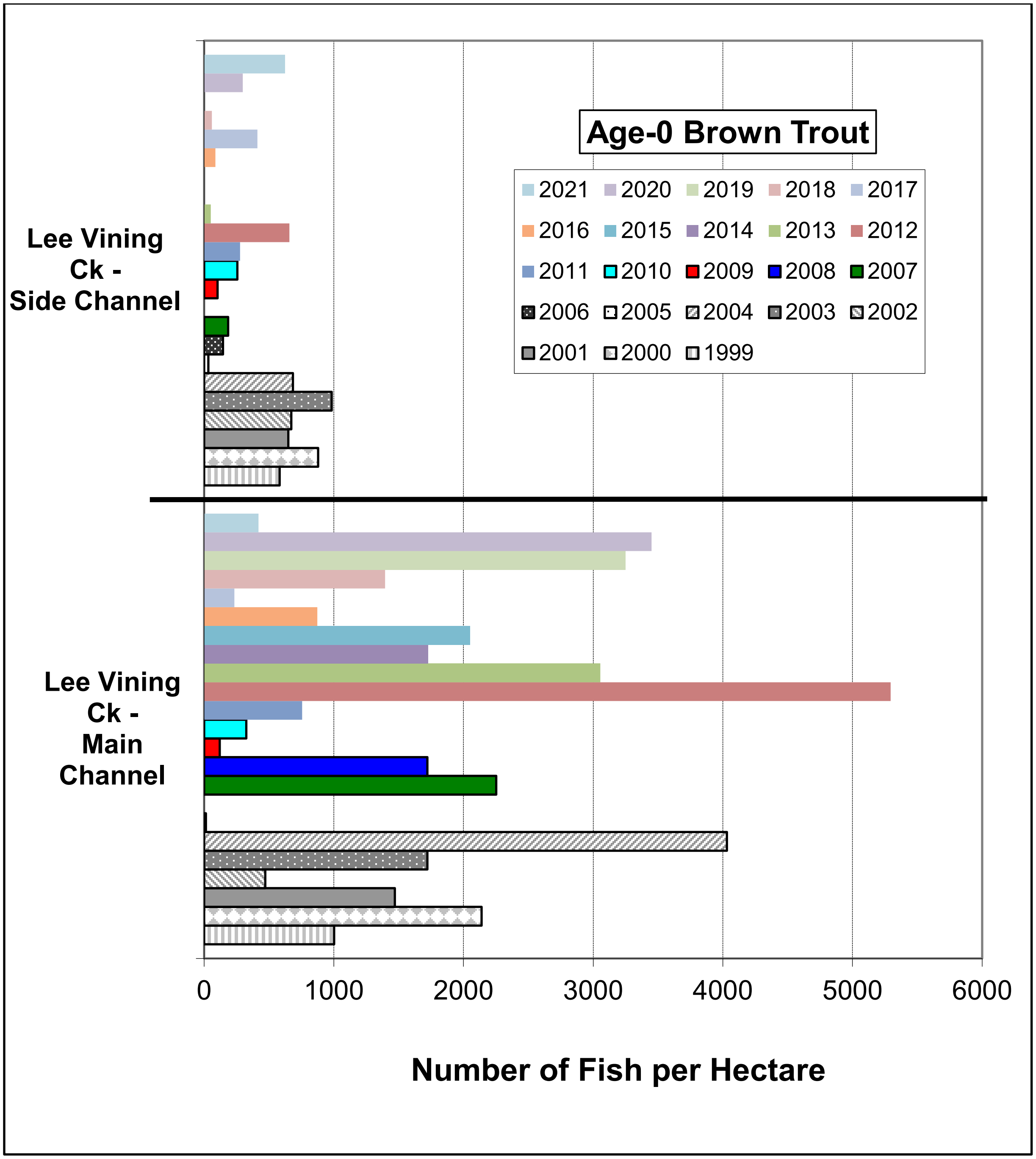

Figure 16. Estimated number of age-0 Brown Trout per hectare in Lee Vining Creek from 1999 to 2021.

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

The Upper Rush section had an estimated density of 2,302 age-1+ Brown Trout/ha in 2021, a decrease of 28% from the 2020 estimate of 3,203 trout/ha (Figure 17). For the Upper Rush section, the 23-year long-term average equaled 1,536 age-1+ Brown Trout/ha.

The estimated density of age-1+ Brown Trout in the Bottomlands section of Rush Creek in 2021 was 1,338 fish/ha, an 11% decrease from the 2020 estimate of 1,495 age-1+trout/ha (Figure 17). For the Bottomlands section, the 14-year long-term average equaled 1,126 age-1+ Brown Trout/ha.

The estimated density of age-1+ Brown Trout in the MGORD section of Rush Creek in 2021 was 411 fish/ha, a 30% decrease from the 2020 estimate of 584 age-1+trout/ha (Figure 17). Since 2001, for the 11 seasons where density estimates were generated for the MGORD, the long-term density estimate of age-1+ Brown Trout averaged 457 fish/ha.

The 2021 density estimate for age-1+ Brown Trout for the Walker Creek section was 3,061 age-1+trout/ha which was a 23% decrease from the 2020 estimate of 3,932 age-1+ trout/ha (Figure 17). For Walker Creek, the 23-year long-term average equaled 1,985 age-1+ Brown Trout/ha.

The 2021 density estimate for age-1+ Brown Trout in the Lee Vining main channel section was 3,350 trout/ha, a 124% increase from the 2020 estimate of 1,499 age-1+ trout/ha (Figure 18). For the Lee Vining Creek main channel section, the 22-year long-term average equaled 1,222 age-1+ Brown Trout/ha.

In 2021, the side channel of Lee Vining Creek supported an estimated density of 357 age-1+ Brown Trout/ha, an increase of 163% from the 2020 estimate of 136 age-1+ Brown Trout/ha (Figure 18). As discussed in previous annual reports, this side channel has experienced variations in the amount of flow that enters the channel due to changes in the geomorphology of the channel’s inlet over time. These variable flows have resulted in highly variable annual wetted areas, which has been a major factor driving density and standing crop estimates for this section. Consequently, the lowest catch of fish (seven in 2015) resulted in the largest density estimate because so little water flowed down the side channel this particular year (Table 5). In September of 2018, more flow continued to enter the top of the side channel, which increased the wetted area within the sampling section to the highest amount since the 2010 and 2011 sampling seasons (Table 5). Since 2018, the side-channel wetted area has decreased; by 12% between 2018 and 2019, by 18% between 2019 and 2020, and in September of 2021 the wetted area decreased by 9% compared to 2020 (Table 5).

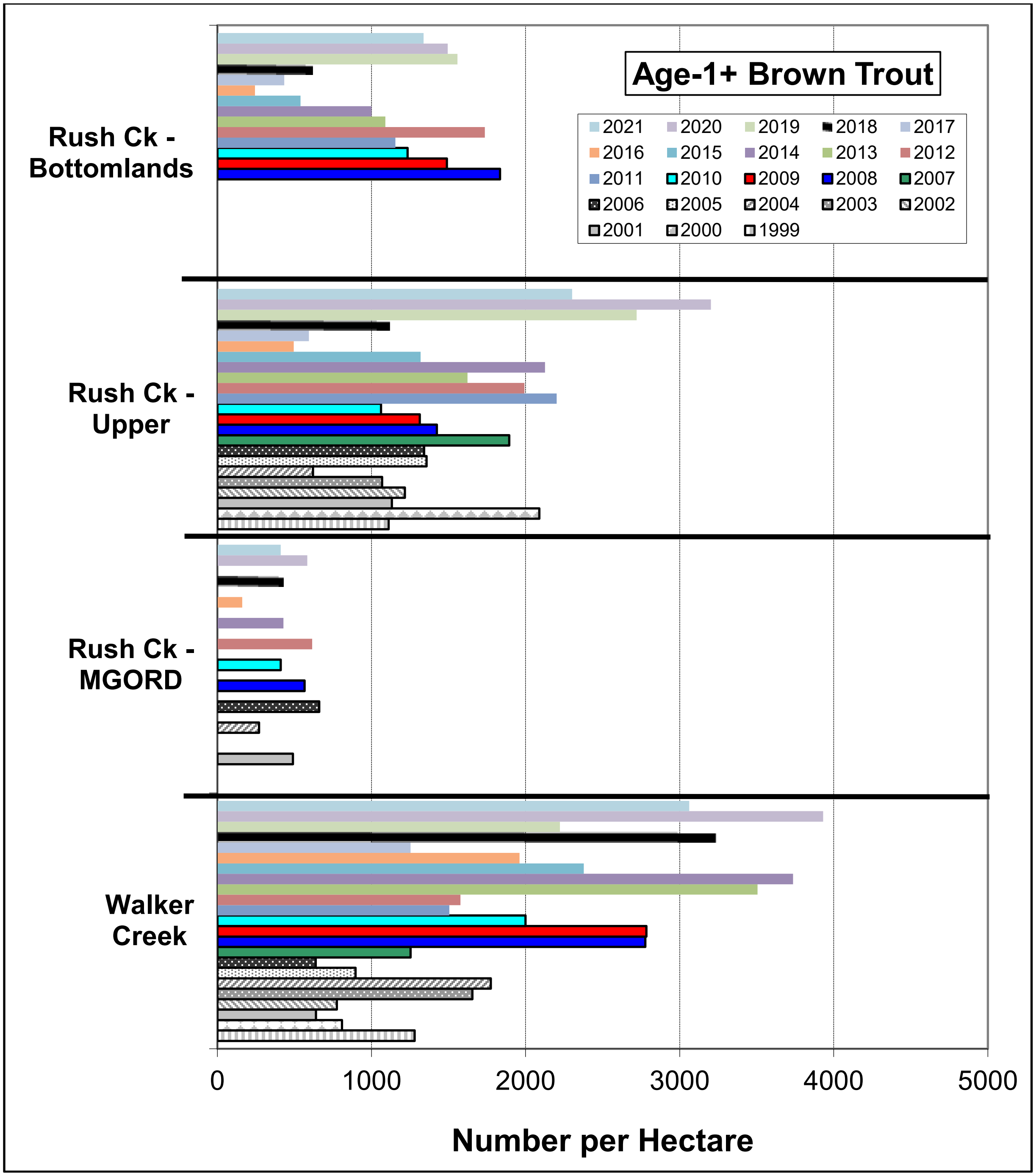

Figure 17. Estimated number of age-1 and older Brown Trout per hectare in sections of Rush and Walker Creeks from 1999 to 2021.

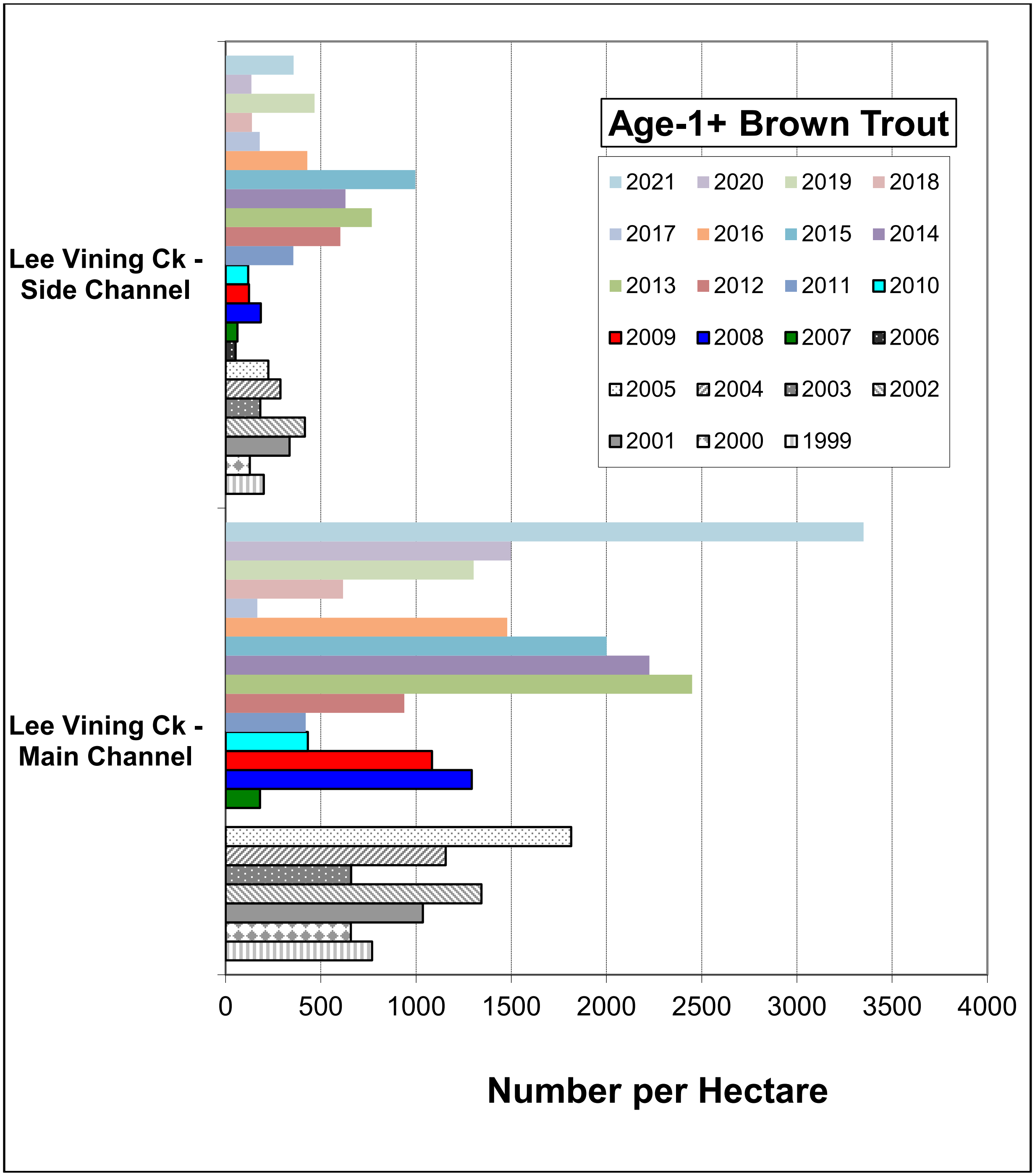

Figure 18. Estimated number of age-1 and older Brown Trout per hectare in sections of Lee Vining Creek from 1999 to 2021.

Table 5. Wetted surface area and total numbers of trout captured in the Lee Vining Creek side channel, from 2007 to 2021.

Sample Year	Wetted Channel Area (m^2)	Total Number of Trout Captured
2007	487.5	22
2008	487.5	20
2009	487.5	26
2010	507.0	20
2011	507.0	30
2012	365.0	45
2013	328.0	16
2014	190.5	12
2015	70.3	7
2016	232.9	12
2017	389.4	23
2018	507.0	10
2019	448.5	21
2020	367.5	16
2021	**336.0**	**33**

Age-0 Rainbow Trout

In 2021, for the 13th consecutive year no age-0 Rainbow Trout were captured in the Lee Vining Creek side channel. In the Lee Vining Creek main channel, two age-0 Rainbow Trout were captured during the 2021 sampling.

The Upper Rush section supported an estimated density of 273 age-0 Rainbow Trout/ha in 2021, a decrease of 68% from the 2020 estimate of 851 age-0 Rainbow Trout/ha.

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

No age-1 and older Rainbow Trout were captured in the Lee Vining Creek side channel during 2021, making it the 11th consecutive year when none were captured. In 2021, a total of two age-1 and older Rainbow Trout were captured in the Lee Vining Creek main channel.

The Upper Rush section supported an estimated density of 121 age-1+ Rainbow Trout/ha in 2021, a decrease of 72% from the 2020 estimate of 431 age-1+ Rainbow Trout/ha.

Estimated Numbers of Trout per Kilometer

The Upper Rush section contained an estimated 2,929 Brown Trout/km (all size classes combined) in 2021, which was a 60% decrease from the 2020 estimate of 7,402 Brown Trout/km (Table 6). The estimated density of age-1+ Brown Trout in 2021 was 1,703 fish/km; a 32% decrease from the 2020 estimate of 2,499 age-1+ fish/km (Table 6).

The Upper Rush section also contained an estimated 291 Rainbow Trout/km (all size classes combined) in 2021, a 73% decrease from 2020's estimate of 1,095 Rainbow Trout/km. In 2021 the density estimate included 89 age-1+ Rainbow Trout/km versus 431 age-1+ Rainbow Trout/km in 2020.

The Bottomlands section contained an estimated 2,432 Brown Trout/km (all size classes combined) in 2021, which was a 3% decrease from the 2020 estimate of 2,501 fish/km (Table 6). In 2021, the estimate of 883 age-1+ Brown Trout/km represented a 10% decrease from the 2020 estimate of 986 age-1+ Brown Trout/km (Table 6).

The Lee Vining Creek main channel contained an estimated 2,001 Brown Trout/km (all size classes combined) in 2021, which was a 21% decrease from the 2020 estimate of 2,526 fish/km (Table 7). In 2021, the estimate of 1,777 age-1+ Brown Trout/km represented a 132% increase from the 2020 estimate of 767 age-1+ trout/km (Table 7).

The Lee Vining side channel contained an estimated 196 Brown Trout/km (all size classes combined) in 2021, a 113% increase from the 2020 estimate of 92 fish/km (Table 7). For age-1+ Brown Trout, the 2021 density estimate was 71 Brown Trout/km which was a 145% increase from the 2020 density estimate 29 fish/km (Table 7).

Table 6. Estimated total numbers (number of age-1 and older in parentheses) of Brown Trout per kilometer of stream channel for Rush Creek sample sections from 2010 to 2021.

Collection Location	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Rush Creek, Upper Rush	5,726 (881)	10,821 (1,833)	8,288 (1,556)	6,105 (1,347)	4,574 (1,530)	2,468 (963)	766 (406)	1,863 (440)	4,835 (963)	8,910 (2,105)	7,402 (2,499)	**2,929 (1,703)**
Rush Creek, Bottom-lands	3,405 (963)	2,725 (929)	3,208 (1,279)	1,980 (817)	1,098 (700)	1,422 (362)	523 (179)	637 (308)	4,608 (471)	2,094 (1,137)	2,501 (986)	**2,432 (883)**

Table 7. Estimated total numbers of Brown and Rainbow Trout (number of age-1 and older in parentheses) per kilometer of stream channel for Lee Vining Creek sample sections from 2010 to 2021.

Collection Location	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Lee Vining, Main Channel	518 (326)	727 (258)	4,361 (506)	3,765 (1,867)	2,444 (1,471)	2,027 (1,043)	1,973 (989)	216 (90)	1,189 (436)	2,299 (675)	2,534 (767)	**2,001 (1,777)**
Lee Vining, Side Channel	103 (36)	159 (87)	257 (123)	131 (123)	95 (95)	100 (100)	97 (97)	130 (40)	51 (36)	108 (108)	92 (29)	**196 (71)**

Estimated Trout Standing Crops (kg/ha)

The total (Brown and Rainbow Trout) estimated standing crop in the Upper Rush section was 138 kg/ha in 2021, a 29% decrease from 195 kg/ha in 2020 (Table 8 and Figure 19). Rainbow Trout comprised 10.8 kg/ha of the 2021 standing crop estimate (Figure 19). For the Upper Rush section, the 23-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 78 kg/ha in 2021, a 7% decrease from 84 kg/ha in 2020 (Table 8 and Figure 19). For the Bottomlands section of Rush Creek, the 14-year average standing crop of Brown Trout equaled 82 kg/ha.

The estimated standing crop for Brown Trout in the MGORD section of Rush Creek was 67 kg/ha in 2021, a 17% decrease from 81 kg/ha in 2020 (Table 8 and Figure 19). For the 11 seasons where Brown Trout standing crop estimates were generated for the MGORD; the average value equaled 86 kg/ha.

The estimated standing crop for Brown Trout in Walker Creek was 158 kg/ha in 2021, a 34% decrease from the 2020 estimate of 240 kg/ha (Table 8 and Figure 19). For Walker Creek, the 23-year average standing crop of Brown Trout equaled 144 kg/ha.

The estimated total standing crop for Brown Trout in the Lee Vining Creek main channel in 2021 was 146 kg/ha; an increase of 52% from the 2020 estimate of 96 kg/ha (Table 9 and Figure 20). The long-term average for the 22-year sampling period is 125 kg/ha.

The estimated standing crop of Brown Trout in the Lee Vining Creek side channel was 22 kg/ha in 2021, which represented a 120% increase from the 2020 estimate of 10 kg/ha (Table 9 and Figure 20). No Rainbow Trout were captured in the Lee Vining Creek side channel in 2021 and none have been sampled in the side channel section for 11 consecutive years (2011-2021).

Table 8. Comparison of Brown Trout standing crop (kg/ha) estimates between 2016 and 2021 for Rush Creek sections. These six years include one dry year of 2016, followed by the extremely wet RY 2017, the normal RY 2018, the wet RY 2019, dry-normal-1 RY 2020 and dry RY 2021.

Collection Location	2016 Total Standing Crop (kg/ha)	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)	Percent Change Between 2020 and 2021
Rush Creek – Upper	62	123	188*	291**	195***	138[#]	-29%
Rush Creek - Bottomlands	34	50	103	91	84	78	-7%
Walker Creek	172	85	245	179	240	158	-34%

*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

Table 9. Comparison of total (Brown and Rainbow Trout) standing crop (kg/ha) estimates between 2016 and 2021 for the Lee Vining Creek sections. These six years include one dry year of 2016, followed by the extremely wet RY 2017, the normal RY 2018, the wet RY 2019, dry-normal-1 RY 2020 and dry RY 2021. The Rainbow Trout portion of the main channel's total estimated biomass is provided within the parentheses.

Collection Location	2016 Total Standing Crop (kg/ha)	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)	Percent Change Between 2020 and 2021
Lee Vining Creek - Main Channel	113 (8.2)	21 (0)	70 (0)	192 (4.6)	96 (0.6)	146 (0)	+52%
Lee Vining Creek – Side Channel	31	20	7	25	10	22	+120%

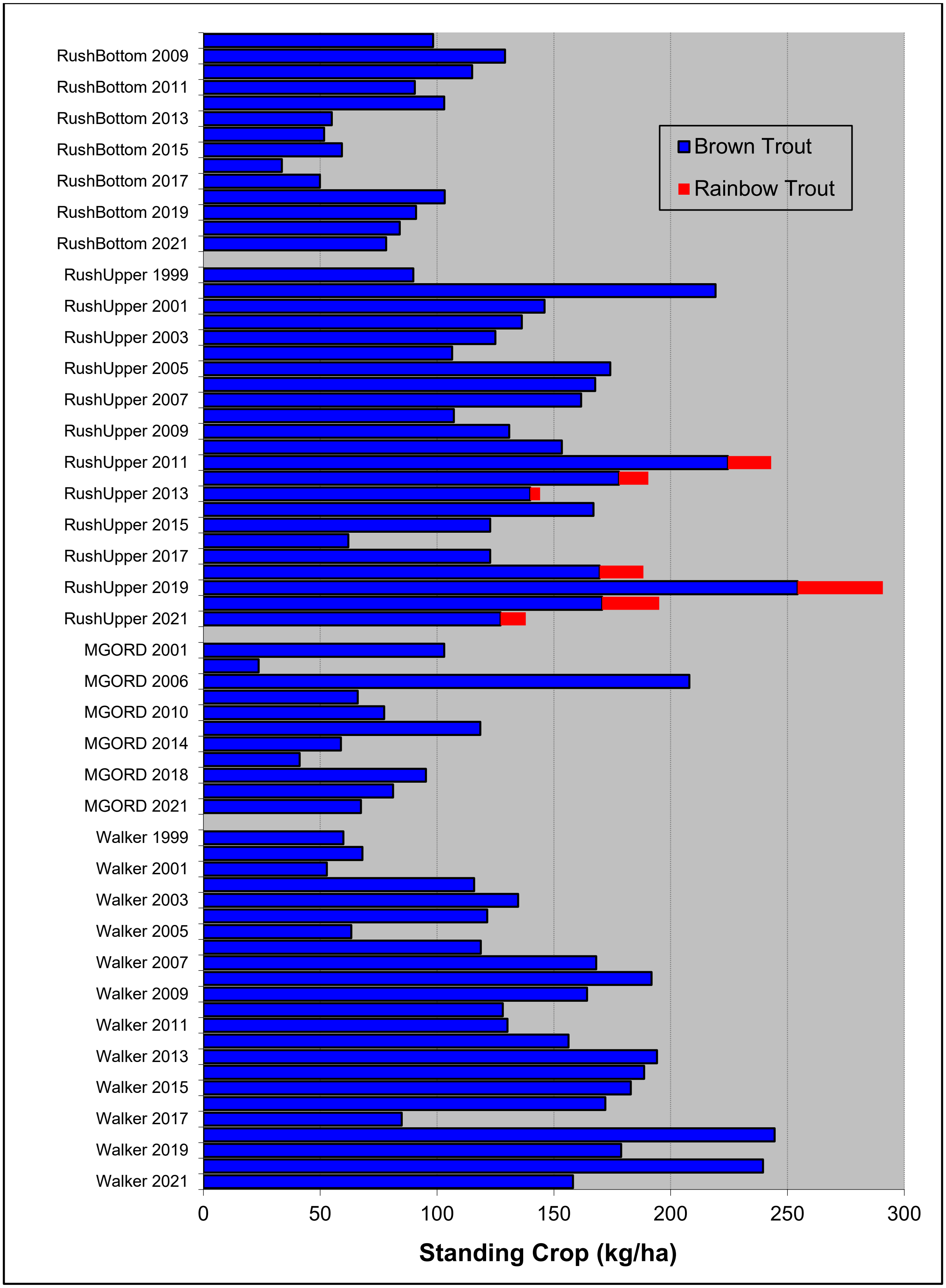

Figure 19. Estimated total standing crop (kilograms per hectare) of Brown Trout in Rush Creek sample sections from 1999 to 2021.

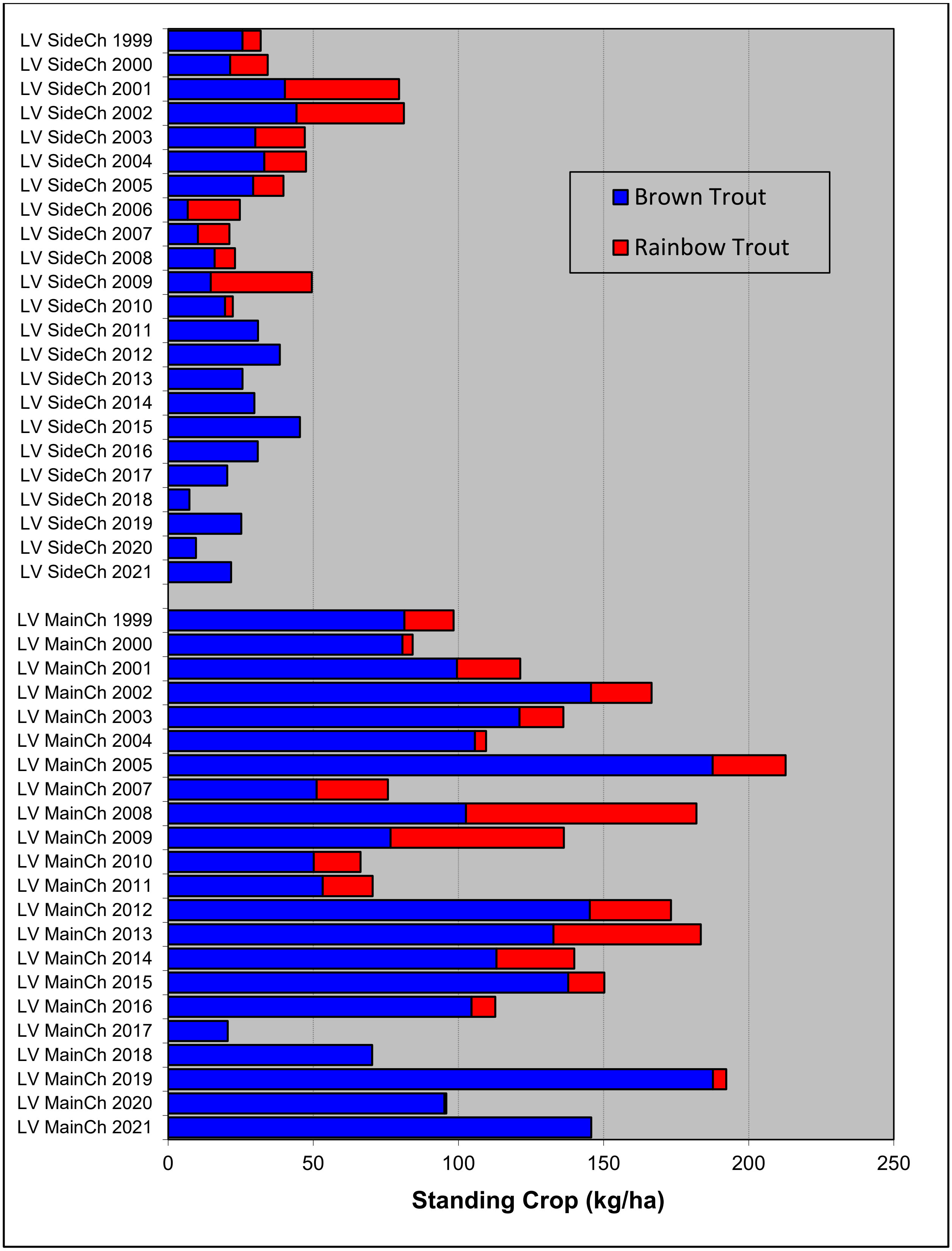

Figure 20. Estimated total standing crop (kilograms per hectare) of Brown Trout and Rainbow Trout (red) in Lee Vining Creek sample sections from 1999 to 2021.

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

In the Upper Rush section, the RSD-225 equaled 6 for 2021, the fourth straight year of large drops from the record RSD-225 value of 78 for 2017 (Table 10). The 2021 RSD-225 value was most likely influenced by greater numbers of fish smaller than 225 mm which comprised 94% of the trout ≥150 mm (Table 10). The RSD-300 value was 1 in 2021, the same as in 2020 (Table 10). This low RSD-300 value was influenced by continued low numbers of Brown Trout >300 mm captured in 2021 (Table 10). Over 22 sampling years, a total of 153 Brown Trout ≥300 mm were captured in the Upper Rush Creek section, an average of 7.0 fish ≥300 mm per year (Table 10).

In the Bottomlands section of Rush Creek, the RSD-225 for 2021 equaled 9, the same value recorded in 2020 (Table 10). As in the Upper Rush section, the Bottomlands 2021 RSD-225 value was most likely influenced by the large numbers of fish smaller than 225 mm which comprised 91% of the trout ≥150 mm. The RSD-300 value was 0 in 2021, although one Brown Trout ≥300 mm was captured in the Bottomlands section (Table 10). Over the 14 sampling years, a total of 27 Brown Trout ≥300 mm were captured in the Bottomlands section, an average of 1.9 fish ≥300 mm per year (Table 10).

In the MGORD, the RSD-225 value has increased from 47 in 2019 to 48 in 2020 and to 53 in 2021 (Table 10). In 2021, the RSD-300 value was 11, a small decrease from the 2020 RSD-300 value of 13 (Table 10). The RSD-375 value increased slightly from 2 in 2020 to 3 in 2021 (Table 10). The two-pass catch of Brown Trout ≥150 mm in the MGORD during the 2021 season was 431 fish, which included 47 fish ≥300 mm in length and 12 of these fish were ≥375 mm in length (Table 10). For sampling conducted between 2001 and 2012, the annual average catch of Brown Trout ≥300 mm equaled 180 fish/year; then for the past nine sampling years the annual average catch of Brown Trout ≥300 mm equaled 40 fish/year (Table 10). 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 five seasons following the five-year drought, the recruitment of larger, older fish appears to be a relatively slow process (Table 10).

RSD values in Lee Vining Creek were generated for the main channel only (Table 11). The RSD-225 value for main channel decreased from 14 in 2020 to 3 in 2021, most likely influenced by larger numbers of trout <225 mm in length that were captured; which comprised 97% of the fish ≥150 mm (Table 11). In 2021, no Brown Trout greater than 300 mm in length were captured in Lee Vining Creek main channel, thus the RSD-300 value was 0 (Table 11).

Table 10. RSD values for Brown Trout in Rush Creek sections from 2000 to 2021.

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	**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	**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 10 (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	**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 11. RSD values for Brown Trout in the Lee Vining Creek main channel section from 2000-2021.

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
Main Channel	**2021**	**175**	**169**	**6**	**0**	**0**	**3**	**0**
Main Channel	2020	80	69	11	0	0	14	0
Main Channel	2019	131	107	22	2	0	18	2
Main Channel	2018	51	39	10	2	0	24	4
Main Channel	2017	23	17	5	1	0	26	4
Main Channel	2016	169	145	24	0	0	14	0
Main Channel	2015	210	192	18	0	0	9	0
Main Channel	2014	200	173	27	0	0	14	0
Main Channel	2013	325	308	16	1	0	5	0
Main Channel	2012	111	72	37	2	0	35	2
Main Channel	2011	60	31	23	5	1	48	10
Main Channel	2010	62	28	32	2	0	55	3
Main Channel	2009	137	106	30	1	0	23	1
Main Channel	2008	149	138	11	0	0	7	0

Table 11 (continued).

Main Channel	2007	29	24	5	0	0	17	0
Main Channel	2006	Not sampled in 2006 due to unsafe high flows						
Main Channel	2005	60	37	20	2	1	38	5
Main Channel	2004	70	60	8	2	0	14	3
Main Channel	2003	52	27	23	2	0	48	4
Main Channel	2002	100	74	23	3	0	26	3
Main Channel	2001	90	71	16	3	0	21	3
Main Channel	2000	51	32	18	1	0	37	2

PIT Tag Recaptures

PIT Tags Implanted between 2009 and 2021

Between 2009 and 2021, a total of 10,676 PIT tags were implanted in Brown Trout and Rainbow Trout within the annually sampled sections of Rush, Lee Vining and Walker Creeks (Appendix B). 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 tabulated in Appendix B.

In 2021, a total of 865 trout received PIT tags and adipose fin clips in Rush and Lee Vining creeks (Table 12). In addition, three recaptured adipose fin-clipped fish had shed their original tags and were re-tagged, thus a total of 868 PIT tags were implanted during the 2021 fisheries sampling (Table 12). Of the 868 trout tagged, 561 were age-0 Brown Trout and 262 were age-1 and older Brown Trout (Table 12). For Rainbow Trout, 36 age-0 fish and nine older fish were tagged (Table 12). Two hundred fifty-nine of the age-1+ Brown Trout tagged in the MGORD section were ≤250 mm in total length and were presumed to be age-1 fish (Table 12). The 115 age-0 Brown Trout tagged in the MGORD were the most age-0 fish tagged in a single season within this section (Table 12). Tagged and recaptured fish provided empirical information to estimate fish growth, tag retention, fish movements, and apparent survival rates.

Table 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

In September of 2021, a total of 129 previously tagged trout (that retained their tags) were recaptured in the Rush Creek watershed (Appendix C). Thirty-two of the recaptures occurred in the Upper Rush section (one Rainbow Trout), followed by five recaptures in the Bottomlands section, 55 recaptures in Walker Creek and 37 recaptures in the MGORD (Appendix C). In September of 2021, a total of 25 previously tagged Brown Trout (that retained their tags) were recaptured in the Lee Vining Creek main channel section (Appendix C). During the 2021 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 2020 and 2021 will be referred to as 2021 growth rates. A 2021 trout refers to a fish recaptured in September of 2021. An age of a PIT tagged trout reflects its age during the sampling year. For instance, an age-1 trout in 2021 indicates that a trout was tagged in September 2020 at age-0 and its length and weight were remeasured in September 2021 when it was recaptured.

Growth of Age-1 Brown Trout between 2020 and 2021

In 2021, a total of 70 known age-1 Brown Trout were recaptured that were tagged as age-0 fish in 2020, for an overall recapture rate of 12.1% (70/581 age-0 fish tagged in 2020). Of the 70 age-1 recaptures; 22 of these fish were from Rush Creek sections, 26 fish were from Walker Creek and 22 fish were from the Lee Vining Creek main channel section. Thus, by creek, the age-1 recapture rates for 2021 were 22% in Lee Vining Creek (14% in 2020, 23% in 2019, 29% in 2018 and 2% in 2017), 7% in Rush Creek (6% in 2020, 7% in 2019, 14% in 2018, 19% in 2017 and 5% in 2016), and 28% in Walker Creek (45% in 2020 and 19% in 2019). These recapture rates suggest survival between age-0 and age-1 in Rush Creek in 2021 remained somewhat comparable to the previous year, decreased in Walker Creek and that survival rates in Lee Vining Creek in 2021 increased from the previous year.

In the Upper Rush section, 20 age-1 Brown Trout were recaptured in 2021 and the average growth rates of these trout were 66 mm and 27 g (Table 13). Compared to 2020 rates, the average growth rates of the 20 age-1 Brown Trout were higher by 11 mm and 6 g (Table 13). 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 13). After the 2017 season, growth rates of age-1 Brown Trout in Upper Rush have remained 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 13).

In the Bottomlands section of Rush Creek, two age-1 Brown Trout were recaptured in 2021 and the average growth rates of these trout were 67 mm and 26 g (Table 13). Compared to 2020 rates, the growth rates of the two age-1 Brown Trout were higher by 3 mm and lower by 3 g (Table 13). In terms of weight, the average growth rates for age-1 Brown Trout in the Bottomlands have dropped for four consecutive years since 2017 (Table 13).

In Walker Creek, 26 age-1 Brown Trout were captured in 2021 and the average growth rates of these 26 trout were 47 mm and 18 g; decreases of 7 mm and 6 g from the 2020 average growth rates (Table 13). The growth rates of age-1 Brown Trout in Walker Creek have typically been lower than the rates documented in Rush and Lee Vining creeks (Table 13).

In Lee Vining Creek, 22 age-1 Brown Trout were recaptured in 2021 and the average growth rates of these trout were 63 mm and 27 g (Table 13). Compared to 2020 rates, the growth rates of the 22 age-1 Brown Trout were lower by 9 mm in length and by 2 g (Table 13). Growth rates (in weight) of age-1 Brown Trout in Lee Vining Creek have decreased for four straight years after the record high rates documented in 2017 (Table 13).

Growth of Age-2 Brown Trout between 2020 and 2021

In 2021, a total of 33 known age-2 Brown Trout were recaptured that were tagged as age-0 fish in 2019, for a recapture rate of 4.6% (33/720 age-0 fish tagged in 2019). Five of these fish were recaptured in Rush Creek, 24 of these fish were recaptured in Walker Creek, and four fish were recaptured in Lee Vining Creek. In addition, within the MGORD section of Rush Creek, 22 Brown Trout were captured in 2021 that were tagged as presumed age-1 fish in 2020 and these presumed age-2 fish had a recapture rate of 16.7% (22/132 age-1 fish tagged in 2020).

Within the Upper section of Rush Creek, four age-2 fish were recaptured in 2021 that had been tagged as age-0 fish in 2019 (Table 13). Between age-1 and age-2, the average growth rates of these four Brown Trout were 54 mm and 42 g (Table 13). Compared to 2020 rates, the growth rates of the four age-2 Brown Trout were higher by 10 mm, but lower by 11 g (Table 13). The 2021 average growth rate (in weight) of age-2 Brown Trout in Upper Rush was the lowest recorded for the past eight years (Table 13).

In the Bottomlands section of Rush Creek, one previously tagged age-2 Brown Trout was recaptured in 2021. Between age-1 and age-2, this fish had grown by 35 mm and gained 33 g in weight (Table 13). By weight, this was the lowest growth rate recorded for an age-2 Brown Trout in the Bottomlands section in the past seven years (Table 13).

In Walker Creek, 24 age-2 fish were recaptured in 2021 that had been tagged as age-0 fish in 2019 (Table 13). Between age-1 and age-2, the average growth rates of these 24 Brown Trout were 25 mm and 19 g (Table 13). The 2021 average growth rates of age-2 Brown Trout in Walker Creek were the lowest recorded for the 12 years of available data (Table 13).

In the Lee Vining Creek main channel section, four age-2 Brown Trout were recaptured in 2021 that had been tagged as age-0 fish in 2019. Between age-1 and age-2, the growth rates of these four Brown Trout were 46 mm and 47 g, nearly a 50% reduction in growth rate from the previous year (Table 13). The 2021 growth rate in weight of age-2 Brown Trout was the lowest recorded in the past seven years for the Lee Vining Creek main channel section (Table 13).

Growth of Age-3 Brown Trout between 2020 and 2021

In 2021, four known age-3 Brown Trout were recaptured in Walker Creek section that were tagged as age-0 fish in 2018 (and were also captured in 2020). Between age-2 and age-3, the growth rates of these four Brown Trout were 12 mm and 18.5 g (Table 13). In the Lee Vining Creek main channel section, one known age-3 Brown Trout was captured that was also caught at age-2 in 2020. Between age-2 and age-3, this fish grew 30 mm in length and gained 48 g (Table 13).

Growth of Age-4 Brown Trout between 2020 and 2021

In 2021, one known age-4 Brown Trout was recaptured in the Upper Rush section that was tagged as age-0 fish in 2017 (and was also captured in 2020) (Table 13). Between 2020 and 2021, this age-4 Brown Trout grew by 38 mm in length and by 144 g in weight (Table 13). This was the first PIT-tagged age-4 fish recaptured in the Upper Rush section (that was also caught at age-3) since 2013 (Table 13).

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 has been focused on age-0 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 250 mm total length was made 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 known age fish.

In 2021, three age-1 Brown Trout were captured in the MGORD that were tagged at age-0 in 2020; all of these fish were tagged in the MGORD. Between 2020 and 2021, the average growth rates of these three fish were 100 mm and 68 g. At age-1, these fish had total lengths of 186, 189 and 225 mm. In weight, the growth rate of age-1 Brown Trout in the MGORD was approximately 2.5 times greater than the age-1 growth rate in the Upper Rush section.

In 2021, 22 Brown Trout were recaptured in the MGORD that were PIT tagged in the MGORD as presumed age-1 fish in 2020. Between age-1 and age-2, the average growth rates of these 22 Brown Trout were 60 mm and 88 g. For comparison, in 2020 five presumed age-2 fish had average growth rates of 49 mm and 68 g. The 22 age-2 fish recaptured in 2021 ranged from 240 mm to 284 mm in FL.

In 2021, two Brown Trout were recaptured in the MGORD that had been PIT tagged in the MGORD as presumed age-1 fish in 2019 and were also recaptured as presumed age-2 fish in the MGORD in 2020. Between age-2 and age-3, the average growth rates of these two fish were 25 mm and 40 g. In comparison, two presumed age-3 Brown Trout recaptured in 2020 had average growth rates of 35 mm and 70 g between age-2 and age-3.

In 2021, one Brown Trout was recaptured in the MGORD that had been PIT tagged in the MGORD as presumed age-1 fish in 2018; this fish had been recaptured each year since 2018. Between 2020 and 2021, this presumed age-4 fish grew by 23 mm and 76 g. At age-4, this fish was 368 mm in total length.

Table 13. Average growth (length and weight) of all Brown Trout recaptured from 2009 through 2021 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
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
	Age 2		58/70	54/73	43/42	41/42		64/69	99/176•	108/239	39/66	48/71	44/55	54/42
	Age 3				14/29		24/41				11/40*	15/27*	41/49*	
	Age 4					12/-22								38/144*
	Age-5													
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
	Age 2		50/54	35/32	30/28	27/22	32/29*	62/62			39/55	36/44*		35/33*
	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
	Age 2		66/95		77/110	33/34	35/29	47/40	47/49	77/128*		60/91*	70/81	46/47
	Age 3			34/92		23/48*	16/20*	27/32	42/75					30/48
	Age 4				21/41*				25/47*					
	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
	Age 2		31/26	60/56	40/33	27/21	39/35		47/44	37/37	42/52		36/30	25/19
	Age 3			28/44	18/12	9/2	20/36	27/29		42/59*	25/37	25/37		12/19
	Age 4				7/2	2/-16*		28/45*			27/37*		8/-5	
	Age-5						0/-10*							

Growth of MGORD Brown Trout from non-consecutive years

One age-2 Brown Trout was captured in the MGORD in 2021 that was tagged as age-0 fish in 2019. This fish grew by 160 mm and 161 g between age-0 and age-2.

Five age-3 Brown Trout were recaptured in the MGORD in 2021 that were tagged as presumed age-1 fish in 2019. During the two years between their initial captures and recaptures, these five fish had average growth rates of 99 mm and 171 g.

The other non-consecutive year recaptures within the MGORD in 2020 were of two large Brown Trout that were 512 mm and 542 mm in length when caught on 9/9/21. The 512 mm fish was first captured and PIT tagged in 2016 and was 496 mm and 1,602 g. Its first recapture occurred in 2017 and it had lost 148 g. This fish was recaptured again in 2018 and had lost another 370 g in weight. This fish was then recaptured a third time in 2021 and had lost another 52 g between 2018 and 2021. Unfortunately, this fish was in poor condition and died during the 2021 electrofishing sampling. This fish was likely nine to 10 years old in 2021.

The 542 mm Brown Trout was originally tagged in 2011 and was 348 mm in length and weighed 410 g. Its first recapture occurred the following year in 2012 and the one-year growth between 2011 and 2012 was 3 mm and 70 g. Its second recapture was in 2017 and five-year growth between 2102 and 2017 was 199 mm and 1,550 g. Its third recapture was in 2020 and in the three years since its previous capture, this fish had lost 10 mm in length and lost 80 g in weight. Its fourth recapture was in 2021 and this fish grew by 2 mm (could be measurement error) and lost another 573 g in weight. This likely age-12 or age-13 Brown Trout was 542 mm in total length and weighed 1,377 g with a condition factor of 0.87 when recaptured on 9/10/21. The prior year, on 9/9/20, its condition factor equaled 1.24, when its weight was 1,950 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). However, in 2021 none of the recaptured PIT tagged fish had moved from another section.

As mentioned in previous reports, a PIT tag antenna array and receiver at the lower end of the MGORD would provide better knowledge of the timing or magnitude of movement of Brown Trout between the Upper Rush and MGORD sections.

PIT Tag Shed Rate of Trout Recaptured in 2021

In 2021, a total of 132 trout with adipose fin clips were recaptured and three of these fish failed to produce a PIT tag number when scanned with the tag reader (one from Walker Creek, one from the MGORD, and one from Upper Rush). Assuming that all these fish were previously PIT tagged, the 2021 calculated shed rate was 2.3% (3 shed tags/132 clipped fish recaptured). This rate was much lower than the 2019 shed rate of 20% and the 2020 shed rate of 6.8%. 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 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 has remained relatively stable.

Comparison of Length-at Age amongst Sample Sections

During the September 2021 sampling, four age-classes of PIT tagged Brown Trout were recaptured within four fisheries monitoring sections in Rush, Walker and Lee Vining creeks (Tables 14 and 15). 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 2013-2021 (Tables 14 and 15).

In Upper Rush, the average length-at-age-1 in 2021 was 154 mm, 9 mm greater than the average length-at-age-1 in 2020, yet was still the second lowest average for the seven years of available data (Table 14). In 2021, age-1 Brown Trout in Upper Rush were 1 mm smaller than age-1 fish in the Bottomlands section (Table 14). In the Bottomlands section, the average length-at-age-1 in 2021 was 155 mm, same as the 2019 average length-at-age-1, and the lowest average value for the past seven years of available data (Table 14).

In Upper Rush, the average length-at-age-2 in 2021 was 198 mm, 23 mm less than the average length-at-age-2 in 2020 and 115 mm lower than in 2017 (Table 14). For Upper Rush, this was the lowest average value for the past seven years of available data (Table 14). In 2021, one age-2 PIT tagged Brown Trout was recaptured in the Bottomlands section and this fish was 186 mm in total length (Table 14).

In 2021, one PIT-tagged age-3 Brown Trout was recaptured in Upper Rush and this fish was 220 mm in total length, the lowest value for the past six years of available data (Table 14). In 2021, two PIT tagged age-3 Brown Trout were recaptured in the Bottomlands section and the average length-at-age-3 equaled 231 mm, 9 mm lower than the 2020 value (Table 14).

In 2021, one PIT-tagged age-4 Brown Trout was recaptured in Upper Rush section and this fish was 325 mm in total length (Table 14). The 2014 sampling season was the last time PIT tagged age-4 Brown Trout were recaptured in Upper Rush or the Bottomlands section (Table 14).

For Walker Creek in 2021, 26 age-1 Brown Trout were recaptured and the average length-at-age-1 was 138 mm, 13 mm less than the average length-at-age-1 in 2020 and the lowest average value for the six years of available data (Table 14). In 2021, 24 PIT tagged age-2 Brown Trout were recaptured in the Walker Creek sampling section and the average length-at-age-2 equaled 175 mm, the lowest average value for the past seven years of available data (Table 14). In 2021, four PIT tagged age-3 Brown Trout were recaptured in the Walker Creek sampling section and the average length-at-age-3 equaled 205 mm, the lowest average value for the past seven years of available data (Table 14).

For the Lee Vining Creek main channel in 2021, 22 age-1 Brown Trout were recaptured and the average length-at-age-1 for these Brown was 154 mm, 1 mm less than in 2020 (Table 15). In 2021, four previously tagged age-2 Brown Trout were recaptured and the average length-at-age-2 equaled 195 mm, 37 mm less than in 2020 (Table 15). In 2021, one age-3 Brown Trout was recaptured and this fish was 246 mm in total length (Table 15).

These findings of average lengths by age-class appear to support the previous conclusions by the Stream Scientist that very few Brown Trout reach age-4 or older on Rush Creek or Lee Vining Creek. 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 be a function of relatively low fish densities and mostly favorable summer water temperature conditions in 2017 and 2018. However, increasing densities of trout during the past several years since 2017 may have influenced the decline in growth rates observed. The record-low length-at-age values documented in 2020 and 2021 were most likely influenced by both fish densities and less than favorable summer water temperature regimes.

Table 14. Size range of PIT tagged fish recaptured in 2013-2021 by age class for Brown Trout at three electrofishing sections on Rush and Walker Creeks. NOTE: years omitted if no fish were caught.

Section	Cohort	Size Range (mm)	Average Length (mm)
Upper Rush	Age-1	**2021** = 126-185 **2020** = 124-167 **2019** = 128-202 **2018** = 158-232 **2017** = 224-264 **2016** = 192-237 **2015** = 169-203	**2021** = 154 2020 = 145 **2019** = 173 **2018** = 193 **2017** = 243 **2016** = 208 **2015** = 187
	Age-2	**2021** = 174-233 **2020** = 209-235 **2019** =203-251 **2018** = 236-305 **2017** = 284-337 **2016** = 289* **2015** = 205-242	**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	**2014** = 298	**2014** = 298
Bottomlands	Age-1	**2021** = 155 **2020** = 141-187 **2019** = 133-196 **2018** = 166-199 **2017** = 189-246 **2016** = 172-217 **2015** = 150-181	**2021** = 155 **2020** = 155 **2019** = 168 **2018** = 181 **2017** = 221 **2016** = 197 **2015** = 169
	Age-2	**2021** = 186 **2019** = 219 **2018** = 251-287 **2015** = 197-239 **2014** = 192 **2013** = 156-196	**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 14 (continued).

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

Table 15. Size range of PIT tagged fish recaptured in 2013-2021 by age class for Brown Trout and Rainbow Trout on Lee Vining Creek. NOTE: years omitted if no fish were caught.

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

Summer Water Temperature

During the past ten 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) and a Dry RY in 2021. 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 2021, a Dry RY with GLR storage levels only three to seven 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 above 70°F in six of the seven Rush Creek monitoring locations (Table 16). At all six of these monitoring locations, the numbers of days with water temperatures above 70°F were the highest ever recorded at these stations (Table 16). In July and August, four of the temperature monitoring locations recorded peak temperatures >75°F and two of these locations (Above Parker and County Road) experienced peaks ≥76°F. Daily mean temperatures and average daily maximum temperatures were the highest recorded at all Rush Creek temperature monitoring locations in 2021 since these data were collected (Table 16).

Similar to the 2013-2020 annual reports, 2021 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 17). 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 2021 varied from zero to 33 days in Rush Creek out of the 92-day period from July 1 to September 30 (Table 17). The range of the number of good thermal days in 2021 was less than the four to 42 days good thermal days in 2020 and the 62 to 76 good thermal days recorded in 2019 (Table 17). For all Rush Creek monitoring locations, the number of days classified as “fair” potential growth days in 2021 ranged from 30 to 55 days (Table 17). In 2021, poor potential growth days and bad thermal days ranged from four days at Dam Site to 54 days at Top of MGORD; the four days of poor potential growth at Dam Site was the first time this site has recorded average daily temperatures in this category (Table 17). 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 17). 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 2013-2020 data, the diurnal temperature fluctuations for July, August and September 2021 were characterized by the one-day maximum fluctuation that occurred each month and by monthly averages (Table 18). Also, for each temperature monitoring location, the highest average diurnal fluctuations over consecutive 21-day durations were determined (Table 18). The diurnal fluctuations throughout the summer of 2021 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 18). 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 2021 as recorded at the Above Parker, Below Narrows and County Road temperature monitoring locations (Table 18). These same three temperature monitoring locations also had 21-day durations with diurnal fluctuations exceeding 12.6°F during the summer of 2020 (Table 18).

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 2021. 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 19). The values from 2013 - 2020 were also included to better illustrate the variability that occurred at all the temperature monitoring locations (Table 19). The 2021 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 19). At the Top of MGORD, hourly water temperatures exceeded 66.2°F 50% of the time and at the three downstream monitoring locations, hourly water temperatures of 66.2°F were exceeded 29% to 43% of the 92-day period (Table 19). In 2021, for the temperature monitoring locations from the Top of MGORD to County Road, the months of July or August had the highest number of hours where temperatures exceeded 66.2°F (Table 19). For August, temperatures exceeding 66.2°F occurred for 79% of the month at the Top of MGORD monitoring location (Table 19). Late July through August was also when these temperature monitoring locations experienced their highest 21-day diurnal fluctuations, including levels detrimental to trout growth (Werley et al. 2007).

In 2021, 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 20% decrease in the number of days with temperatures exceeding 70°F and 50% more good growth thermal days immediately downstream of the tributaries' accretions (Tables 16-19). 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 and large diurnal fluctuations were documented. Conversely, the At Damsite water temperature monitoring location continued to provide data documenting the thermal loading in Rush Creek as flow passes through GLR and the MGORD (Tables 16-19). This thermal loading during the summer of 2021 included a 4.7°F increase in daily mean temperature (3.4°F in 2020) and a 7.7°F increase in average daily maximum temperature (6.8°F in 2020) (Table 16). The number of days with temperatures >70°F was zero days At Damsite and 44 days at Bottom of MGORD (Table 16).

Summer water temperatures in Lee Vining Creek were all within the range of good growth potential during 2021. 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 2021 data.

Table 16. Summary of water temperature data during the summer of RY 2021 (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 2013-2020 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. – At Damsite	2016 = 58.9 2017 = 58.1 2018 = 59.7 2019 = 57.8 2020 = 59.8 **2021 = 61.1**	2016 = 58.3 2017 = 57.5 2018 = 58.9 2019 = 57.4 2020 = 59.0 **2021 = 60.2**	2016 = 59.5 2017 = 58.7 2018 = 60.4 2019 = 58.5 2020 = 60.7 **2021 = 62.1**	2016 = 0 2017 = 0 2018 = 0 2019 = 0 2020 = 0 **2021 = 0**	2016 = 3.2 2017 = 2.1 2018 = 2.4 2019 = 2.3 2020 = 4.7 **2021 = 3.3**	8/11/16 9/07/17 8/22/18 8/21/19 7/10/20 **8/3/21**
Rush Ck. – Top of MGORD	2013 = 63.1 2014 = 64.8 2015 = 64.4 2016 = 63.8 2017 = 57.0 2018 = 60.7 2019 = 58.5 2020 = 63.2 **2021 = 65.9**	2013 = 62.7 2014 = 64.6 2015 = 64.1 2016 = 63.0 2017 = 56.5 2018 = 59.6 2019 = 57.2 2020 = 62.1 **2021 = 65.2**	2013 = 63.7 2014 = 65.0 2015 = 64.8 2016 = 64.7 2017 = 58.1 2018 = 61.9 2019 = 59.9 2020 = 64.4 **2021 = 66.8**	2013 = 0 2014 = 0 2015 = 0 2016 = 0 2017 = 0 2018 = 0 2019 = 0 2020 = 0 **2021 = 5**	2013 = 3.4 2014 = 3.9 2015 = 2.1 2016 = 6.5 2017 = 5.4 2018 = 6.7 2019 = 8.2 2020 = 6.4 **2021 = 6.5**	7/09/13 8/13/14 7/03/15 7/07/16 9/07/17 8/20/18 8/10/19 7/02/20 **7/13/21**
Rush Ck. – Bottom MGORD	2013 = 63.2 2014 = 64.8 2015 = 64.4 2016 = 63.8 2017 = 57.1 2018 = 61.0 2019 = 58.7 2020 = 63.2 **2021 = 65.8**	2013 = 60.9 2014 = 62.9 2015 = 62.3 2016 = 61.8 2017 = 56.5 2018 =58.9 2019 = 56.6 2020 = 60.5 **2021 = 63.4**	2013 = 67.1 2014 = 68.5 2015 = 68.0 2016 = 66.9 2017 = 58.5 2018 = 63.9 2019 = 61.3 2020 = 67.5 **2021 = 69.8**	2013 = 1 2014 = 20 2015 = 20 2016 = 1 2017 = 0 2018 = 0 2019 = 0 2020 = 17 **2021 = 44**	2013 = 9.0 2014 = 8.3 2015 = 8.4 2016 = 8.0 2017 = 6.4 2018 = 8.7 2019 = 8.1 2020 = 10.0 **2021 = 8.5**	7/09/13 7/13/14 7/06/15 7/04/16 9/07/17 7/05/18 8/10/19 8/03/20 **7/24/21**
Rush Ck. – Old Highway 395 Bridge/Upper Rush section	2013 = 62.6 2014 = 64.0 2015 = N/A 2016 = 63.5 2017 = 59.0 2018 = 60.9 2019 = 58.7 2020 = 62.6 **2021 = 65.0**	2013 = 58.8 2014 = 60.5 2015 = N/A 2016 = 60.1 2017 = 57.5 2018 = 58.0 2019 = 56.1 2020 = 58.5 **2021 = 61.2**	2013 = 68.7 2014 = 69.8 2015 = N/A 2016 = 68.8 2017 = 61.0 2018 = 65.3 2019 = 62.3 2020 = 68.4 **2021 = 70.8**	2013 = 40 2014 = 51 2015 = N/A 2016 = 47 2017 = 0 2018 = 0 2019 = 0 2020 = 30 **2021 = 63**	2013 = 13.5 2014 = 13.3 2015 = N/A 2016 = 12.5 2017 = 7.6 2018 = 10.9 2019 = 10.7 2020 = 14.0 **2021 = 12.8**	7/09/13 7/13/14 N/A 7/11/16 9/07/17 7/10/18 9/14/19 8/03/20 **8/02/21**
Rush Ck. – Above Parker	2016 = 63.2 2017 = 59.0 2018 = 60.9 2019 = 58.4 2020 = 62.2 **2021 = 64.4**	2016 = 58.8 2017 = 57.2 2018 = 57.2 2019 = 55.5 2020 = 57.1 **2021 = 59.6**	2016 = 69.4 2017 = 61.9 2018 = 66.3 2019 = 62.3 2020 = 68.6 **2021 = 70.8**	2016 = 55 2017 = 0 2018 = 0 2019 = 0 2020 = 40 **2021 = 61**	2016 = 13.7 2017 = 8.6 2018 = 13.4 2019 = 11.8 2020 = 16.1 **2021 = 14.4**	7/11/16 9/08/17 7/10/18 9/14/19 8/03/20 **8/02/21**

Table 16 (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**	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**	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**	2013 = 24 2014 = 46 2015 = 0 2016 = 34 2017 = 0 2018 =0 2019 = 0 2020 = 16 **2021 = 49**	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**	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**
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**	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**	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**	2013 = 7 2014 = 24 2015 = 2 2016 = 32 2017 = N/A 2018 = N/A 2019 = 0 2020 = 42 **2021 = 57**	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**	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**

Table 17. Classification of 2013-2021 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. – At Damsite	2016 = 69 (75%) 2017 = 88 (96%) 2018 = 53 (58%) 2019 = 76 (83%) 2020 = 42 (46%) **2021 = 33 (36%)**	2016 = 23 (25%) 2017 = 4 (4%) 2018 = 39 (42%) 2019 = 16 (17%) 2020 = 50 (54%) **2021 = 55 (60%)**	2016 = 0 2017 = 0 2018 = 0 2019 = 0 2020 = 0 **2021 = 4 (4%)**	2016 = 0 2017 = 0 2018 = 0 2019 = 0 2020 = 0 **2021 = 0**
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**	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%)**	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%)**	2013 = 18 (20%) 2014 = 48 (52%) 2015 = 60 (65%) 2016 = 33 (36%) 2017 = 0 2018 = 0 2019 = 0 2020 = 24 (26%) **2021 = 54 (59%)**

Table 17 (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%)**	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%)**	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%**	2013 = 23 (25%) 2014 = 54 (59%) 2015 = 59 (64%) 2016 = 36 (39%) 2017 = 0 2018 = 0 2019 = 0 2020 = 20 (22%) **2021 = 35 (38%)**
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%)**	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%)**	2013 = 33 (36%) 2014 = 27 (29%) 2015 = N/A 2016 = 19 (21%) 2017 = 0 2018 = 0 2019 = 0 2020 = 17 (18%) **2021 = 11 (12%)**	2013 = 4 (4%) 2014 = 33 (36%) 2015 = N/A 2016 = 33 (36%) 2017 = 0 2018 = 0 2019 = 0 2020 = 10 (11%) **2021 = 27 (29%)**
Rush Ck. – Above Parker Ck.	2016 = 17 (18%) 2017 = 65 (71%) 2018 = 28 (30%) 2019 = 67 (73%) 2020 = 24 (26%) **2021 = 30 (33%)**	2016 = 26 (28%) 2017 = 27 (29%) 2018 = 64 (70%) 2019 = 25 (27%) 2020 = 41 (45%) **2021 = 34 (37%)**	2016 = 24 (26%) 2017 = 0 2018 = 0 2019 = 0 2020 = 21 (23%) **2021 = 10 (11%)**	2016 = 25 (27%) 2017 = 0 2018 = 0 2019 = 0 2020 = 10 (11%) **2021 = 18 (20%)**
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%)**	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%)**	2013 = 6 (7%) 2014 = 18 (20%) 2015 = 22 (24%) 2016 = 16 (17%) 2017 = 0 2018 = 0 2019 = 0 2020 = 2 (2%) **2021 = 10 (11%)**	2013 = 0 2014 = 3 (3%) 2015 =2 (2%) 2016 = 2 (2%) 2017 = 0 2018 = 0 2019 = 0 2020 = 1 (1%) **2021 = 17 (18%)**
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%)**	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%)**	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%)**	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%)**

Table 18. Diurnal temperature fluctuations in Rush Creek for 2021: 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: 2020 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. – At Damsite	Max = 2.3°F (4.7) Ave = 1.7°F (1.8)	Max = 3.3°F (2.6) Ave = 2.0°F (1.9)	Max = 2.7°F (3.0) Ave = 1.9°F (1.4)	2.1 °F (1.9) Aug 2 – 22
Rush Ck. – Top of MGORD	Max = 6.5°F (6.4) Ave = 2.3°F (3.3)	Max = 3.4°F (4.5) Ave = 1.5°F (2.7)	Max = 2.2°F (2.5) Ave = 1.0°F (0.8)	1.6°F (3.4) July 30 – Aug 19
Rush Ck. – Bottom MGORD	Max = 8.5°F (9.7) Ave = 5.4°F (6.9)	Max = 8.4°F (10.0) Ave = 7.3°F (7.6)	Max = 7.4°F (9.2) Ave = 6.5°F (6.3)	7.5°F (8.2) Aug 1 – 21
Rush Ck. – Old Hwy 395 Bridge	Max = 11.4°F (13.3) Ave = 8.1°F (10.1)	Max = 12.8°F (14.0) Ave = 10.9°F (10.4)	Max = 12.0°F (12.6) Ave = 9.8°F (9.1)	10.6°F (11.6) July 27 – Aug 16
Rush Ck. – Above Parker Ck.	Max = 13.5°F (14.2) Ave = 9.6°F (12.0)	Max = 14.4°F (16.1) Ave = 12.7°F (11.9)	Max = 13.7°F (14.2) Ave = 11.4°F (10.3)	12.8°F (13.1) Aug 1 – 21
Rush Ck. – below Narrows	Max = 13.9°F (14.5) Ave = 10.0°F (12.3)	Max = 14.9°F (15.7) Ave = 12.9°F (12.2)	Max = 14.9°F (15.1) Ave = 12.3°F (11.6)	12.7°F (13.2) Aug 13 – Sept 2
Rush Ck. – County Road	Max = 15.6°F (17.5) Ave = 12.8°F (14.9)	Max = 16.7°F (18.2) Ave = 15.5°F (14.4)	Max = 17.4°F (17.5) Ave = 13.9°F (12.8)	15.9°F (15.5) Aug 19 – Sept 8

Table 19. 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 - 2021. 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. – At Damsite	2016 = 0 hrs 2017 = 0 hrs 2018 = 0 hrs 2019 = 0 hrs 2020 = 0 hrs **2021 = 0 hrs**	2016 = 0 hrs 2017 = 0 hrs 2018 = 0 hrs 2019 = 0 hrs 2020 = 0 hrs **2021 = 0 hrs**	2016 = 0 hrs 2017 = 0 hrs 2018 = 0 hrs 2019 = 0 hrs 2020 = 0 hrs **2021 = 0 hrs**	2016 = 0 hrs 2017 = 0 hrs 2018 = 0 hrs 2019 = 0 hrs 2020 = 0 hrs **2021 = 0 hrs**
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%)**	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%)**	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%)**	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%)**

Table 19 (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%)**	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%)**	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%)**	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%)**
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%)**	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%)**	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%)**	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%)**
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%)**	2016 = 269 hrs (36%) 2017 = 0 hrs 2018 = 68 hrs (9%) 2019 = 11 hrs (2%) 2020 = 257 hrs (35%) **2021 = 316 hrs (42%)**	2016 = 65 hrs (9%) 2017 = 14 hrs (2%) 2018 = 44 hrs (6%) 2019 = 27 hrs (4%) 2020 = 73 hrs (10%) **2021 = 122 hrs (17%)**	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%)**
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%)**	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%)**	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%)**	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%)**
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%)**	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%)**	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%)**	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%)**

Discussion

The 2021 sampling year documented fish populations responding with low growth rates and poor condition factors in Rush Creek to the Dry RY and thermally challenging water temperature conditions during the summer months. The 2021 sampling was also marked by being the final year in which LADWP staff participated in the annual fisheries sampling. On October 1, 2021; the SWRCB issued Order 2021-86, which amended LADWP's license and signaled the start of the 10-year post-settlement monitoring period. During the next 10 years, all monitoring activities will be conducted by consultants, with oversight from the monitoring advisory team (MAT).

Thus, this report's Discussion is focused on the trout populations' response to the Dry RY2021, the unfavorable summer water temperatures and the resulting low growth rates and poor condition factors of fish. An examination of Lee Vining air temperature is also made, in context of how air temperatures influence water temperatures.

2021 Summer Water Temperature, Fish Densities and Trout Growth Rates

The 2021 Brown Trout growth, as measured by weight gains of PIT tagged fish, between age-0 and age-1 in the Upper Rush and Bottomlands sampling sections were extremely low (Table 20). In the Upper Rush section, the weight gain of age-1 fish was 27 g in 2021, the second lowest average weight gain recorded for this section and 23 g less than the 14-year long-term average (Table 20). Similarly, in the Bottomlands section, the 2021 weight gain of age-1 Brown Trout was 26 g, 16 g lower than the long-term average and the lowest value recorded since the first two years of the five-year drought period (Table 20).

The Upper Rush section's age-2 recaptures gained an average of 42 g between 2020 and 2021; a growth rate 44 g lower than the average growth rate (86 g) for the 13 years of available tag return data (Table 21). The 2021 average growth rate of age-2 recaptures in Upper Rush was the lowest value recorded for this section since the first two years of the five-year drought period. Only one PIT-tagged age-2 Brown Trout was recaptured in the Bottomlands section of Rush Creek in 2021 and this fish's weight gain on 33 g was 7 g lower than the long-term average of 40 g (Table 21).

We also know growth rates were extremely low in the Upper Rush section from the PIT tag recaptures of age-1 fish that were 126 mm, 137 mm in length and 139 mm in length. In addition, a total of 40 presumed age-1 Brown Trout caught in Upper Rush in 2021 were between 125 and 135 mm in length. Similarly, a total of 25 Brown Trout caught in the Bottomlands section in 2021 were between 125 and 135 mm in length.

Studies have determined that trout growth in streams is a complex interaction of population density, water temperature and food availability (Baerum et al. 2013). Conditions in Rush Creek during 2017 were favorable for the record growth we documented, with respect to multiple variables, especially extremely low fish densities and cool summer water temperatures. Then in

2018 growth rates dropped with mostly favorable summer water temperatures, but Brown Trout densities increased in all monitoring sections. In 2019, the wet-year runoff resulted in more favorable summer water temperatures than 2018, yet growth rates continued to drop as fish densities increased. Density-dependent growth in stream-dwelling salmonids is well researched and there's broad support for the hypothesis that density-dependent growth occurs at low population densities, probably due to exploitive completion (Grant and Imre 2005). One study used controlled reaches of a small stream and determined that population density affected growth in trout parr (yearlings and older) and that competition and population regulation was not just limited to early life-stages, as suggested by other researchers (Bohlin et al. 2002). Another analysis used data collected from 19 trout populations (six species and 16 different studies) and determined that 15 of the 19 populations showed evidence of decreased growth rates with increasing densities (Grant and Imre 2005). This analysis was focused primarily on age-0 trout (Grant and Imre 2005). For Upper Rush, 16 years (2006-2021) of age-0 Brown Trout and total Brown Trout population estimates were plotted versus the average weights of age-0 Brown Trout from those sample years (Figure 21). Trend lines through each of the population estimates suggests that density-dependent growth of age-0 fish does occur in the Upper Rush section (Figure 21). However; the 2021 age-0 and total population estimates were relatively low (age-0 estimate 2nd lowest and total estimate 4th lowest of the 16 years); yet the average weight of the age-0 Brown Trout was relatively low too (7.9 g) (Figure 21). This suggests that another factor besides densities of fish factored into 2021's low growth rates, most likely the summer water temperatures.

Table 20. Annual growth rate (g) for PIT tagged or fin-clipped age-0 to age-1 Brown Trout in two sections of Rush Creek by year. N/A = not available

Age Class	Growth Years	Upper Rush Growth (g)	Bottomlands Growth (g)	Fin clip or PIT Tag
Age-0 to Age-1	2006-2007	32	N/A	Ad Clip
	2008-2009	51	43	Ad Clip
	2009-2010	48	40	PIT Tag
	2010-2011	48	36	PIT Tag
	2011-2012	33	25	PIT Tag
	2012-2013	35	25	PIT Tag
	2013-2014	N/A	N/A	N/A
	2014-2015	55	41	PIT Tag
	2015-2016	77	62	PIT Tag
	2016-2017	129	96	PIT Tag
	2017-2018	56	42	PIT Tag
	2018-2019	39	38	PIT Tag
	2019-2020	21	29	PIT Tag
	2020-2021	**27**	**26**	PIT Tag
	Long-term Ave.	*50*	*42*	PIT Tag

Table 21. Annual growth rate (g) for PIT tagged or fin-clipped age-1 to age-2 Brown Trout in two sections of Rush Creek by year. N/A = not available

Age Class	Growth Years	Upper Rush Growth (g)	Bottomlands Growth (g)	Fin clip or PIT Tag
Age-1 to Age-2	2008-2009	N/A	N/A	Ad Clip
	2009-2010	70	54	PIT Tag
	2010-2011	73	32	PIT Tag
	2011-2012	42	28	PIT Tag
	2012-2013	42	22	PIT Tag
	2013-2014	N/A	29	PIT Tag
	2014-2015	69	62	PIT Tag
	2015-2016	176	N/A	PIT Tag
	2016-2017	239	N/A	PIT Tag
	2017-2018	66	55	PIT Tag
	2018-2019	71	44	PIT Tag
	2019-2020	55	N/A	PIT Tag
	2020-2021	**42**	**33**	PIT Tag
	Long-term Ave.	*86*	*40*	

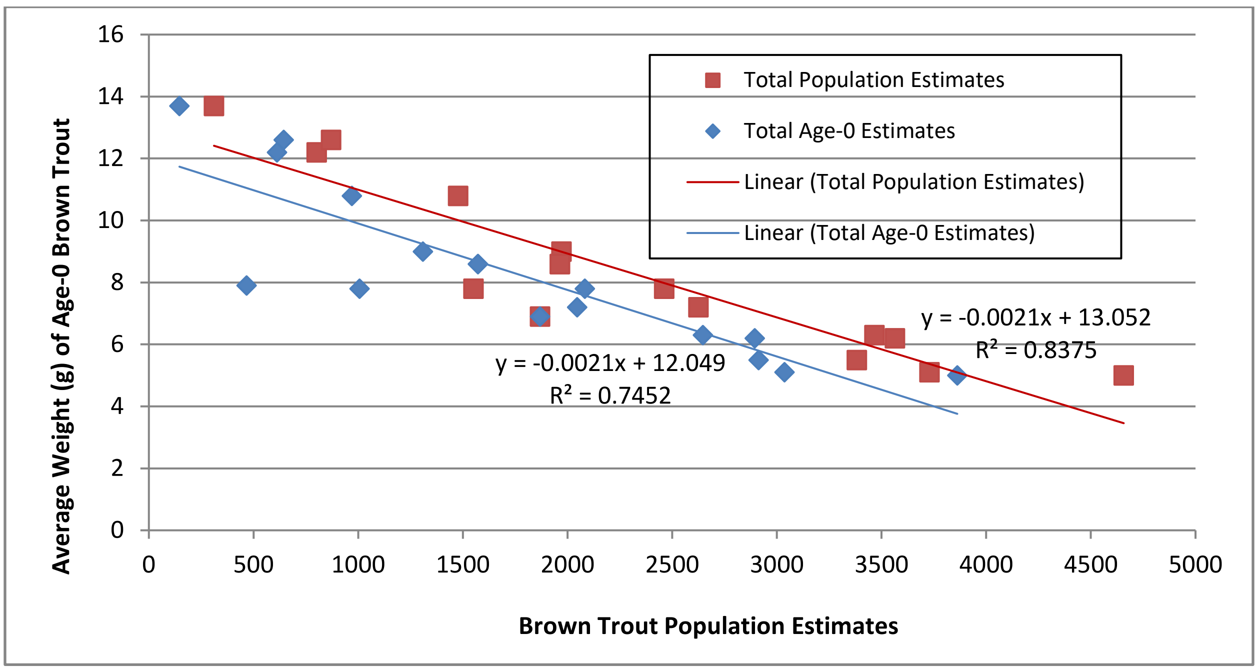

Figure 21. Relationship between average weights of age-0 brown trout and population estimates (age-0 and all trout) in the Upper Rush sampling section, 2006-2021.

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) have been 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 7.2ºF 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 15ºC (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 14.4ºF 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).

As previously described, the 2021 water temperatures in Rush Creek downstream of GLR were unfavorable at all temperature monitoring locations for some periods of the summer, defined as the months of July, August and September. These conditions occurred when GLR’s storage elevation was three to seven feet above the 7,100-foot elevation recommended in the Synthesis Report as a minimum storage level 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 2020 summer storage levels were ≥20 feet higher than 7,100 feet and that 2021’s storage levels were three to seven feet higher than 7,100 feet; yet both 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 its actual storage volume is significantly less than 47,000 acre-feet, thus the 1993 modeled predictions of storage level versus water temperature are 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 32 years (Table 22). 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 22). The average maximum temperature in the 1990's equaled 80.4°F and in the first two years of the 2020's, the average maximum temperature was 87.1°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 temperatures ≥90°F versus in the most recent six years (2016-2021), five of the years experienced at least 10 days with maximum temperatures ≥90°F (Table 22).

Studies have shown that a combination of air temperature, direct solar radiation, basin-specific hydrology, channel bed morphology, 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's 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 (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 bounced back quickly in the numbers of fish, their growth rates and condition factors. Thus, changing climate and variable snowpack conditions in the eastern Sierras will most likely dictate the long-term fate and viability of Rush Creek's Brown Trout fishery.

Table 22. Thirty-two 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 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**

Table 22 (continued).

YEAR	Ave Max Temp (ºF)	Ave Min Temp (ºF)	Ave Ave Temp (ºF)	Number of Days ≥90ºF
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
2020's Averages	**87.1**	**47.6**	**65.5**	**20.5**

Apparent Survival Rates

Apparent survival rates of age-1 Brown Trout were calculated with the following equation: [# age-1 recaps in 2021/capture probability of age-1 fish] ÷ [# age-0 tagged in 2020 - # shed tags]. For mark-recapture sections, capture probabilities were derived from the recapture run data: # of recaptures/# of captures. Compared to the 2020 survival rates; the 2021 apparent survival rates increased by 33% in Upper Rush Creek and decreased by 1.3% in the Bottomlands section of Rush Creek (Table 23). Between 2020 and 2021, the age-1 Brown Trout apparent survival rate increased by 27.9% in the Lee Vining Creek main channel section (Table 23). Walker Creek's apparent survival rate decreased by 12.1% between 2020 and 2021 (Table 23).

Table 23. 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 2021	No. Age-0 Tagged in 2020	No. Shed Tags	Apparent Survival Rate
Rush – Upper	0.15	20	242	1	2016 = 22.7% 2017 = 106% 2018 = 50.2% 2019 = 17.4% 2020 = 22.2% **2021 = 55.3%**
Rush - Bottomlands	0.36	2	65	0	2016 = 9.7% 2017 = 72.3% 2018 = 66.8% 2019 = 12.0% 2020 = 9.8% **2021 = 8.5%**
Walker Creek	0.83	26	92	1	2016 = 37.8% 2017 = 7.0% 2018 = N/A 2019 = 19.8% 2020 = 46.5% **2021 = 34.4%**
Lee Vining Creek	0.39	22	102	0	2016 = 46.3% 2017 = 4.8% 2018 = 70.6% 2019 = 40.0% 2020 = 27.4% **2021 = 55.3%**

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.

The PIT tagging program was continued during the September 2021 sampling and tags were implanted primarily in age-0 fish and 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 (or 12 inches). Continuation of the PIT tagging program is recommended as the fisheries monitoring program moves into the post-settlement phase.

Trout size classes (0-124, 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.

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 in mid-September. Allowances for flow variances to allow for safe wading conditions and effective sampling were included in the new SWRCB Order 2021-86 (the amended license).

The eastern Sierras experienced a wet December of 2021 (168% of normal), followed by a very dry January of 2022 (14% of normal) and the overall snow pack near Mammoth was approximately 90% of normal on 2/15/22. The extremely dry conditions extended through February and March of 2022. The preliminary April 1st forecast for RY2022 is for a Dry year-type. Entering 2022, the water level in GLR was at the second-lowest level it's been in the past nine years (Figure 4). If GLR fails to enter the summer of 2022 at a nearly full level, Rush Creek downstream of GLR will most likely experience less than favorable summer water temperature conditions, which will likely translate into a third consecutive year of poor growth rates and condition factors.

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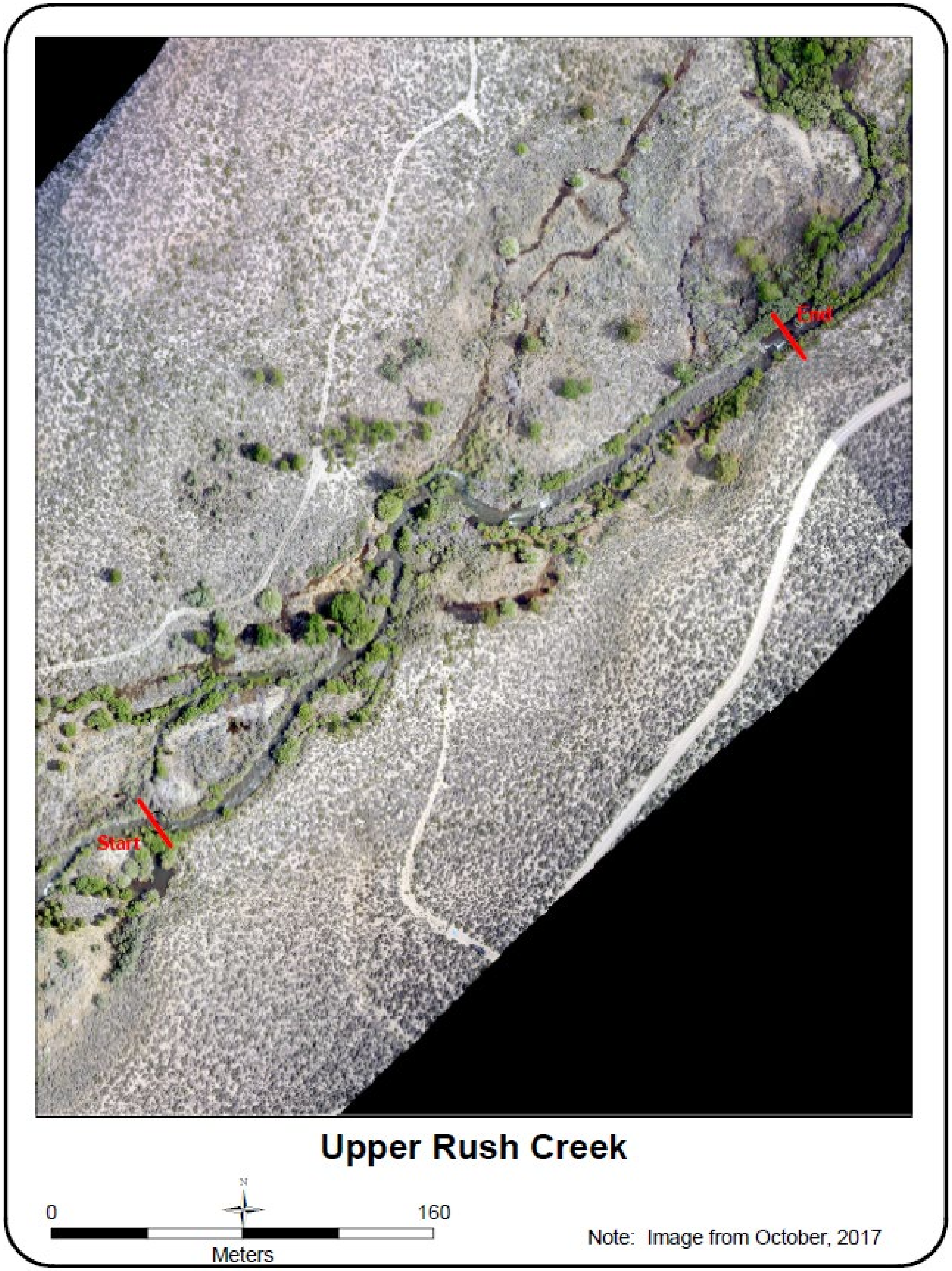
Appendices for the 2021 Mono Basin Annual Fisheries Report

Appendix A: Aerial Photographs of Annual Sample Sites on Rush, Walker and Lee Vining Creeks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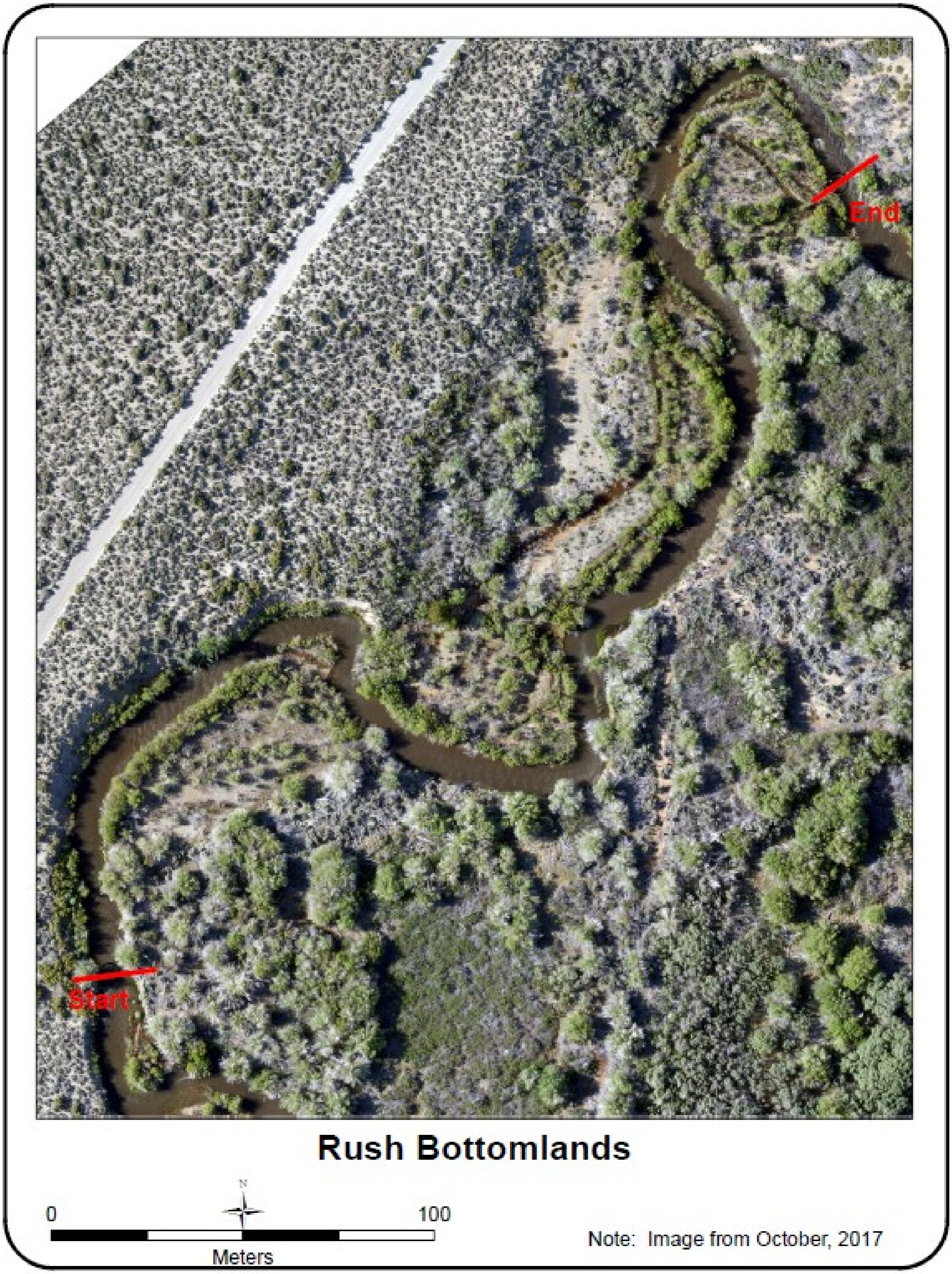
Start
Upper Rush Creek
N
0
160
Meters
Note: Image from October, 2017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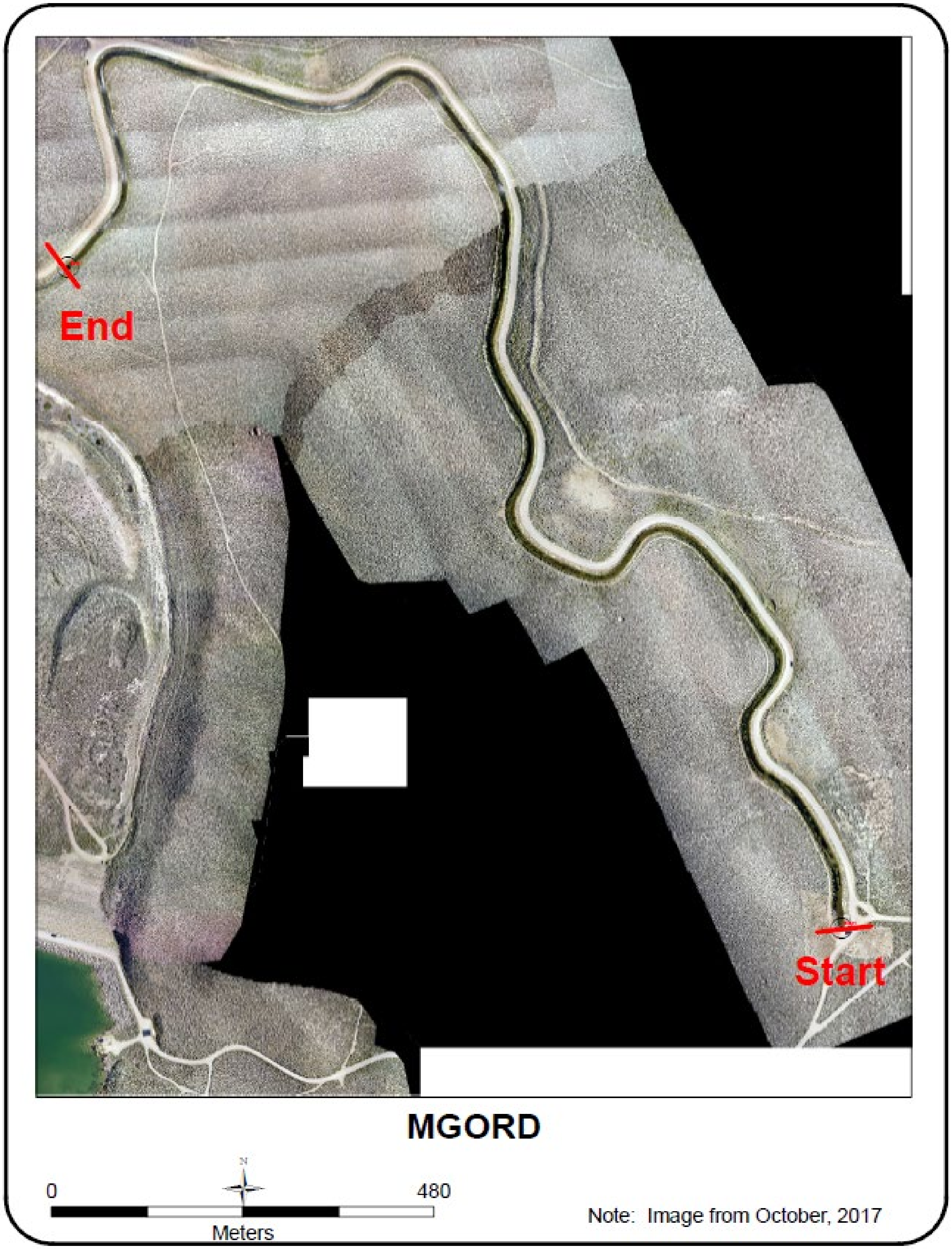
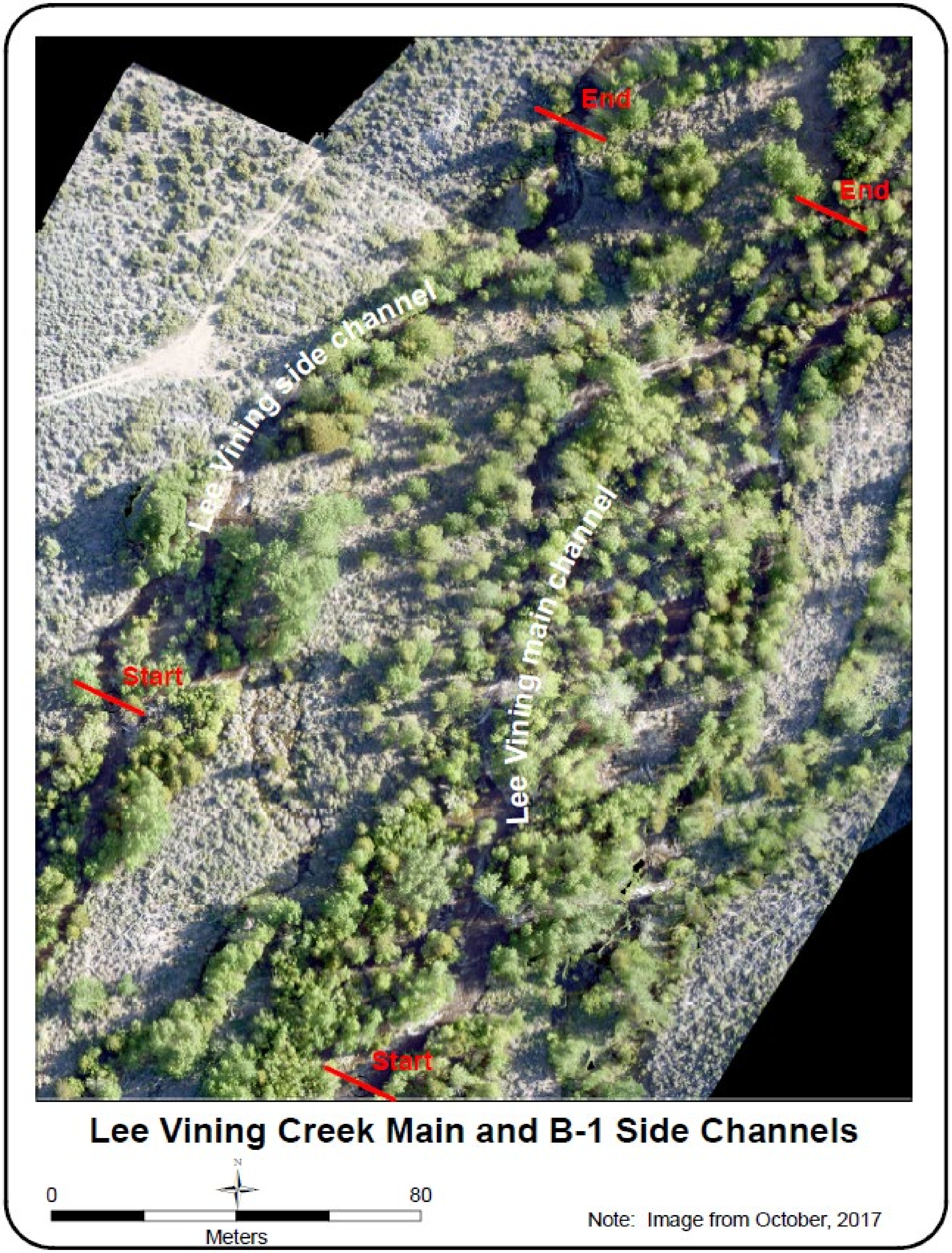
Start
Rush Bottomlands
0
100
N
Meters
Note: Image from October, 2017


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


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



Lee Vining Creek Main and B-1 Side Channels

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 2020

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

Appendix C: Table of PIT-tagged Fish Recaptured during September 2021 Sampling

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2021 Recapture	Location of Initial Capture and Tagging	Comments
9/7/2021	BNT	325	338	989001006111571	UpperRush	Upper Rush	
9/7/2021	BNT	220	109	989001028113996	UpperRush	Upper Rush	
9/7/2021	BNT	174	50	989001031371645	UpperRush	Upper Rush	
9/7/2021	BNT	198	74	989001031371763	UpperRush	Upper Rush	
9/7/2021	BNT	233	127	989001031371775	UpperRush	Upper Rush	
9/7/2021	BNT	195	69	989001031371784	UpperRush	Upper Rush	
9/7/2021	BNT	201	72	989001031372274	UpperRush	Upper Rush	
9/7/2021	BNT	170	45	989001038116493	UpperRush	Upper Rush	
9/7/2021	BNT	173	52	989001038116496	UpperRush	Upper Rush	
9/7/2021	BNT	142	27	989001038116542	UpperRush	Upper Rush	
9/7/2021	BNT	241	166	989001038116569	UpperRush	Upper Rush	
9/7/2021	BNT	152	28	989001038116584	UpperRush	Upper Rush	
9/7/2021	BNT	137	23	989001038116604	UpperRush	Upper Rush	
9/7/2021	BNT	139	28	989001038116629	UpperRush	Upper Rush	
9/7/2021	**RBT**	185	83	989001038116661	UpperRush	Upper Rush	
9/7/2021	BNT	137	26	989001038116709	UpperRush	Upper Rush	
9/7/2021	BNT	149	33	989001038116723	UpperRush	Upper Rush	
9/14/2021	BNT	195	70	989001031371622	UpperRush	Upper Rush	
9/14/2021	BNT	183	65	989001031371765	UpperRush	Upper Rush	
9/14/2021	BNT	214	93	989001031371787	UpperRush	Upper Rush	
9/14/2021	BNT	187	54	989001031372321	UpperRush	Upper Rush	
9/14/2021	BNT	159	39	989001038116462	UpperRush	Upper Rush	
9/14/2021	BNT	126	18	989001038116478	UpperRush	Upper Rush	
9/14/2021	BNT	148	30	989001038116568	UpperRush	Upper Rush	
9/14/2021	BNT	147	31	989001038116572	UpperRush	Upper Rush	
9/14/2021	BNT	157	36	989001038116576	UpperRush	Upper Rush	
9/14/2021	BNT	145	31	989001038116586	UpperRush	Upper Rush	
9/14/2021	BNT	152	34	989001038116592	UpperRush	Upper Rush	
9/14/2021	BNT	184	56	989001038116642	UpperRush	Upper Rush	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2021 Recapture	Location of Initial Capture and Tagging	Comments
9/14/2021	BNT	166	43	989001038116682	UpperRush	Upper Rush	
9/14/2021	BNT	157	35	989001038116732	UpperRush	Upper Rush	
9/14/2021	BNT	144	29	989001038116755	UpperRush	Upper Rush	
9/8/2021	BNT	186	64	989001031371544	Bottomlands	Bottomlands	
9/8/2021	BNT	155	34	989001038116942	Bottomlands	Bottomlands	
9/15/2021	BNT	248	147	989001028114299	Bottomlands	Bottomlands	
9/15/2021	BNT	214	100	989001028114492	Bottomlands	Bottomlands	
9/15/2021	BNT	155	32	989001038116856	Bottomlands	Bottomlands	
9/9/2021	BNT	542	1377	985121021867358	MGORD	MGORD	
9/9/2021	BNT	512	1032	~~989001006111367~~	MGORD	MGORD	MORT - tag removed
9/9/2021	BNT	299	247	989001031371664	MGORD	MGORD	
9/9/2021	BNT	328	283	989001031371702	MGORD	MGORD	
9/9/2021	BNT	286	213	989001031371751	MGORD	MGORD	
9/9/2021	BNT	336	330	989001031371753	MGORD	MGORD	
9/9/2021	BNT	301	272	989001031371865	MGORD	MGORD	
9/9/2021	BNT	293	243	989001031371875	MGORD	MGORD	
9/9/2021	BNT	263	172	989001031371888	MGORD	MGORD	
9/9/2021	BNT	278	242	989001038116663	MGORD	MGORD	
9/9/2021	BNT	245	129	989001038116701	MGORD	MGORD	
9/9/2021	BNT	257	167	989001038116762	MGORD	MGORD	
9/9/2021	BNT	189	62	989001038116822	MGORD	MGORD	
9/9/2021	BNT	280	205	989001038116849	MGORD	MGORD	
9/9/2021	BNT	186	71	989001038116851	MGORD	MGORD	
9/9/2021	BNT	273	180	989001038116943	MGORD	MGORD	
9/9/2021	BNT	276	186	989001038116989	MGORD	MGORD	
9/9/2021	BNT	287	224	989001038117006	MGORD	MGORD	
9/9/2021	BNT	235	114	989001038117014	MGORD	MGORD	
9/9/2021	BNT	260	165	989001038117019	MGORD	MGORD	
9/9/2021	BNT	271	184	989001038117021	MGORD	MGORD	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2021 Recapture	Location of Initial Capture and Tagging	Comments
9/9/2021	BNT	267	171	989001038117053	MGORD	MGORD	
9/16/2021	BNT	315	285	989001028114368	MGORD	MGORD	
9/16/2021	BNT	368	438	989001028114826	MGORD	MGORD	
9/16/2021	BNT	275	200	989001031371743	MGORD	MGORD	
9/16/2021	BNT	240	137	989001031372301	MGORD	MGORD	
9/16/2021	BNT	277	191	989001038116677	MGORD	MGORD	
9/16/2021	BNT	253	155	989001038116803	MGORD	MGORD	
9/16/2021	BNT	260	165	989001038116803	MGORD	MGORD	
9/16/2021	BNT	240	132	989001038116811	MGORD	MGORD	
9/16/2021	BNT	225	94	989001038116954	MGORD	MGORD	
9/16/2021	BNT	262	175	989001038116964	MGORD	MGORD	
9/16/2021	BNT	251	149	989001038116969	MGORD	MGORD	
9/16/2021	BNT	284	208	989001038116981	MGORD	MGORD	
9/16/2021	BNT	254	144	989001038116990	MGORD	MGORD	
9/16/2021	BNT	245	155	989001038117007	MGORD	MGORD	
9/16/2021	BNT	258	170	989001038117040	MGORD	MGORD	
9/11/2021	BNT	212	110	989001028114139	Walker Ck	Walker Creek	
9/11/2021	BNT	206	88	989001028114162	Walker Ck	Walker Creek	
9/11/2021	BNT	200	75	989001028114180	Walker Ck	Walker Creek	
9/11/2021	BNT	201	75	989001028114224	Walker Ck	Walker Creek	
9/11/2021	BNT	175	50	989001031371657	Walker Ck	Walker Creek	
9/11/2021	BNT	178	53	989001031371667	Walker Ck	Walker Creek	
9/11/2021	BNT	169	40	989001031371669	Walker Ck	Walker Creek	
9/11/2021	BNT	183	55	989001031371685	Walker Ck	Walker Creek	
9/11/2021	BNT	185	65	989001031371725	Walker Ck	Walker Creek	
9/11/2021	BNT	174	44	989001031372369	Walker Ck	Walker Creek	
9/11/2021	BNT	167	44	989001031372378	Walker Ck	Walker Creek	
9/11/2021	BNT	184	58	989001031372382	Walker Ck	Walker Creek	
9/11/2021	BNT	182	51	989001031372383	Walker Ck	Walker Creek	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2021 Recapture	Location of Initial Capture and Tagging	Comments
9/11/2021	BNT	170	46	989001031372387	Walker Ck	Walker Creek	
9/11/2021	BNT	182	62	989001031372397	Walker Ck	Walker Creek	
9/11/2021	BNT	175	51	989001031372402	Walker Ck	Walker Creek	
9/11/2021	BNT	169	46	989001031372406	Walker Ck	Walker Creek	
9/11/2021	BNT	178	56	989001031372407	Walker Ck	Walker Creek	
9/11/2021	BNT	173	50	989001031372409	Walker Ck	Walker Creek	
9/11/2021	BNT	172	47	989001031372414	Walker Ck	Walker Creek	
9/11/2021	BNT	160	42	989001031372417	Walker Ck	Walker Creek	
9/11/2021	BNT	181	50	989001031372427	Walker Ck	Walker Creek	
9/11/2021	BNT	181	58	989001031372428	Walker Ck	Walker Creek	
9/11/2021	BNT	169	46	989001031372432	Walker Ck	Walker Creek	
9/11/2021	BNT	187	67	989001031372434	Walker Ck	Walker Creek	
9/11/2021	BNT	173	53	989001031372435	Walker Ck	Walker Creek	
9/11/2021	BNT	186	58	989001031372450	Walker Ck	Walker Creek	
9/11/2021	BNT	155	39	989001031372454	Walker Ck	Walker Creek	
9/11/2021	BNT	171	48	989001031372458	Walker Ck	Walker Creek	
9/11/2021	BNT	130	22	989001038117109	Walker Ck	Walker Creek	
9/11/2021	BNT	145	27	989001038117259	Walker Ck	Walker Creek	
9/11/2021	BNT	150	33	989001038117262	Walker Ck	Walker Creek	
9/11/2021	BNT	142	26	989001038117263	Walker Ck	Walker Creek	
9/11/2021	BNT	125	18	989001038117266	Walker Ck	Walker Creek	
9/11/2021	BNT	126	24	989001038117279	Walker Ck	Walker Creek	
9/11/2021	BNT	125	19	989001038117280	Walker Ck	Walker Creek	
9/11/2021	BNT	125	19	989001038117281	Walker Ck	Walker Creek	
9/11/2021	BNT	135	21	989001038117284	Walker Ck	Walker Creek	
9/11/2021	BNT	129	21	989001038117290	Walker Ck	Walker Creek	
9/11/2021	BNT	140	25	989001038117301	Walker Ck	Walker Creek	
9/11/2021	BNT	127	19	989001038117302	Walker Ck	Walker Creek	
9/11/2021	BNT	133	22	989001038117305	Walker Ck	Walker Creek	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2021 Recapture	Location of Initial Capture and Tagging	Comments
9/11/2021	BNT	153	35	989001038117309	Walker Ck	Walker Creek	
9/11/2021	BNT	141	28	989001038117314	Walker Ck	Walker Creek	
9/11/2021	BNT	154	37	989001038117322	Walker Ck	Walker Creek	
9/11/2021	BNT	121	16	989001038117324	Walker Ck	Walker Creek	
9/11/2021	BNT	137	23	989001038117327	Walker Ck	Walker Creek	
9/11/2021	BNT	144	28	989001038117328	Walker Ck	Walker Creek	
9/11/2021	BNT	138	27	989001038117345	Walker Ck	Walker Creek	
9/11/2021	BNT	127	18	989001038117346	Walker Ck	Walker Creek	
9/11/2021	BNT	145	31	989001038117349	Walker Ck	Walker Creek	
9/11/2021	BNT	152	32	989001038117351	Walker Ck	Walker Creek	
9/11/2021	BNT	147	31	989001038117354	Walker Ck	Walker Creek	
9/11/2021	BNT	148	31	989001038117355	Walker Ck	Walker Creek	
9/11/2021	BNT	143	28	989001038117356	Walker Ck	Walker Creek	
9/10/2021	BNT	182	56	989001038117184	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	156	34	989001038117187	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	131	20	989001038117193	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	155	33	989001038117205	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	158	38	989001038117212	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	157	38	989001038117221	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	157	33	989001038117227	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	158	36	989001038117230	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	136	25	989001038117234	Lee Vining CK	Lee Vining Ck	
9/10/2021	BNT	157	37	989001038117237	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	193	65	989001031371982	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	186	72	989001031372074	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	205	81	989001031372075	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	184	53	989001031372133	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	136	22	989001038117162	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	154	33	989001038117169	Lee Vining CK	Lee Vining Ck	

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2021 Recapture	Location of Initial Capture and Tagging	Comments
9/17/2021	BNT	172	48	989001038117179	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	152	32	989001038117183	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	140	26	989001038117192	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	172	44	989001038117204	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	153	32	989001038117206	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	160	39	989001038117209	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	164	35	989001038117225	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	141	24	989001038117235	Lee Vining CK	Lee Vining Ck	
9/17/2021	BNT	165	43	989001038117254	Lee Vining CK	Lee Vining Ck	
9/7/2021	BNT	163	48	989001039661041	UpperRush	Upper Rush	Shed tag, new tag implanted
9/9/2021	BNT	219	107	989001038117128	MGORD	MGORD	Shed tag, new tag implanted
9/11/2021	BNT	119	15	989001039661194	Walker Ck	Walker Creek	Shed tag, new tag implanted

Section 3

RY 2021 Mono Basin Stream Monitoring Report

RY2021
Mono Basin Stream Monitoring Report

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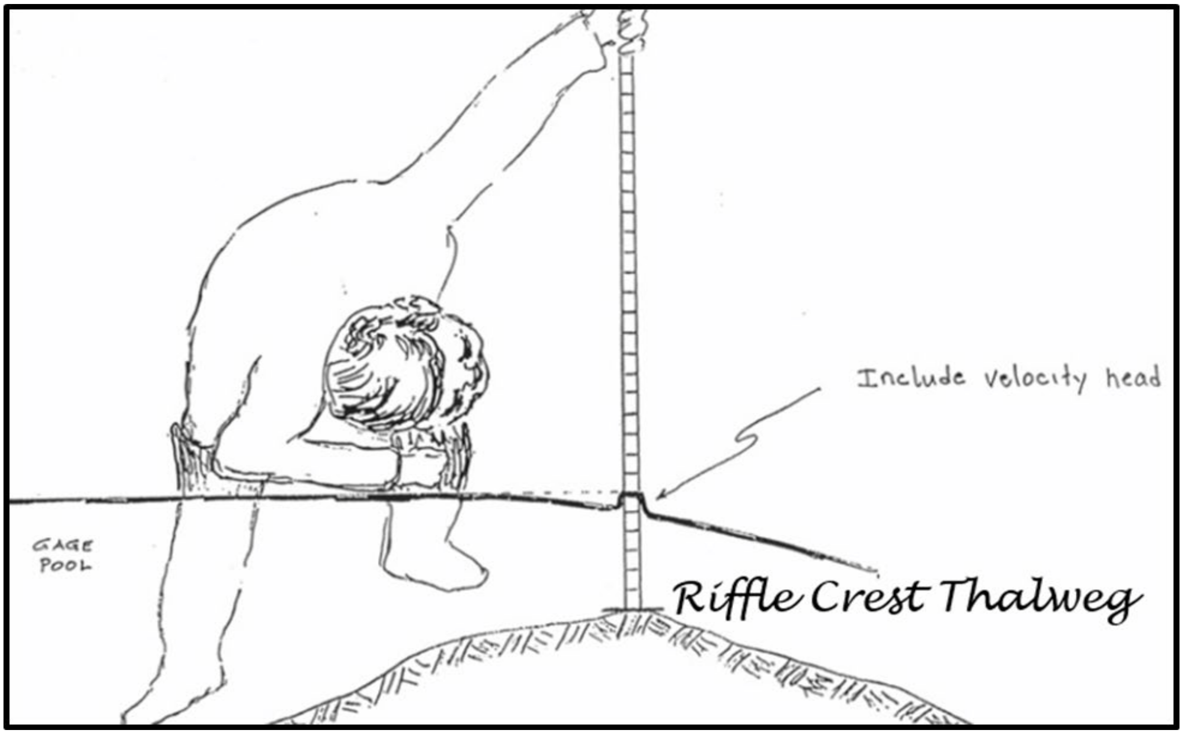
April 11, 2022

RY2021 Monitoring Objectives

The Mono Basin Stream Monitoring Project for RY2021 had two primary objectives consistent with previous runoff years (RYs): (1) continue channel morphology and riparian floodplain monitoring beginning RY2015 toward developing baseline conditions for the future monitoring program and (2) recommend monitoring techniques that objectively measure cause-effect outcomes throughout a multi-year monitoring period as unambiguously and efficiently (w/r to effort, cost, and information acquired) as possible.

The following three tasks in RY2021 were performed addressing both objectives:
(1) Survey residual pool/run depths for Lower Rush Creek mainstem from the 10 Falls downstream to the Ford and compare results to previous RYs; Construct RCT-Q rating curves for the same mainstem reach using two analytical approaches and compare results,
(2) Survey a pre-1941 channel cross-sections and construct pre-1941 RTC-Q rating curves,
(3) Inventory Annual ABI for willows and cottonwoods on the 8-Floodplain and 4-Floodplain and compare to previous RY ABI exceedence curves, and



Task No.1.
Monitoring Lower Rush Creek
RY2021 Mainstem Channel Morphology

Mainstem channel morphology was measured for the 10 Falls-to-Ford mainstem channel reach between August 23-25 RY2021 using: (1) residual pool/run depths, (2) riffle crest thalweg (RCT) depths at the active channel streamflow stage height (QActive = 240 cfs), and (3) RCT depths at the ambient stream flow (measured Q = 37 cfs). Refer to LADWP Compliance Report May2020 Section 4. Stream Monitoring Report (pp.1-9) for background information and discussion regarding the RCT, RCT-Q rating curve, and the active channel streamflow

(QActive). Once streamflow passes through the Narrows, there will be net losses primarily attributable to shallow groundwater recharge. But an accurate magnitude of this retention is difficult to estimate from one RY to the next, or sometimes even day-today. LADWP stream gaging estimated total streamflow passing the Narrows on August 23 (arrival to Mono Basin) was ~42 cfs. Measured streamflow at the old gage site immediately downstream of the former 10-Falls at 10:30 AM 23August was Q = 37 cfs (referenced as the 'ambient streamflow').

Residual Pool/Run Depth Monitoring

Each annual residual pool/run survey locates the thalweg with the lowest channelbed elevation (i.e., the greatest pool or run depth) in each hydrologic unit (HU) encountered. This bed elevation is not measured as an absolute elevation (i.e., relative to a known benchmark), but relative to the thalweg with the highest channelbed elevation in each HU, which is at the riffle crest thalweg (RCT). Maximum water depth in the pool or run subtracted from the water depth at the riffle crest thalweg equals residual pool or run depth (RPD) (Figure 1). Residual depth is therefore independent of the ambient streamflow at the time of measurement, making straightforward comparisons of inter-annual residual depth surveys possible. Robbie DiPaola of the Mono Lake Committee and Bill Trush did the fieldwork.

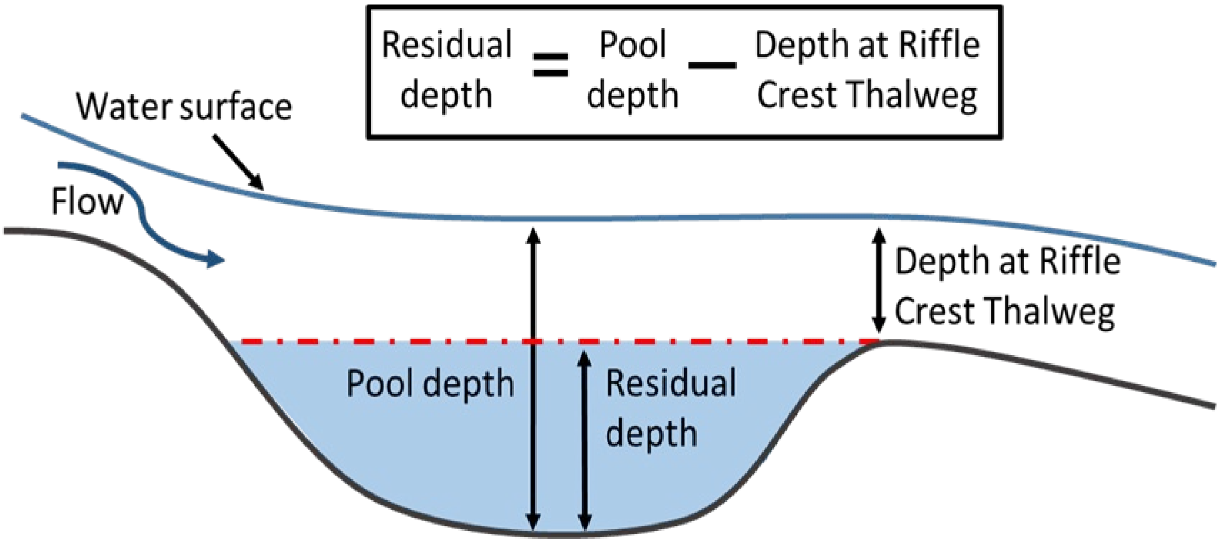


Figure 1. Measuring residual pool depth (RPD).

Residual pool/run depths (RPD) were measured in Lower Rush Creek from the Ford upstream to the former 10-Falls on August 24-25, 2021. A total of 24 HUs was measured. Change in RPD was linear between P-values = 20% and 80% when plotted as a cumulative probability exceedence curve (Figure 2). P-value is the probability of equaling or exceeding a given residual pool/run depth. Median RPD (P-value = 50%) was 2.11 ft; a P-value of 20% equaled or exceeded 2.73 ft and P-value of 80% equaled or exceeded 1.52 ft. RPD for four pools exceeded 2.73 ft generating an abrupt upturn in the exceedence curve above a P-value equaling 20% to 25%.

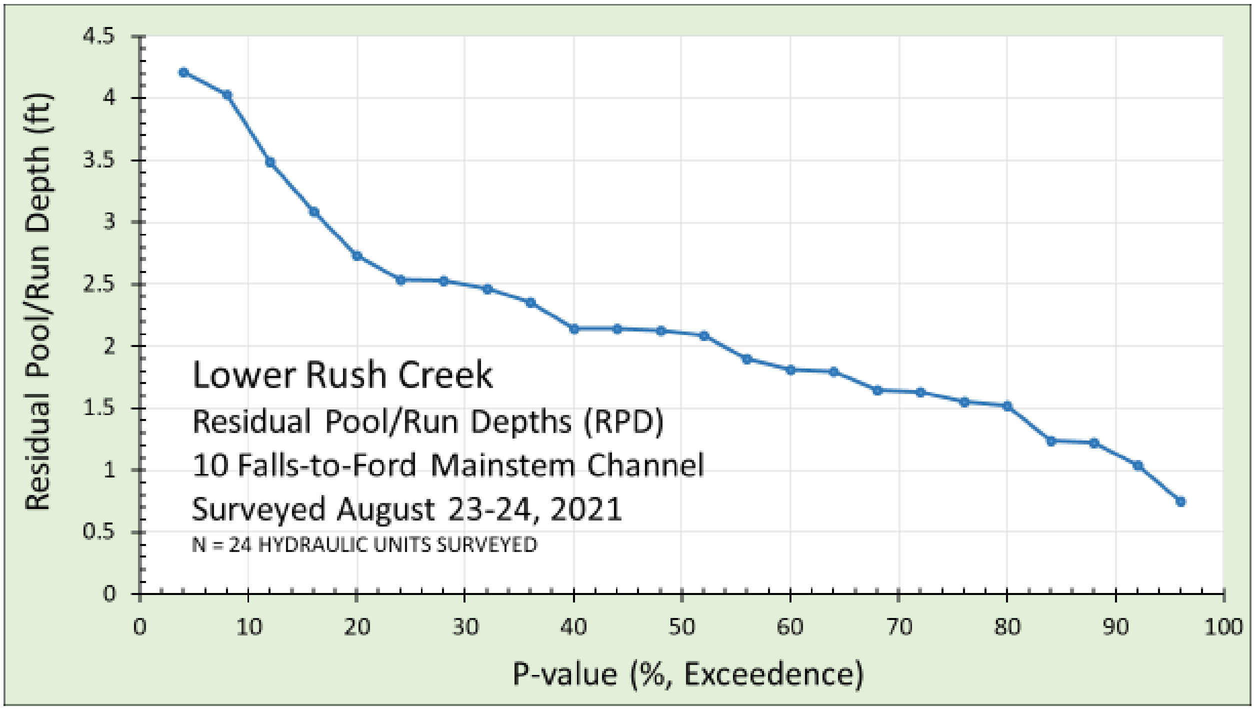


Figure 2. Residual Pool/Run Depth (RPD) exceedence curve for 10 Falls-to-Ford Lower Rush Creek in RY2021.

RPDs are presented to the hundredth foot. Achieving a hundredth foot accuracy was unlikely, but indicated care was maintained throughout the survey. The greatest uncertainties were measuring maximum pool depth in bigger pools requiring deep wading, treading water, or leaning-out from the bank while keeping the stadia rod vertical (especially in the few pools with complex log-jams).

Considerably more HU's were measured in RY2018 and RY2021 than in RY 2016 within the 10 Falls-to-Ford reach. The major RY2017 flood peak (QPK = 900+ cfs) created an additional HU, possibly two, but not 5 to 10 more. RPD surveys generally focus on pools offering good, usually deep, trout habitat; the RY2016 survey was limited to RPD > 1.0 ft deep. However, as shown in Figure 3, only a few HUs had RPDs < 1.0 ft deep in either the RY2018 or RY2021 survey. The most plausible explanation is that many runs encountered during the RY2016 survey appeared too shallow to warrant measurement. If all runs were measured, the present RY2016 exceedence would have shifted to the left. Future monitoring should include all pools and runs, even if some runs offer only poor fish habitat. Residual pool depth is a geomorphic variable as well as a fish habitat variable. Unfortunately, I have not developed a way to predict RPD exceedence curve shape for a recovered 10 Falls-to-Ford mainstem channel. Nevertheless, comparisons between runoff years and/or peak flood events are informative.

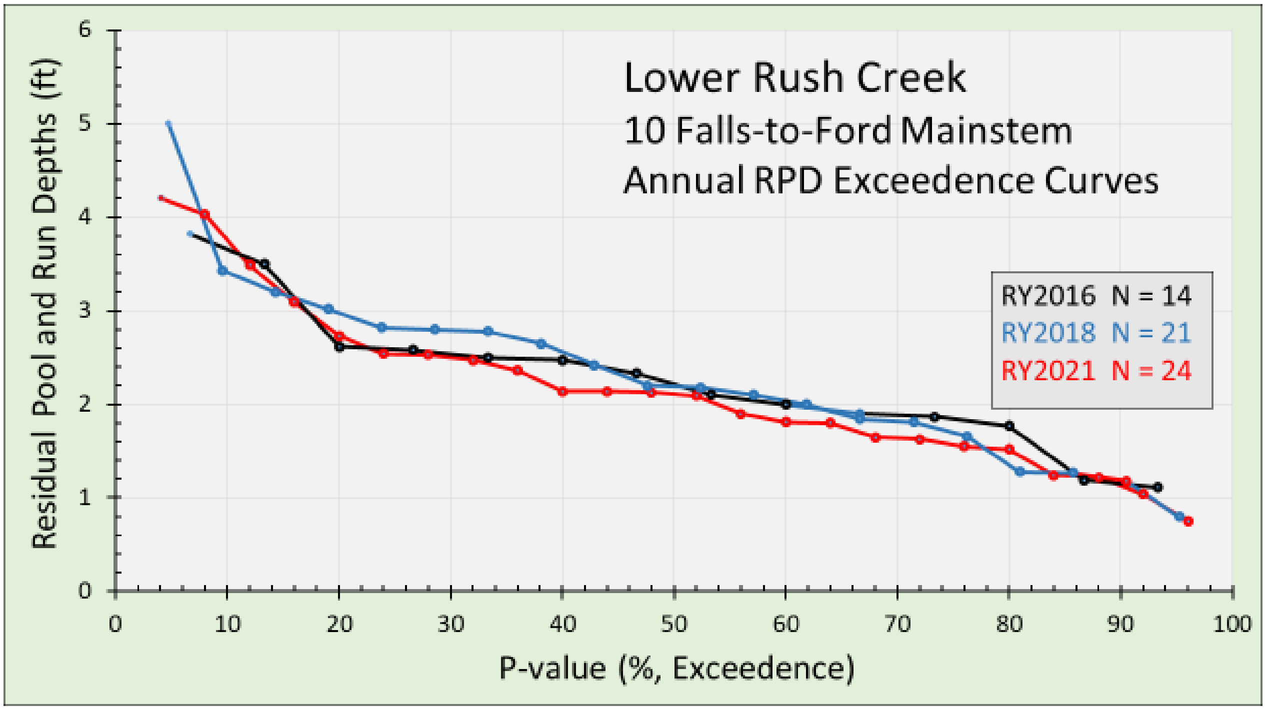


Figure 3. RPD exceedence curves for RY2016, RY2018, and RY2021 for 10 Falls-to-Ford Lower Rush Creek.

With all three RY's plotted together (Figure 3), two observations stood-out. First, the three surveys all had an abrupt upswing in their respective exceedence curves at P-values ranging between 20 to 25 percent. Second, the RPD exceedance curve for RY2018 was deeper than RY2021 by 0.20 ft to 0.40 ft for P-values ranging from 20% to 80%. RY2017 QPK could have initially scoured (deepened) pools and runs, and remained deeper in RY2018 but gradually aggraded by RY2021. Both observations may seem of minor significance, but incremental, cyclical change (i.e., responding to single flood events) often is rarely dramatic over the short-term.

A Brief Riffle Crest Thalweg (RCT) Refresher

(Modified from: LADWP 2020 Annual Compliance Report for RY2019, Section 2, p. 9)

The greatest importance of RCT contributing to our understanding of how stream ecosystems work under past, present, and future environments, is not because of its usefulness as a universal depth measure, but because of its rate of change in depth as streamflow changes, i.e., when we treat RCT as a verb rather than noun. RCT-Q rating curves are power function fits to the relationship between RCT depth and streamflow (Q). In the USGS website, Q is the dependent variable (Y-axis) presented as a function of stage height, the independent variable (X-axis). In RCT-Q rating curves, streamflow (Q) also is the dependent variable with RCT depth the independent variable.

Switching independent/dependent variables is often necessary, e.g., when estimating RCT depth from a streamflow measurement. Figure 4 illustrates the computational steps necessary to switch axes. The exponent of the RCT-Q rating curve, where Q is the dependent variable, is called the Power Function Exponent (PFE). 'Coefficient A' is the power function coefficient when RCT is the dependent variable (i.e., in Figure 4 Coefficient 'A' is 0.3050). 'Coefficient B' is the power function coefficient when Q is the dependent variable (i.e., in Figure 4 Coefficient 'B' is 24.3623). In lieu of RCT depths at known streamflows, field estimates for PFE and Coefficient 'A' based on profession judgement … much like using Manning's roughness coefficient 'n' by the USGS to estimate peak flows … historic RCT-Q rating curves were constructed for Lower Rush Creek.

$$RCT = a\ Q^{exp} \text{ Power Function}$$

$$RCT = 0.3050\ Q^{0.3719}$$

$$Q = (1/a)^{(1/exp)}\ RCT^{(1/exp)}$$

$$\boxed{Q = 24.3623\ RCT^{2.6890}}$$

$$PFE = 2.6890$$

Figure 4. Switching axes in RCT-Q rating curves. Blue rectangle is the RCT-Q rating curve (i.e., with Q as the dependent variable).

Riffle crest cross-sections function similarly as man-made weirs (refer to LADWP Compliance Report May2020 Section 4. Stream Monitoring Report, pp. 5-7). Power function rating curves for rectangular weirs have a PFE of 1.50 whereas 'V' shaped weirs have a PFE of 2.50. As Lower Rush Creek recovers a new baseline elevation, the mainstem channel will become more complex geomorphically, and consequently more complex hydraulically. For the family of HUs comprising the 10 Falls-to-Ford mainstem, their collective RCT-Q rating curves have been evolving from riffle crest (RC) cross sections rectangular (i.e., low PFEs) to RC cross sections more irregular and triangular (i.e., higher PFEs). Hydraulic complexity will increase by: (1) coarsening the channelbed, (2) narrowing the active channel by encroaching willows and cottonwoods accelerating sediment deposition on the floodplain, (3) supplying, forming, and retaining large woody debris (LWD), and (4) not interfering with beaver activity. PFE and Coefficient 'A' (related to relative hydraulic roughness) are expected to continue increasing with future recovery.

RY2021 RCT-Q Rating Curve Monitoring

While surveying residual pool/run depths, the following hydraulic features also were surveyed: (1) RCT depth at the ambient streamflow Q = 37 cfs and (2) RCT depth of the RY2021 QPK=240 cfs using flood debris lines. The best tool for assessing channel recovery is an individual RCT-Q rating curve for each HU in the reach. Although only two RCT data points, RCT-Q rating curves were constructed for the 10 Falls-to-Ford mainstem reach in RY2021. The PFE of each HU's rating curve is one good measure of channel complexity. From the survey data, fourteen two-point RCT-Q rating were constructed. The 14 estimated PFEs were ranked highest to lowest, then plotted as an exceedence curve (Figure 5). A P-value (exceedence probability) of 20% had a PFE = 2.1983. This means 20% of PFEs among all hydraulic units within the 10 Falls-to-Ford mainstem have a PFE equal to, <u>or greater than</u>, 2.1983. The exceedence curve is basically linear from P-value = 25% to P-value = 85%.

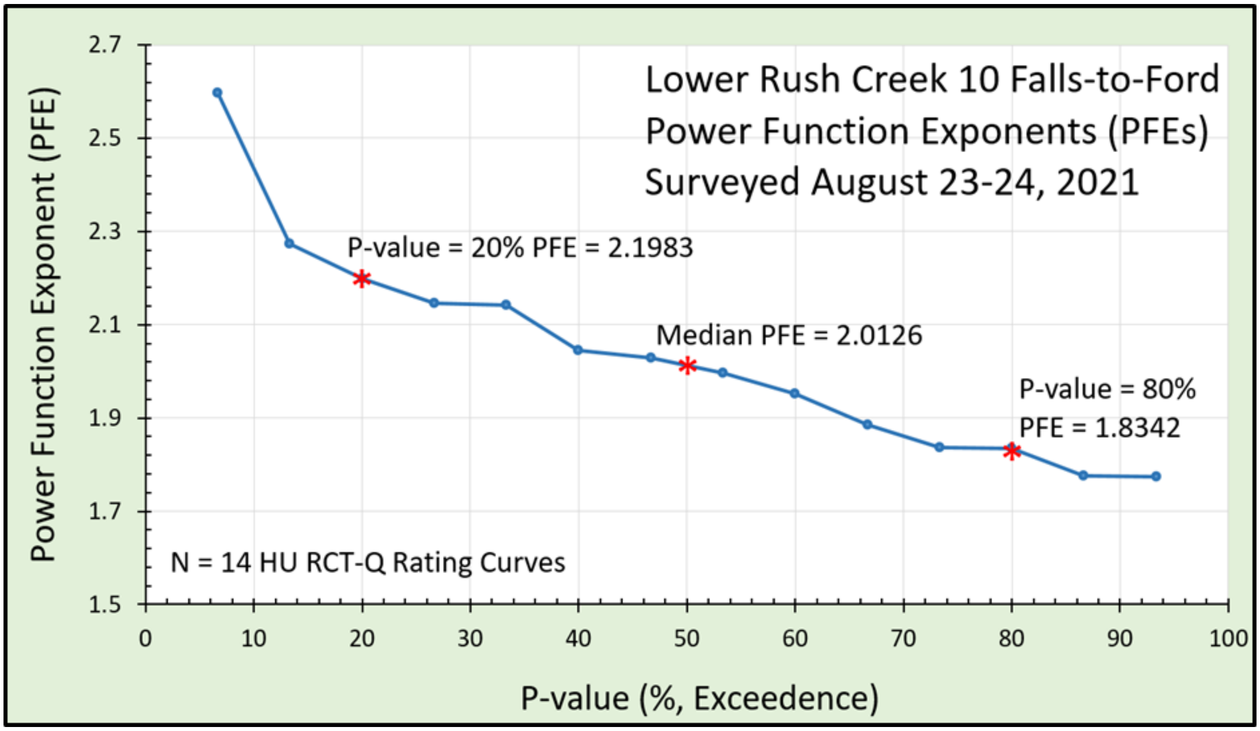


Figure 5. PFE exceedence curve derived from 14 RCT-Q rating curves surveyed August 23 to 24, 2021.

Generating individual RCT-Q rating curves requires considerable effort and training. An alternative approach accomplishes the overall goal of tracking and predicting PFE recovery but with substantially less time and resource investment. Rather than considering each RCT rating curve of each HU individually, consider them all collectively. This was accomplished by first computing RCT depths for the 20%. 50%, and 80% P-values in their exceedence curves for the two streamflows monitored in RY2021 (Figures 6 and 7). Then constructing an RCT rating curve from each exceedence curve.

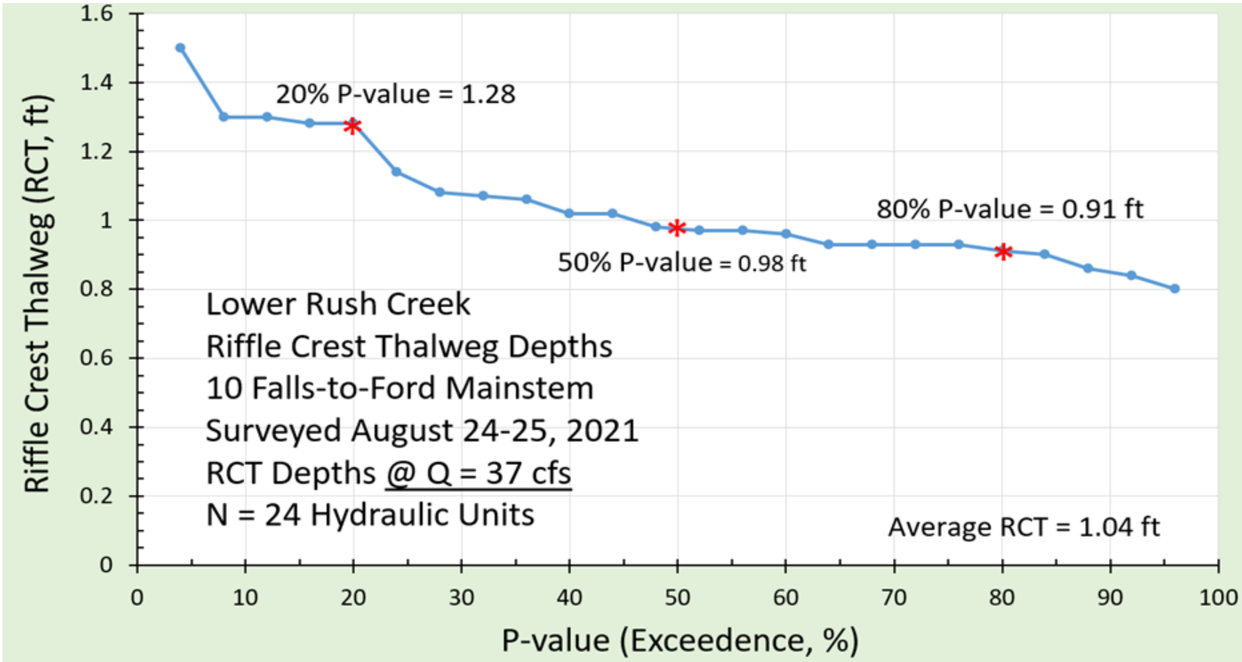


Figure 6. Exceedence curve for RCT at Q = 37 cfs.

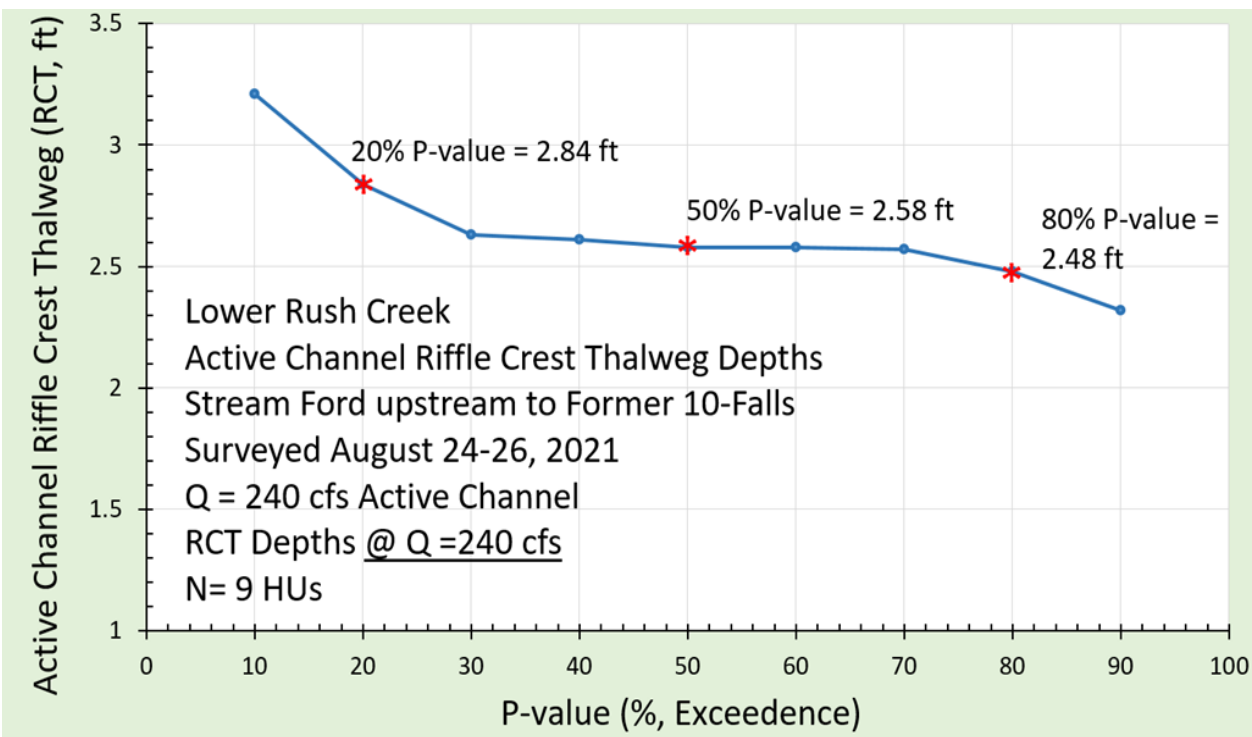


Figure 7. Exceedence curve for RCT at Q = 240 cfs.

For example, the median RCT depth for the ambient streamflow of 37 cfs in the 10 Falls-to-Ford reach was 0.98 ft; for the active channel streamflow of 240 cfs, the median RCT depth was 2.58 ft. A power function fit to just the 2-point median RCT-Q rating curve (Figure 8) was:

$$mRCT = 0.1501\ Q^{0.5192}$$
$$Q = 38.6029\ mRCT^{1.9264}.$$

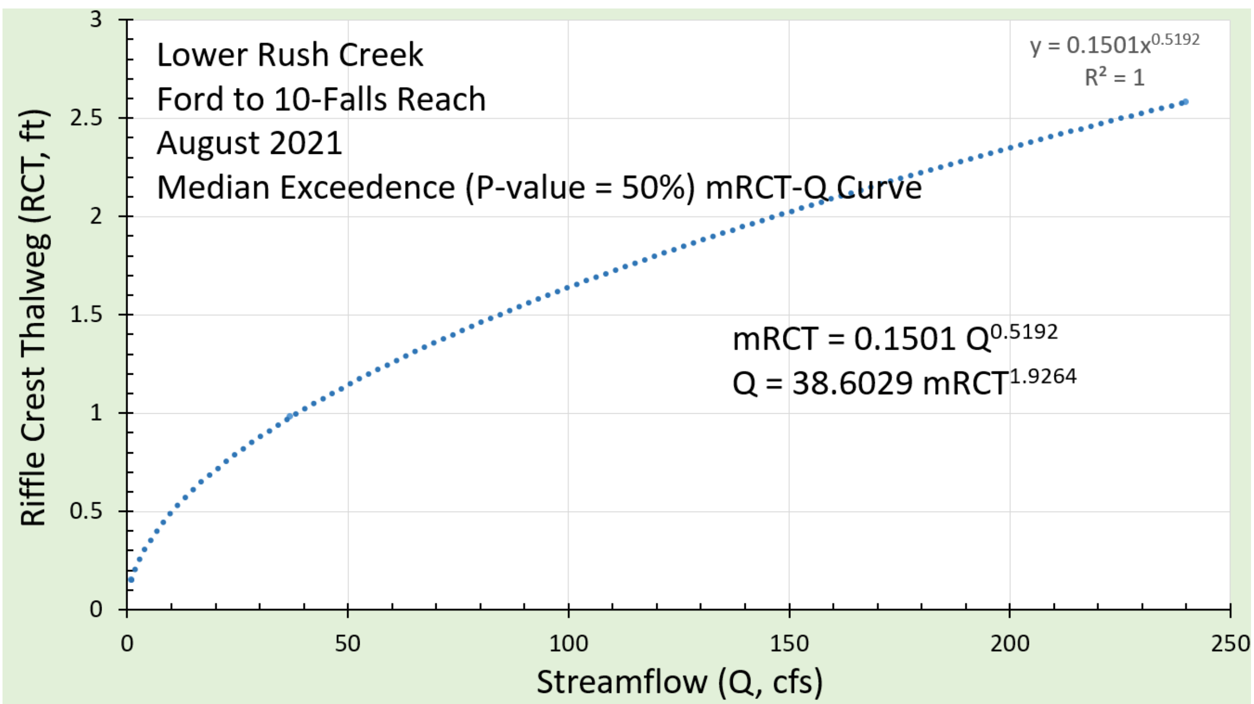


Figure 8. RY2021 Median RCT-Q rating curve for Lower Rush Creek 10 Falls-to-Ford.

The final step was plotting all three RY2021 exceedence RCT-Q rating curves, one for each selected P-value (Figure 9). Note the relatively large gap between the P =

50% exceedence curve and the 20%exceedence curve, as opposed to the gap between 80% P-value and 50% P-value exceedence curves. The explanation can be traced back to Figure 3, the PFE exceedence curve. The sharp upswing in PFEs above ~25% is the cause. HUs with PFEs much greater than 2.1000 are relatively unique, typically associated with recent LWD jam formations and/or recent abrupt lateral channel shifting (e.g., after the RY2017 QPK). During recovery, the three exceedence curves will shift upward absolutely (i.e., in RCT depths) and relatively (i.e., the exceedence curves will steepen). The median RCT-Q exceedence curve should increase significantly compared to the 20% and 80% exceedence curves, signaling a more hydraulically complex mainstem channel.

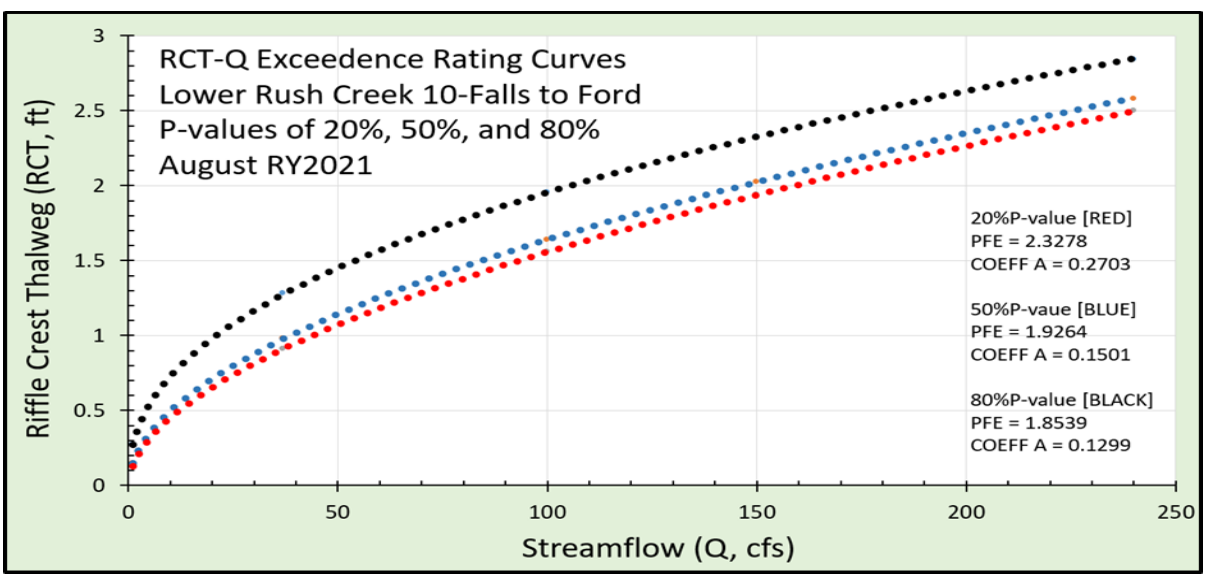


Figure 9. Plotted P-value 20%, 50%, and 80% exceedence RCT-Q rating curves for RY2021.

RCT-Q rating curves are tools for evaluating ecological streamflow thresholds recommended in the Synthesis Report's desired ecological outcomes (Synthesis Report Table 3-1, 27Jan2010). Ecological outcomes for 'stream productivity and brown trout habitat' and recommended lower and upper streamflow thresholds were summarized in Figure 10. The specific desired ecological outcome of creating 'off-channel spring/early-summer stream connectivity' requires a range in streamflow thresholds between 90 cfs and 160 cfs. At 90 cfs in the 10 Falls-to-Ford mainstem, RCT depth equaled 1.86 ft @ P-value = 20%, 1.55 ft @ P-value = 50%, and 1.47 ft @ P-value = 80%. The approximate 0.40 ft difference in RCT depth between 20% and 80% was 0.40 ft for the same streamflow. For a stream ecosystem where 0.20 ft changes in stage are important, a 0.40 ft difference would significantly affect floodplain connectivity. Therefore, increases in RCT depth (with increasing PFE) at a threshold streamflow will improve hydraulic floodplain connectivity, with greater magnitude and duration of side-channel surface flows.

Synthesis Report: Ecological Outcomes Lower Rush Creek Stream Productivity and Brown Trout Habitat Recommended Productivity and Brown Trout Streamflow Thresholds (Q_T)			
No. #	Desired Ecological Outcomes	Q_T RANGE: LOW	Q_T RANGE: HIGH
1	Abundant, High Quality Brown Trout Winter Holding	16 cfs	35 cfs
2	High Quality Mainstem and Channel Margin Brown Trout Fry Rearing	40 cfs	60 cfs
3	Abundant, High Quality Brown Trout Foraging and Holding Habitat	25 cfs	40 cfs
4	Productive Benthic Macro-Invertebrate (BMI) Riffles	40 cfs	110 cfs
5	Off-Channel Spring/Early-Summer Streamflow Connectivity	90 cfs	160 cfs

Figure 10. Ecological outcomes in the Synthesis Report with low and high streamflow thresholds for stream productivity and brown trout habitat.

An Example of Mainstem Channel Recovery Monitoring

The Gary Smith Overlook meander bend in Lower Rush Creek, upstream of the Ford, was extremely wide and shallow in 1987 (Figure 11). Its RC cross section was rectangular and shallow. Based on field experience, after examining 100s of RCT-Q rating curves elsewhere, its PFE would likely have been between 1.65 and 1.75 with a Coefficient ‘A’ between 0.06 and 0.10. Using both professional estimates, a reasonable RCT-Q rating curve in 1987 was (underlined values are estimated Coefficient ‘A’ and PFE):

$$\text{EST: RCT} = \underline{0.0800}^{\ Q0.5714}$$
$$\text{EST: Q} = 83.0984\ \text{RCT}^{\underline{1.7500}}.$$

In comparison, the surveyed RCT-Q rating curve measured from field data for the Gary Smith Overlook on 23August2021 (Figure 12) was:

$$\text{RCT} = 0.1488\ \text{Q}^{0.4927}$$
$$\text{Q} = 47.7843\ \text{RCT}^{2.0296}.$$

As willows and cottonwoods further encroach this RB point bar, accelerated sediment deposition will begin to significantly confine and build the active channel. PFE and Coefficient ‘A’ should continue increasing as active channel confinement increases.

Figure 11. Gary Smith Overlook, looking downstream, in 1987 with estimated RCT-Q rating curve.

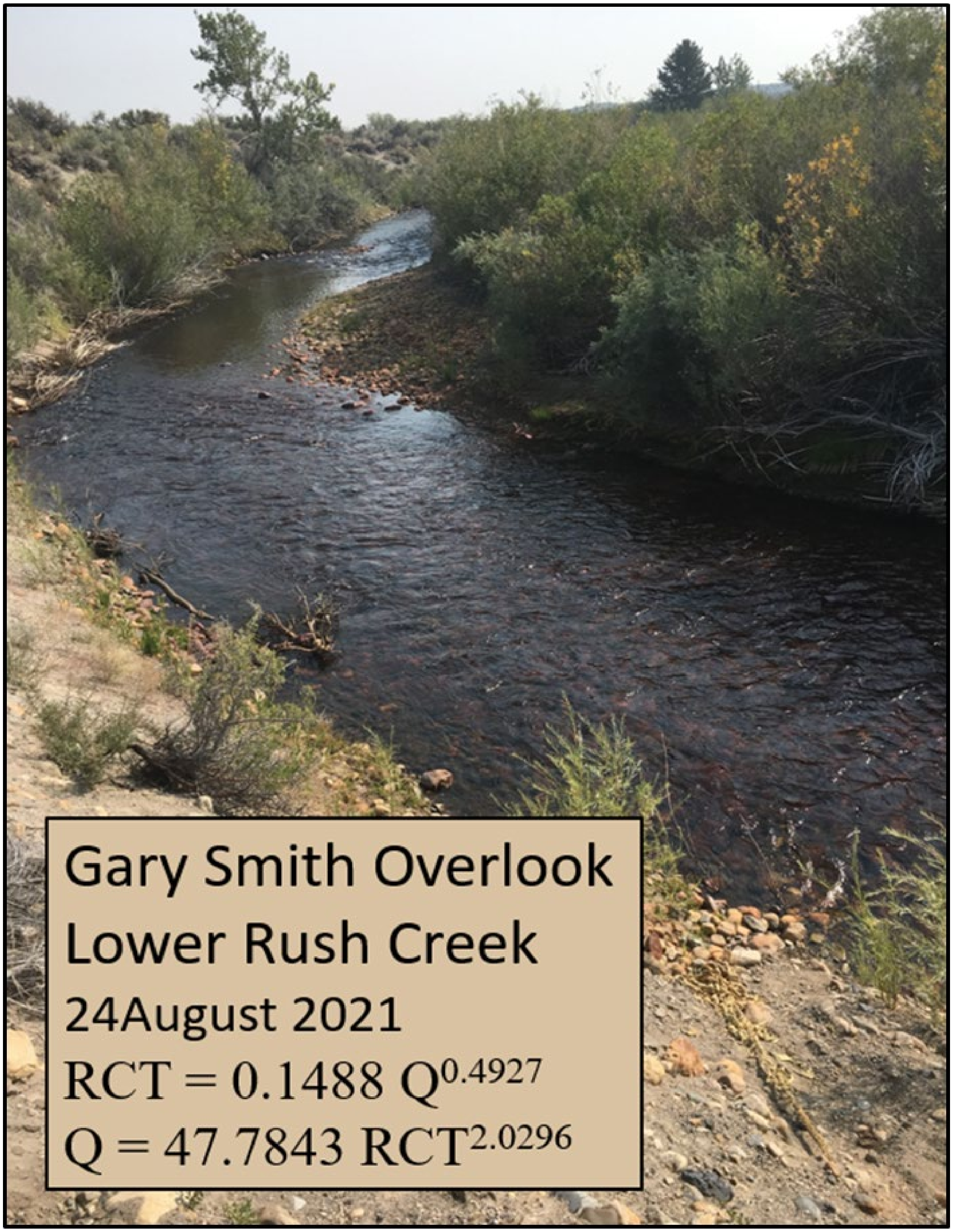


Figure 12. Gary Smith Overlook, looking downstream, on August 25, 2021 with RCT-Q rating curve.

DEPARTMENT OF THE INTERIOR
FRANKLIN K. LANE, Secretary

UNITED STATES GEOLOGICAL SURVEY
GEORGE OTIS SMITH, Director

Water-Supply Paper 390

SURFACE WATER SUPPLY OF THE UNITED STATES
1914

PART X. THE GREAT BASIN

TASK No.2
Pre-1941 Channel Cross-Section and RCT-Q Rating Curves

Restoring Lower Rush Creek to its pre-1941 condition is a daunting task especially given the difficulty in quantitatively describing/understanding how the channel once looked and hydraulically performed. Two possible ‘quantitative looks’ are explored in Task 2. First, investigating early USGS gaging records for gage rating curve data that could be converted into an RCT-Q rating curve. Two sources were located: one for Rush Creek and one for Lee Vining Creek. Second, while surveying channel headcutting near the 4-Floodplain’s 4bii Side Channel Entrance, a short hike farther upstream in

RY2018, above the headcutting, encountered a hydraulic unit that appeared to have its pre-1941 morphology intact. In RY2021, Robbie DiPaolo and I returned to survey a complete channel cross-section with the added task to reconstruct its RCT-Q rating curve.

Pre-1941 Rush Creek RCT-Q Rating Curve

The USGS operated Gaging Station No.10287500 'Rush Creek nr Mono Lake' from November 16, 1910 to September 30, 1914 (Figure 13). Gage location was 'one fourth mile above mouth of creek and three miles below mouth of Walker Creek.' USGS Water Supply Paper No.390 for WY1914 Great Basin provided sufficient stream gage stages and streamflows to construct the station's RCT-Q rating curve. To convert stream gage rating curve data into a RCT-Q rating curve, stage height must be transformed into RCT depths (Figure 14). This required an estimate of stage height at the 'point-of-zero surface flow' (i.e., the stage height at which no surface flow passes the RCT at the downstream tail of the gaged pool).

RUSH CREEK NEAR MONO LAKE, CAL.

Location.—In the NE. ¼ sec. 13, T. 1 N., R. 26 E., at highway bridge, one-fourth mile above mouth of creek, 3 miles below mouth of Walker Creek, and about 8 miles southeast of Mono Lake post office.

Drainage area.—Not measured.

Records available.—November 16, 1910, to September 30, 1914.

Gage.—Original vertical staff, fastened to a cottonwood tree on right bank just above bridge, was washed out June 24, 1911; a new gage was installed July 6, 1911, at an independent datum, as the bench mark was also destroyed by the flood; September 15, 1911, the gage was moved and the datum was raised 0.90 foot.

Discharge measurements.—Made from highway bridge or by wading.

Channel and control.—One channel at all stages at the bridge; stream bed is composed of sand and fine gravel and shifts during high water.

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Extremes of discharge.—Maximum stage recorded during year, 5.8 feet June 5 and 28 (discharge, 615 second-feet); minimum stage recorded, 2.36 feet October 11 (discharge, 14 second-feet).

1910–1914: Maximum stage recorded, 8.4 feet June 18, 1911 (discharge, 1,280 second-feet); minimum stage recorded, 2.30 feet September 27–31 and October 27 to November 7, 1912 (discharge, 11 second-feet).

Winter flow.—Discharge relation somewhat affected by ice.

Diversions.—Water is diverted above the station for irrigation.

Regulation.—Flow affected somewhat by diversions.

Accuracy.—Rating curve fairly well defined. As gage-height record is incomplete, estimates of monthly discharge have not been prepared.

Cooperation.—Gage heights, except from March 16 to May 12, furnished by the United States Forest Service. Gage-height record March 16 to May 12, 1914, furnished by R. G. McDonald.

Discharge measurements of Rush Creek near Mono Lake, Cal., during the year ending Sept. 30, 1914.

Date.	Made by—	Gage height.	Discharge.	Date.	Made by—	Gage height.	Discharge.
		Feet.	*Sec.-ft.*			*Feet.*	*Sec.-ft.*
Oct. 11	Mathias and Clark	2.36	14	Mar. 9	F. B. Clark	2.94	62
Feb. 28	F. B. Clark	2.44	17				

Figure 13. USGS Gaging Station No. 10287500 in USGS Water Supply Paper No.390 for WY1914 Great Basin.

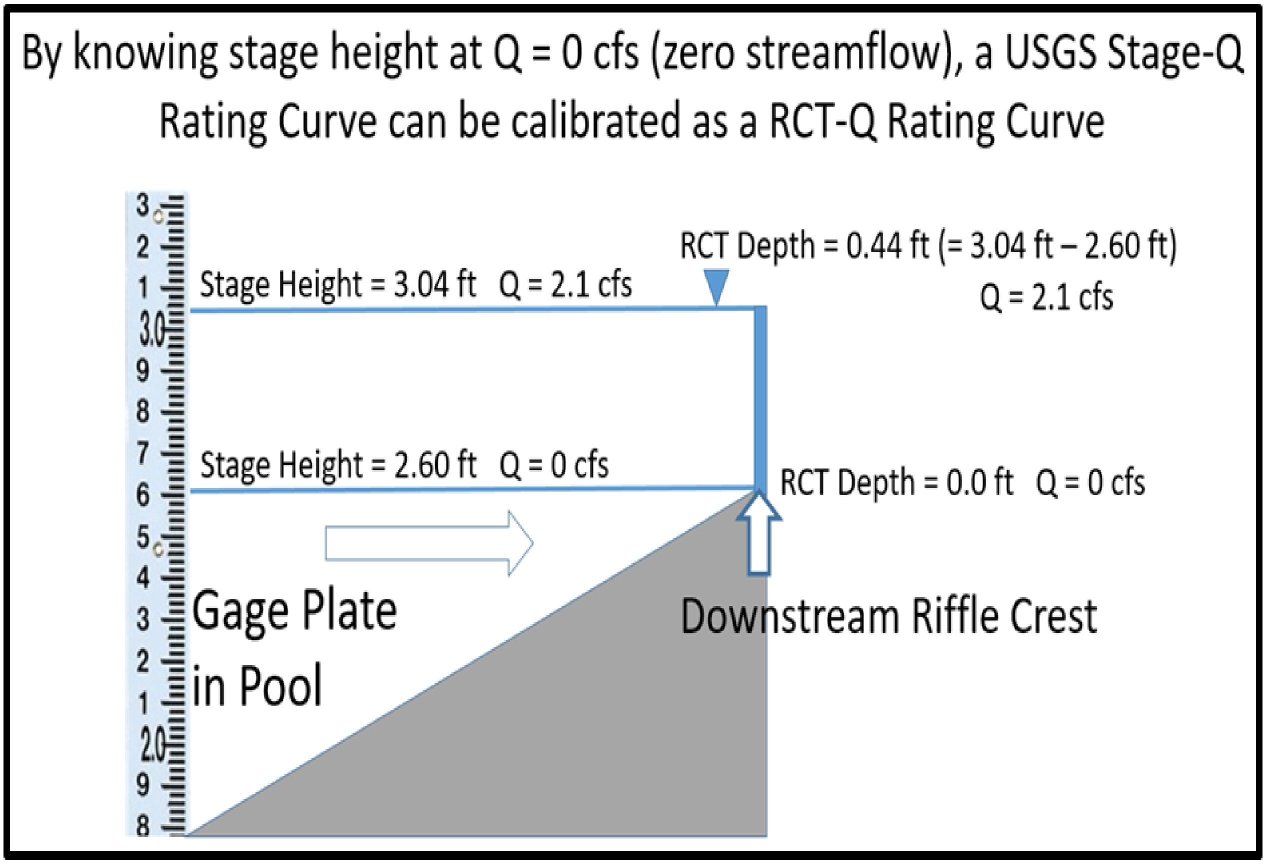


Figure 14. Working example for converting gage height into RCT depth.

The computed RCT-Q rating curve (Figure 15) had a PFE of 1.9084:

$$RCT = 0.1379\ Q^{0.5240}$$

$$Q = 43.8587\ RCT^{1.9084}$$

Referring back to Figure 5, the PFE exceedence curve for mainstem 10-Falls-to-Ford in RY20221 placed a PFE of 1.9084 at a P-value of 66%. This means approximately 34% of the HUs in RY2921 within the 10 Falls-to-Ford mainstem reach have PFE's less than, or equal to, this pre-1941 PFE of 1.9084. As mainstem Rush Creek recovers, the PFEs of most HUs will increase.

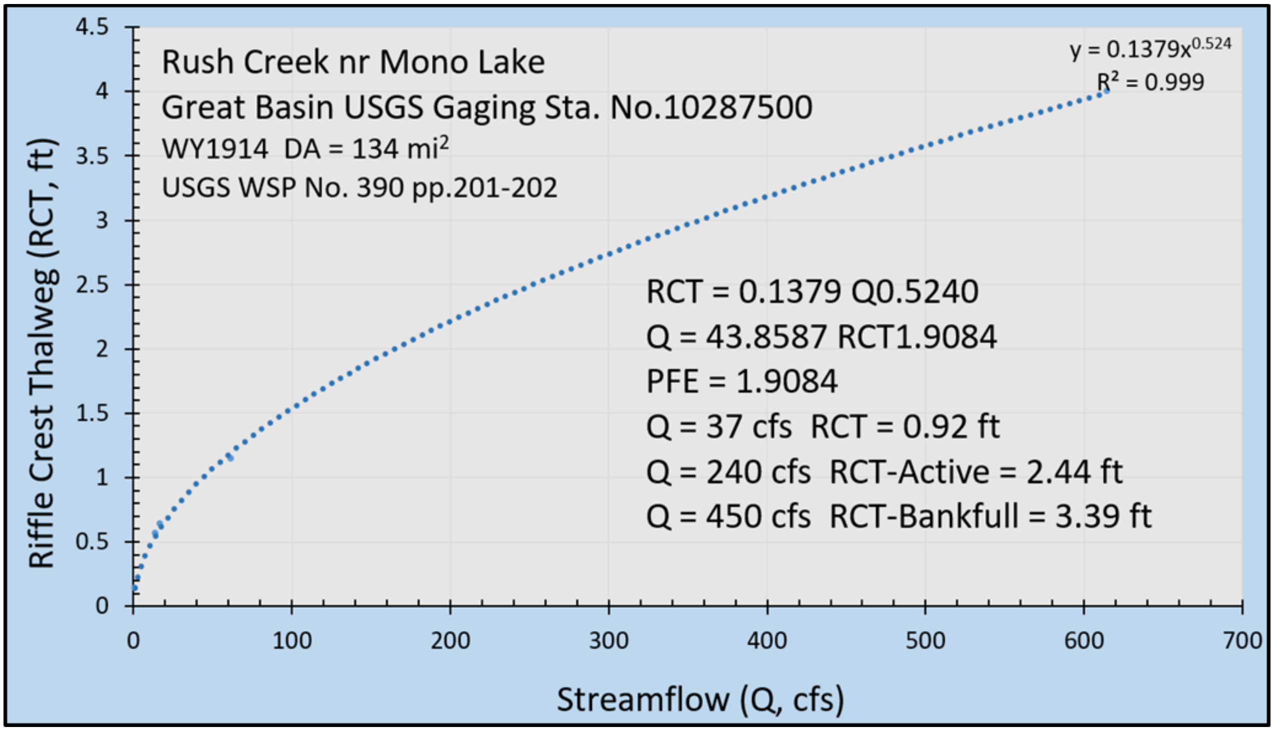


Figure 15. RCT-Q rating curve for Lower Rush Creek in WY1914.

Pre-1941 Lower Rush Creek Channel Cross-Section and RCT-Q Rating Curve

During fieldwork in RY2018, a hydraulic unit (HU) approximately 600 ft upstream of the 4bii Side Channel entrance to the 4-Floodplain was found that appeared to have maintained its pre-1941 channel morphology. A large, old growth cottonwood on the left bank (LB) was rooted at the historic bankfull channel elevation (Figures 16 to 18), with the trunk buried 0.8 ft deep in fine silty flood deposits. This cottonwood was just downstream of a dense, old-growth red willow stand.

A primary objective for RY2021 was to survey a cross section through this HU, assign streamflows to multiple stage heights, then construct its RCT-Q rating curve. This RCT-Q rating curve representing the pre-1941 channel morphology could then serve as a baseline condition from which to gage/track stream channel recovery in Lower Rush Creek.

Figure 16. Grand Old Cottonwood hydraulic unit (HU) old-growth cottonwood on left bank back from the channel bank’s edge. August 25, 2021.

Figure 17. Grand Old Cottonwood hydraulic unit (HU), standing downstream of its RCT and looking upstream on August 25, 2021 (Q = 37 cfs). The cross–section survey tape is visible.

Figure 18. Grand Old Cottonwood hydraulic unit (HU), standing on its riffle crest thalweg (RCT), looking downstream on August 25, 2021 (Q = 37 cfs).

The cross-section, labeled 'Grand Old Cottonwood Cross-Section,' was surveyed August 25, 2021 by Robbie DiPaolo on the engineers level and Bill Trush as rodman (Figure 19). Survey notes are provided in Figure 20. Its unorthodox presentation, sideways instead of vertical, kept the notes to a single page per cross-section yet allowed larger font size for clarity. The cross-section was not placed directly on the riffle crest (RC) but rather in HU's downstream pool tail. The last station entry locates the RCT bed location (at the bottom of the pool tail) immediately downstream of the actual cross section. Relative to the LB (left bank looking downstream) rebar benchmark elevations of 100.00 ft, the channelbed elevation at the RCT = 95.19 ft. This will be useful for future routine annual surveys documenting potential channel downcutting.

Although an engineers level would be best, a reliable measure of RCT channelbed elevation can be made with a stadia rod, piece of string, and simple bubble level provided a benchmark is nearby. First, measure the height of the adjacent present water surface relative to the LB top rebar that has been assigned an elevation of 100.00 ft. If the water surface is 0.75 ft below the top of the LB rebar (i.e., by stretching a leveled string from the rebar top over the water surface), the water surface elevation will equal 100.00 ft – 0.75 ft = 99.25 ft. Next, measure

RCT depth on the RC cross-section. Subtract RCT depth from 99.25 ft to estimate RCT channelbed elevation. If RCT depth is 1.29 ft, then the RCT channelbed elevation equals 99.25 ft – 1.29 ft = 97.96 ft (i.e., the stage at zero streamflow). The greatest source of error is locating the RCT, provided the location of the cross section is not far above the RCT. Grand Old Cottonwood Cross-Section also provided RTC depths to construct a 2-point RCT-Q rating curve at the riffle crest cross-section for this HU (Figure 21).

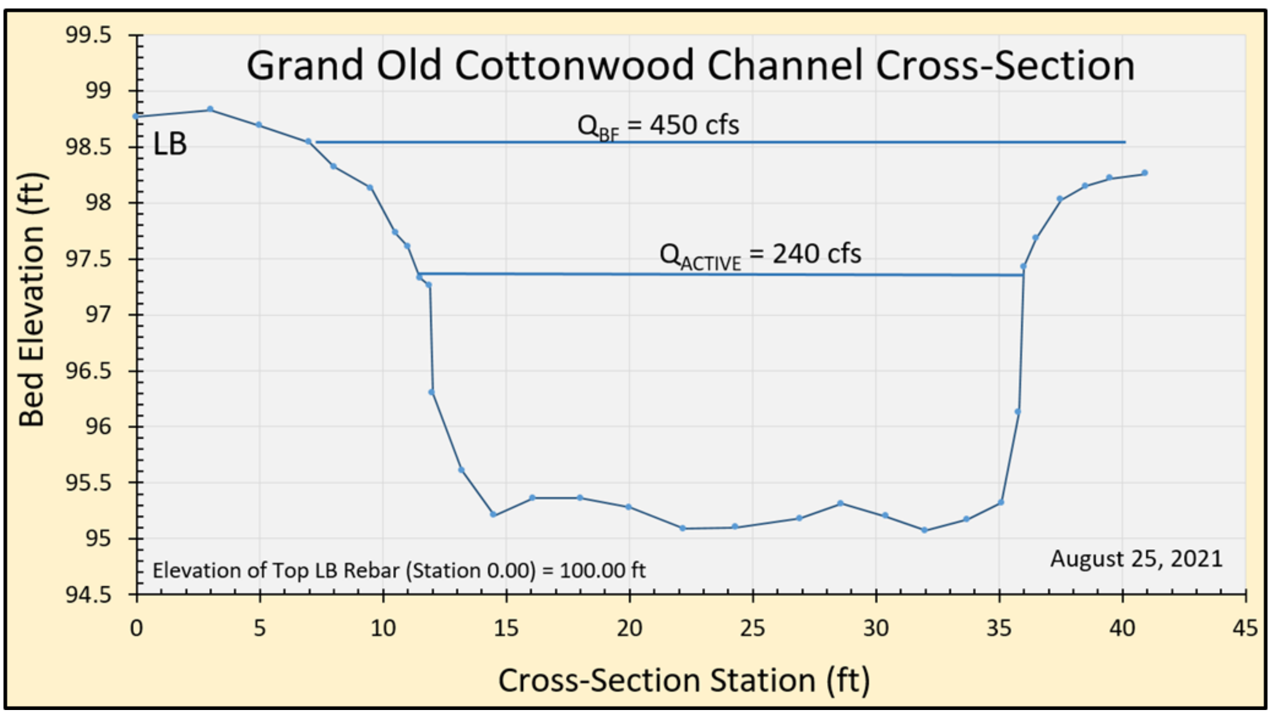
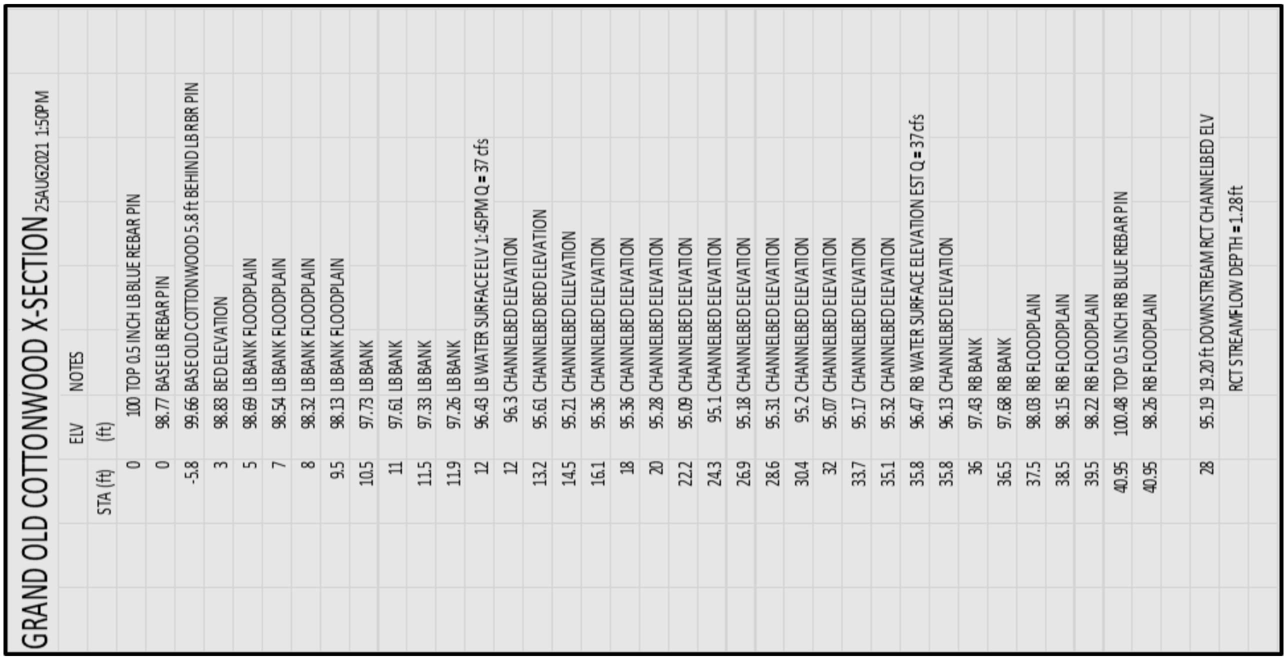


Figure 19. Grand Old Cottonwood Cross-Section surveyed 25Aug2021.



GRAND OLD COTTONWOOD X-SECTION 25AUG2021 1:50PM

STA (ft)	ELV (ft)	NOTES
0	100	TOP 0.5 INCH LB BLUE REBAR PIN
0	98.77	BASE LB REBAR PIN
-5.8	99.66	BASE OLD COTTONWOOD 5.8 ft BEHIND LB RBR PIN
3	98.83	BED ELEVATION
5	98.69	LB BANK FLOODPLAIN
7	98.54	LB BANK FLOODPLAIN
8	98.32	LB BANK FLOODPLAIN
9.5	98.13	LB BANK FLOODPLAIN
10.5	97.73	LB BANK
11	97.61	LB BANK
11.5	97.33	LB BANK
11.9	97.26	LB BANK
12	96.43	LB WATER SURFACE ELV 1:45PM Q = 37 cfs
12	96.3	CHANNELBED ELEVATION
13.2	95.61	CHANNELBED BED ELEVATION
14.5	95.21	CHANNELBED ELLEVATION
16.1	95.36	CHANNELBED ELEVATION
18	95.36	CHANNELBED ELEVATION
20	95.28	CHANNELBED ELEVATION
22.2	95.09	CHANNELBED ELEVATION
24.3	95.1	CHANNELBED ELEVATION
26.9	95.18	CHANNELBED ELEVATION
28.6	95.31	CHANNELBED ELEVATION
30.4	95.2	CHANNELBED ELEVATION
32	95.07	CHANNELBED ELEVATION
33.7	95.17	CHANNELBED ELEVATION
35.1	95.32	CHANNELBED ELEVATION
35.8	96.47	RB WATER SURFACE ELEVATION EST Q = 37 cfs
35.8	96.13	CHANNELBED ELEVATION
36	97.43	RB BANK
36.5	97.68	RB BANK
37.5	98.03	RB FLOODPLAIN
38.5	98.15	RB FLOODPLAIN
39.5	98.22	RB FLOODPLAIN
40.95	100.48	TOP 0.5 INCH RB BLUE REBAR PIN
40.95	98.26	RB FLOODPLAIN
28	95.19	19.20 ft DOWNSTREAM RCT CHANNELBED ELV
		RCT STREAMFLOW DEPTH = 1.28 ft

Figure 20. Grand Old Cottonwood Cross Section notes.

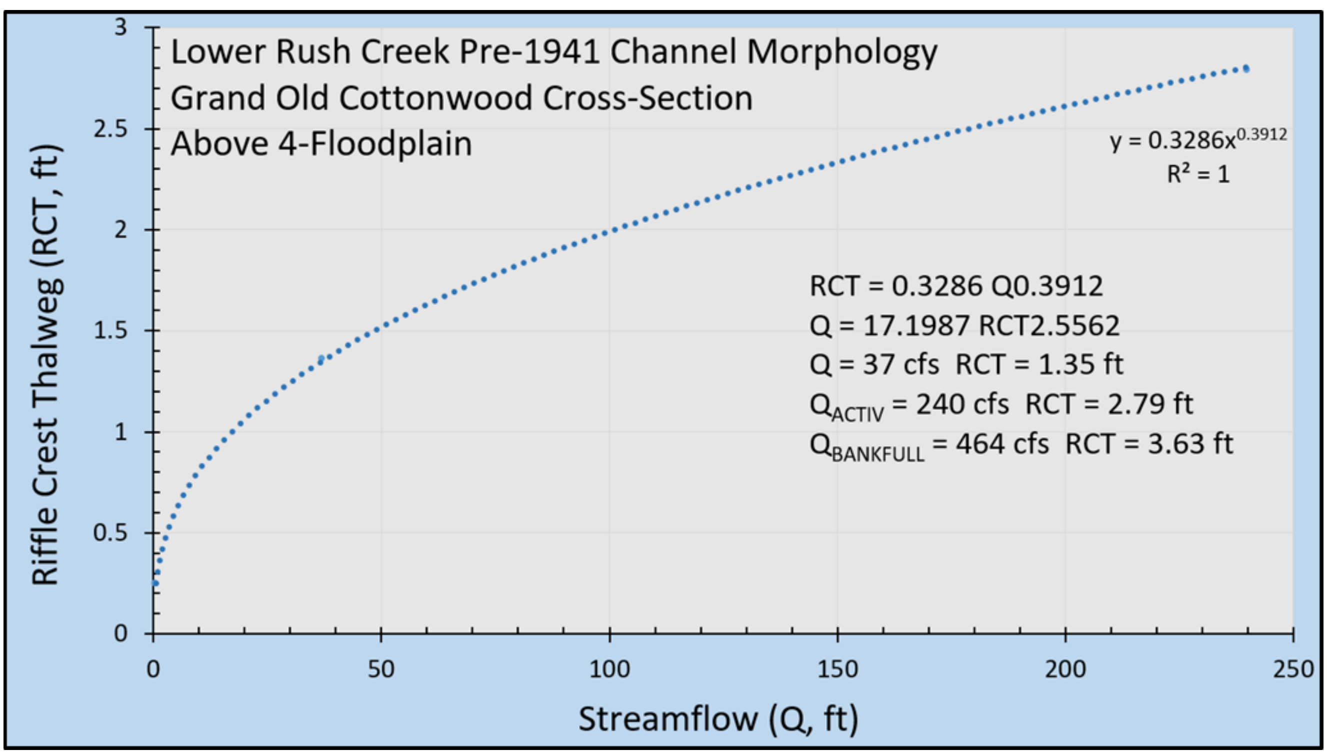


Figure 21. Grand Old Cottonwood RCT-Q rating curve.

Both pre-1941 RCT-Q rating curves plotted together (Figure 22) defined a ‘playing-field’ for Lower Rush Creek mainstem recovery. Every mainstem channel reach is a family of HUs, and therefore family of PFEs. Recovery will entail not just an increase in PFE and Coefficient ‘A’ magnitude, but a wider diversity in magnitude among all HUs.

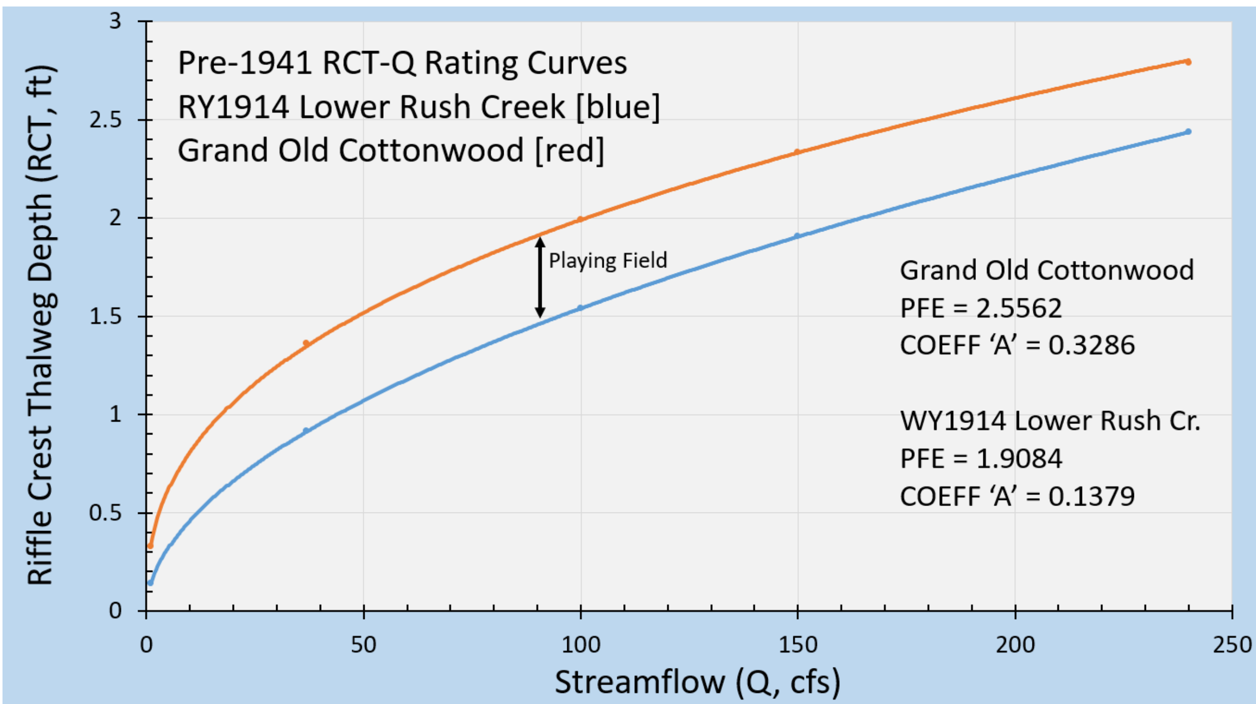


Figure 22. Two Pre-1941 Lower Rush Creek RCT-Q rating curves.

Task No. 3.
Monitoring Floodplain Riparian Vigor

One field day was dedicated to monitoring riparian vigor measuring annual branch increment (ABI). With 80 to 85 percent of Rush Creek mainstem streamflow flowing down the 8-Channel in late-August 2021, water availability in the 8-Floodplain was extremely high if not excessive. What was dry chaparral in the 8-Floodplain before the RY2017 QPK, is now becoming an expansive wetland sprouting cattails (Figure 23). The prominent solitary Jeffrey pine in the upper 8-Floodplain died as continuous surface flow from the 8-Side Channel brought the shallow groundwater table to the surface.

Figure 23. Dead Jeffrey pine and cattails in Lower Rush Creek's Upper 8-Floodplain.

Refer to LADWP Compliance Report May2019 [RY2018 monitoring] Section 4. Stream Monitoring Report (pp. 27-50) for background information addressing the riparian vigor monitoring plan and measurement techniques. Two yellow willows and two cottonwoods in the 8-Floodplain (Figure 24) were measured for riparian vigor in RY2021. The dead Jeffrey pine is 200 ft to the left of Cottonwood R4_03 in the upper right corner in Figure 24.

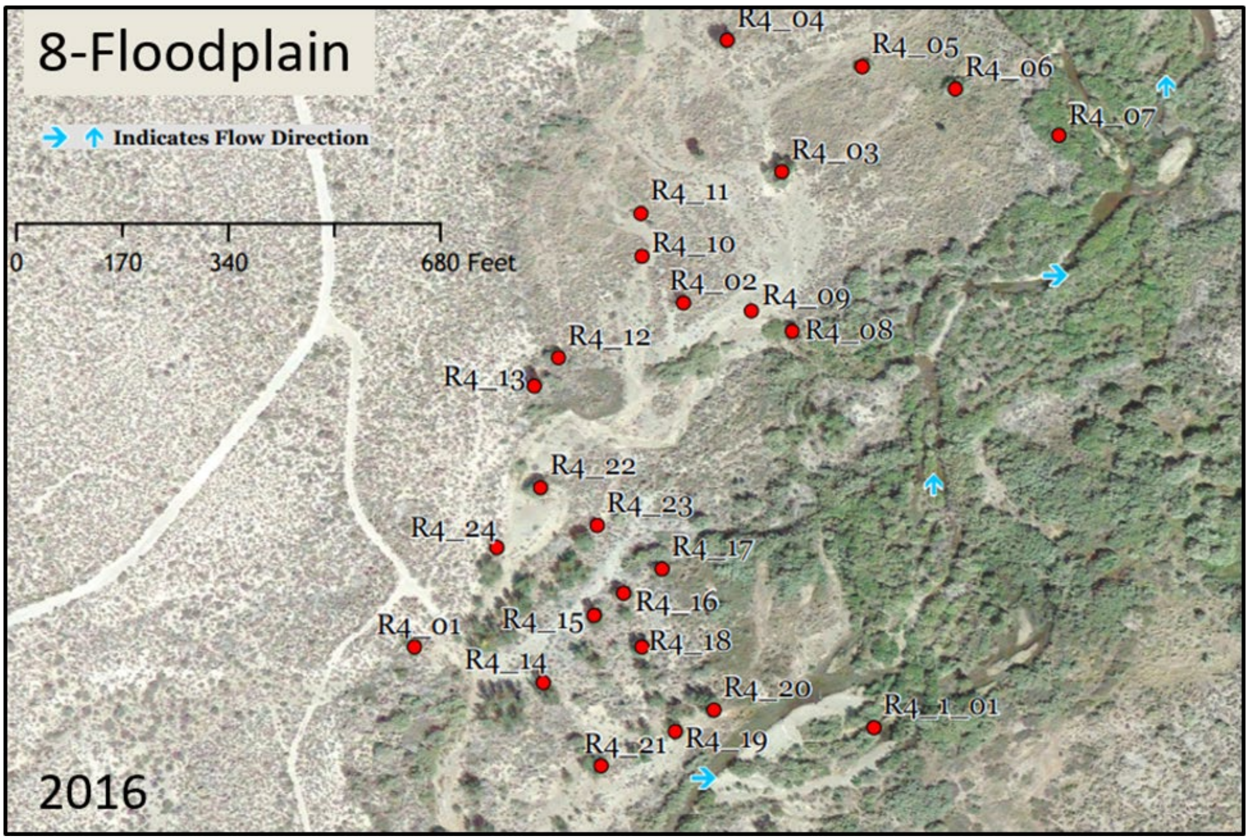


Figure 24. Black cottonwood and yellow willow ABI sample tree locations in Upper 8-Floodplain.

Prior to RY2017, limited streamflow entering the 8-Channel rapidly returned to the mainstem. In RY2018, streamflow was still flowing down the upper 8-Floodplain through early-autumn but was observed rapidly drying-up farther downstream. By RY2021 a long, re-furbished beaver dam spans the mainstem channel, backwatering the 8-Channel outlet and re-directing most 8-Channel surface flow back-onto the 8-Floodplain (Figures 25A&B) rather than returning to the mainstem channel.

Robbie DiPaolo (Mono Lake Committee) and I tracked this re-directed floodplain surface flow farther downstream. Several small reconnections (less than 1 cfs each) back into the mainstem were observed. But most of this 8-Floodplain surface flow re-entered the mainstem just downstream of the ‘million-dollar bend’ only a few hundred feet upstream of the former 10-Falls. If the 8-Channel Entrance had not greatly increased its streamflow capacity, the channel-spanning beaver dam likely would have backwatered mainstem flows onto the 8-Flooodplain regardless.

Figure 25A. Beaver dam spanning the mainstem Rush Creek backwaters streamflow toward the 8-Floodplain. August 25, 2021. Beaver dam barely visible in the background.

Figure 25B. Standing with back to the beaver dam backwater in Figure 25A and looking out onto a re-watering 8-Floodplain. August 25, 2021.

RY2021 Yellow Willows ABI Vigor in Upper 8-Floodplain

On August 27, 2021, RY2021 annual branch increments (ABI) were measured for a pair of yellow willows, R4_05 and R4_06 (Figures 26 and 27), in the upper 8-Floodplain of Lower Rush Creek. Both willows were measured for ABIs in RY2016, RY2017, and RY2018. This willow pair provided the opportunity to compare patterns in vigor response (i.e., ABI exceedence probability curves) under distinctly different RYs. In Figure 25B, the two prominent green ‘clumps’ in the distant 8-Floodplain (and slightly to the left of center) are yellow willows R4-05 and R4-06. When sampling a tree, the objective was to systematically measure branches surrounding the entire tree, conceding the capability of measuring branches out-of-reach. A small sub-set of stems with relatively long ABI’s were encountered in all RYs and in most trees sampled. These distinctly high annual ABIs, almost always growing vertically, create an upswing in most ABI probability curves at P-values < 20%. Especially for more mature black cottonwoods. Branches with much higher ABIs were found growing: (1) from the tree crown, (2) at the apex of lateral branches, (3) as runners under the canopy, and (4) at canopy openings were a large branch had broken-off exposing a patch of available sunlight. ABIs with exceedence ranging between P-values of 20% and 80% provided a more reasonable basis for comparing vigor between RYs.

Figure 26. Yellow Willow R4_05 in Lower Rush Creek 8-Floodplain. August 27, 2021.

Figure 27. Yellow Willow R4_06 in Lower Rush Creek 8-Floodplain. August 27, 2021.

ABIs with exceedence ranging between P-values of 20% and 80% (Figures 28 and 29) provided baselines for comparing ABIs between RYs and both trees. Simplifying even more, median ABIs (i.e., P-value = 50%) at R4_05 for the four RYs were 39.5, 61.5, 54.5, and 90.0 mm respectively (WY2016-21); median R4_06 ABIs for the four WYs were 8.0 mm, 219 mm, 97 mm, and 139 mm respectively. Under *saturated* shallow groundwater WY2021 conditions (observed standing water around both trees), one median ABI was 90 mm while the other 139 mm. Their median ABIs for WY2016 under *dry* shallow groundwater conditions were 8.0 mm and 39.5 mm. Widely different median ABIs between trees. But the pair exhibited similar patterns of vigor in response to each runoff year.

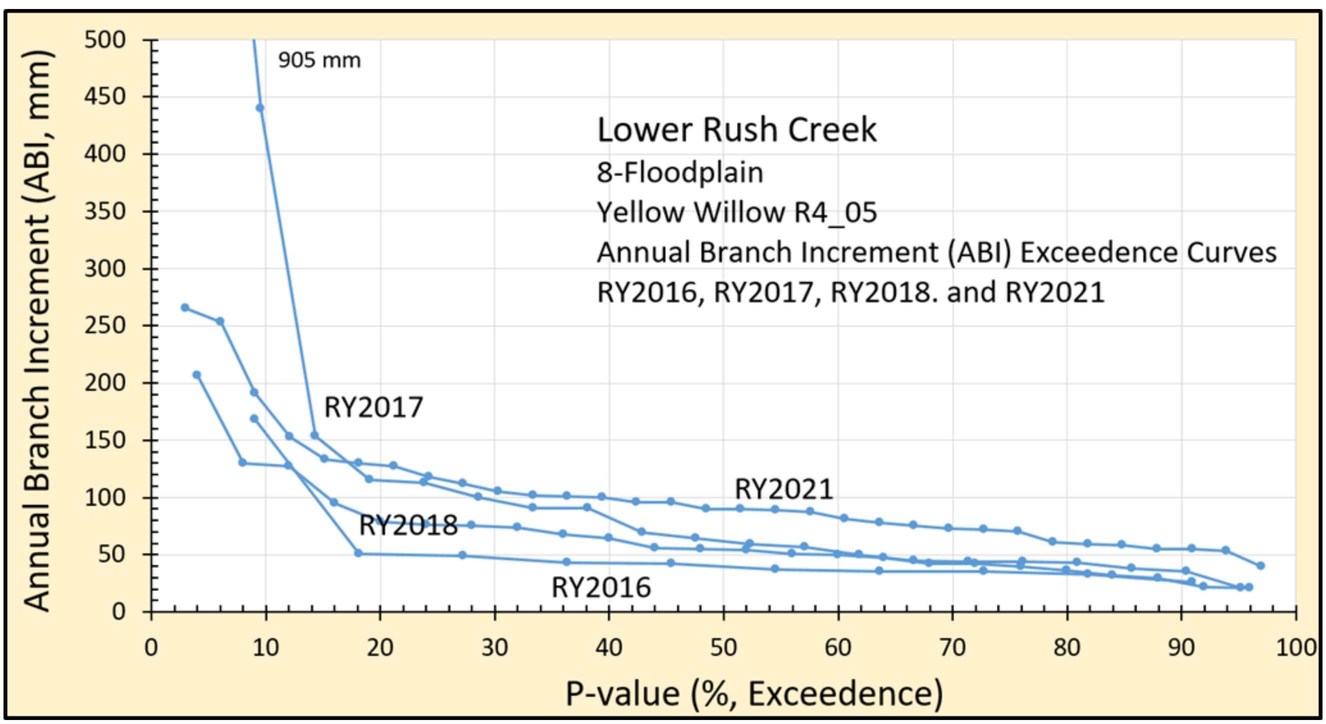


Figure 28. ABI Yellow Willow R4_05 exceedance curves for RY2016, 2017, 2018, and 2021 in Upper 8-Floodplain.

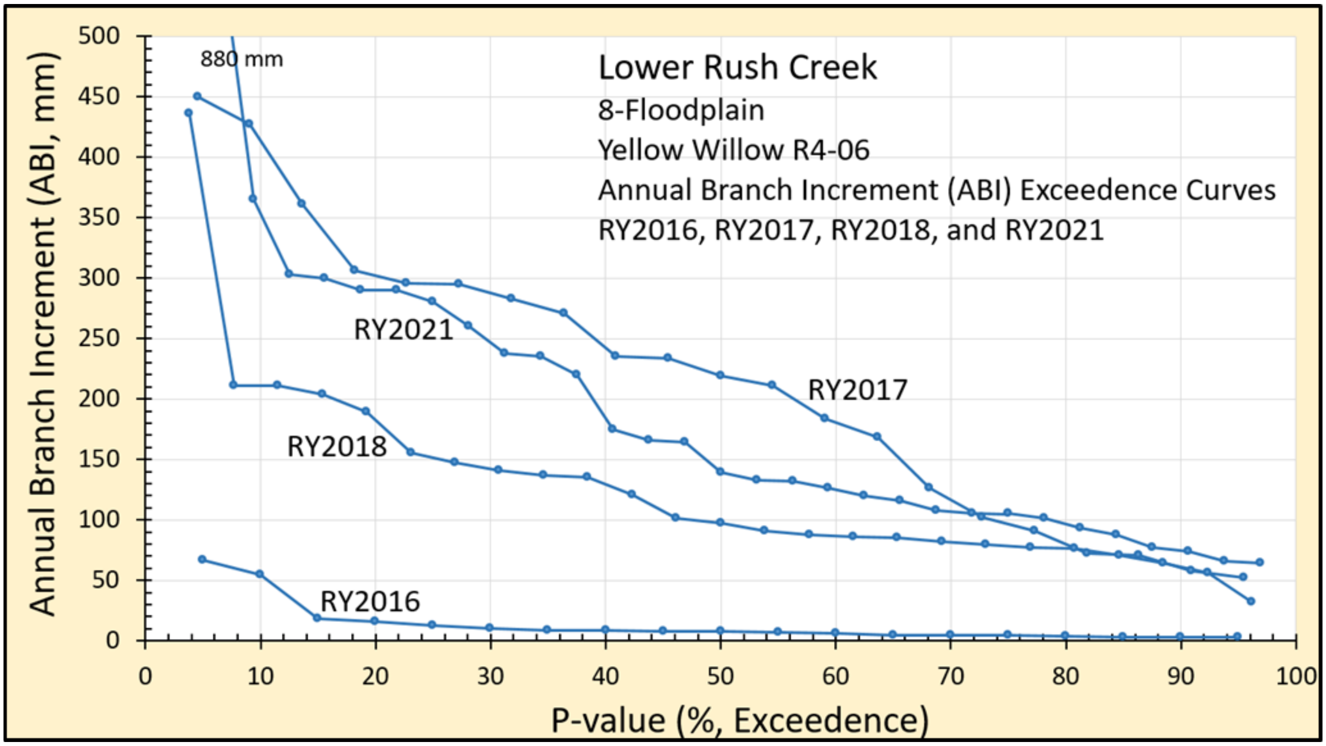


Figure 29. ABI exceedence curves for Yellow Willow R4_06 in RY2016, 2017, 2018, and 2021.

RY2021 Black Cottonwoods ABI Vigor in Upper 8-Floodplain

Two cottonwoods, R4_03 and R4_04, were also measured within the same upper 8-Floodplain location (Figure 24) and over the same RYs as the two yellow willow trees. R4_03 (Figure 30) had a dead old growth trunk 2.20 ft in diameter surrounded by six stems with diameters of 0.56, 0.62, 0.66, 0.82. 0.98, and 1.02 ft. R4-04 (Figure 31) had two trunks with diameters of 0.98 and 1.05 ft, with one recently having its top break-off.

Figure 30. Black cottonwood R4_3 in Upper 8-Floodplain 25August2021.

Figure 31. Black cottonwood R4_4 in Upper 8-Floodplain 25August2021.

Median ABIs (i.e., P-value = 50%) at R4_03 for the four RYs (Figure 32) were 65.0, 65.0, 136, and 25.0 mm respectively (WY2016-21); median R4_04 ABIs for the four WYs (Figure 33) were 15.0 mm, 54.0 mm, 88.5 mm, and 264 mm respectively. Under *saturated* shallow groundwater WY2021 conditions (saturated soil surrounding both trees, but no standing water), one median ABI was 25 mm and the other 264 mm. Under dry WY2016 conditions their median ABIs were 65.0 mm and 15 mm respectively. Again, widely different medians. Having each tree serve as its own baseline for vigor seems the most informative and consistent, given the many small channels throughout Upper 8-Floodplain.

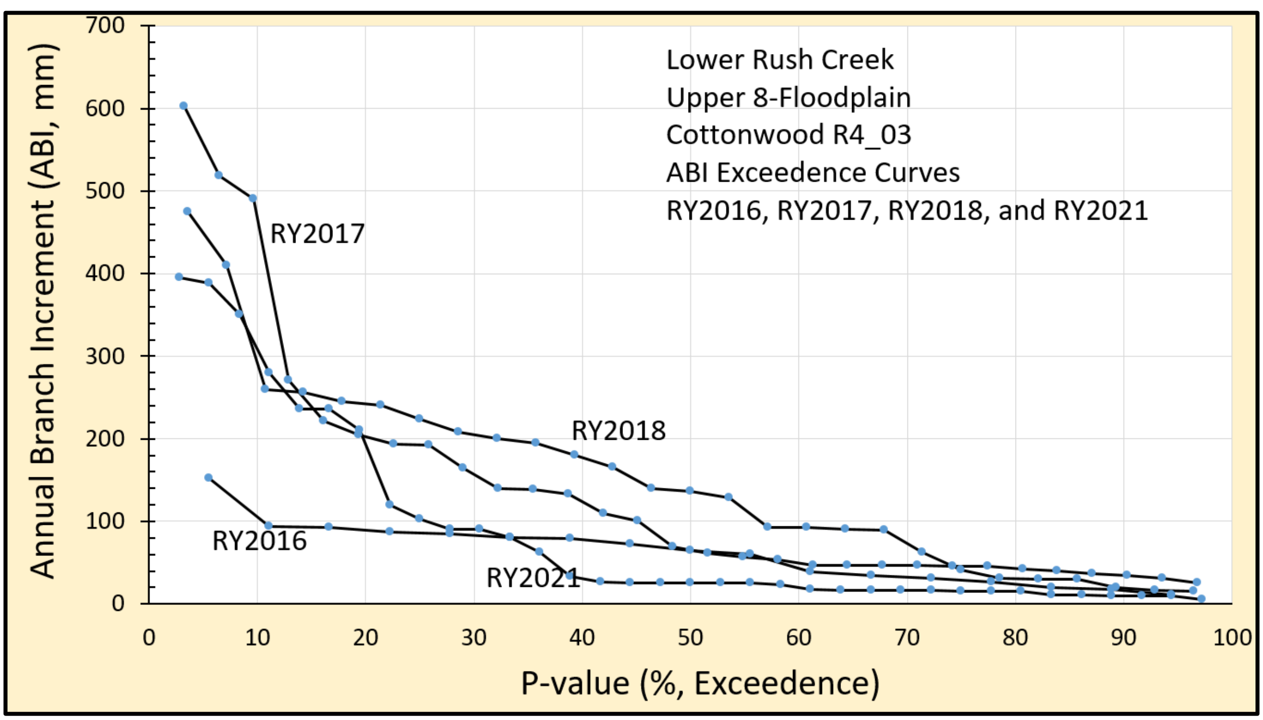


Figure 32. R4_03 ABI exceedence curves for RY2016, RY2017, RY2018, and RY2021.

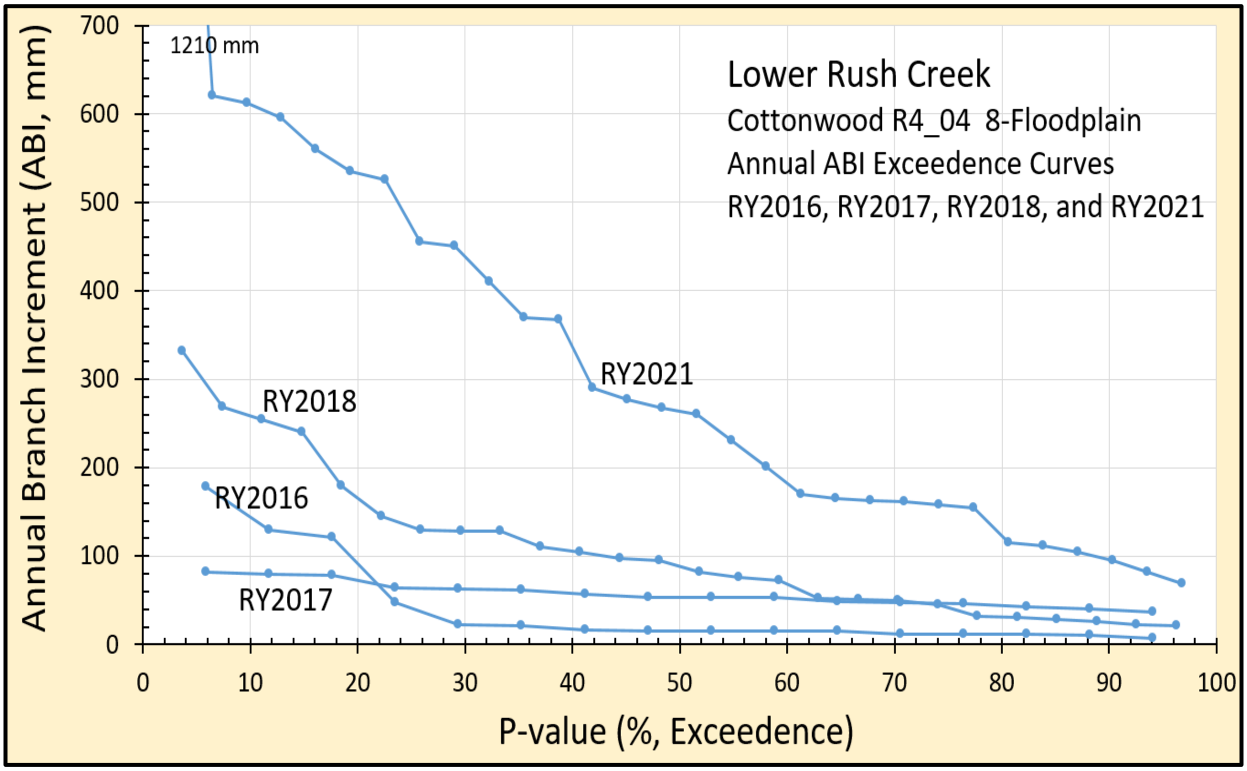


Figure 33. R4_04 ABI exceedence curves for RY2016, RY2017, RY2018, and RY2021.

RY2021 Black Cottonwood ABI Vigor in Central 4-Floodplain

Cottonwood R5_28 is in the central section of the 4-Floodplain (Figure 34), approximately mid-way between the RB valley wall and mainstem Rush Creek. This tree was sampled August 27th in RY2021 (Figure 35) thinking it was a tree surveyed RY2016 through RY2018.

However, when later reviewing the RY2021 field maps for tree locations, this location was listed as a yellow willow. Given the uncertainty, I was going to omit the sample but there remained one potential use. Cottonwood R5_28 only receives surface flows when the mainstem streamflow enters via the 4bii Side-Channel at the upstream end of the 4-Floodplain. A survey of the 4bii Entrance on 26August2021 with Robbie Di Paolo determined that the RY2021 peak flow (QPK = 240 cfs) did not inundate the 4Bii side-entrance invert by 0.16 ft. The ABI exceedence curve for R5_28 in RY2021 does reveal poor vigor (Figure 36). By examining Figure 36 alone, this would be difficult, if not impossible, to conclude.

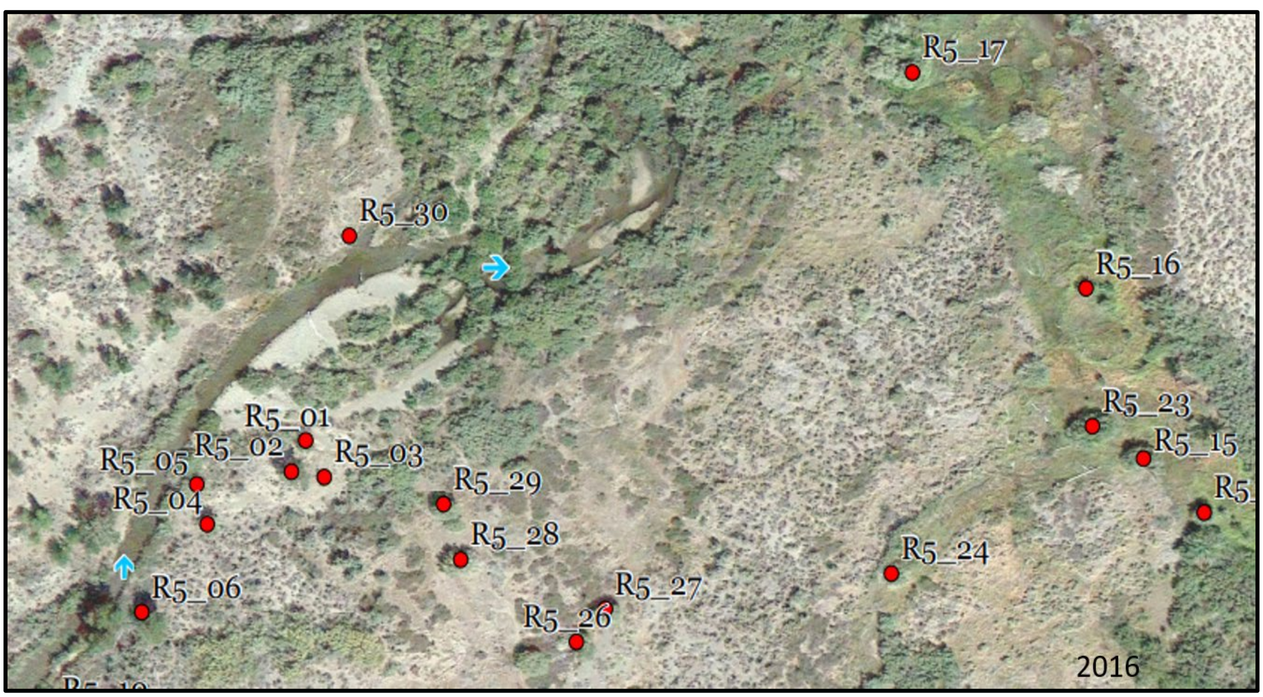


Figure 34. Central 4-Floodplain with black cottonwood and yellow willow ABI tree locations.

Figure 35. Cottonwood R5_28 in Central 4-Floodplain on 27August2021.

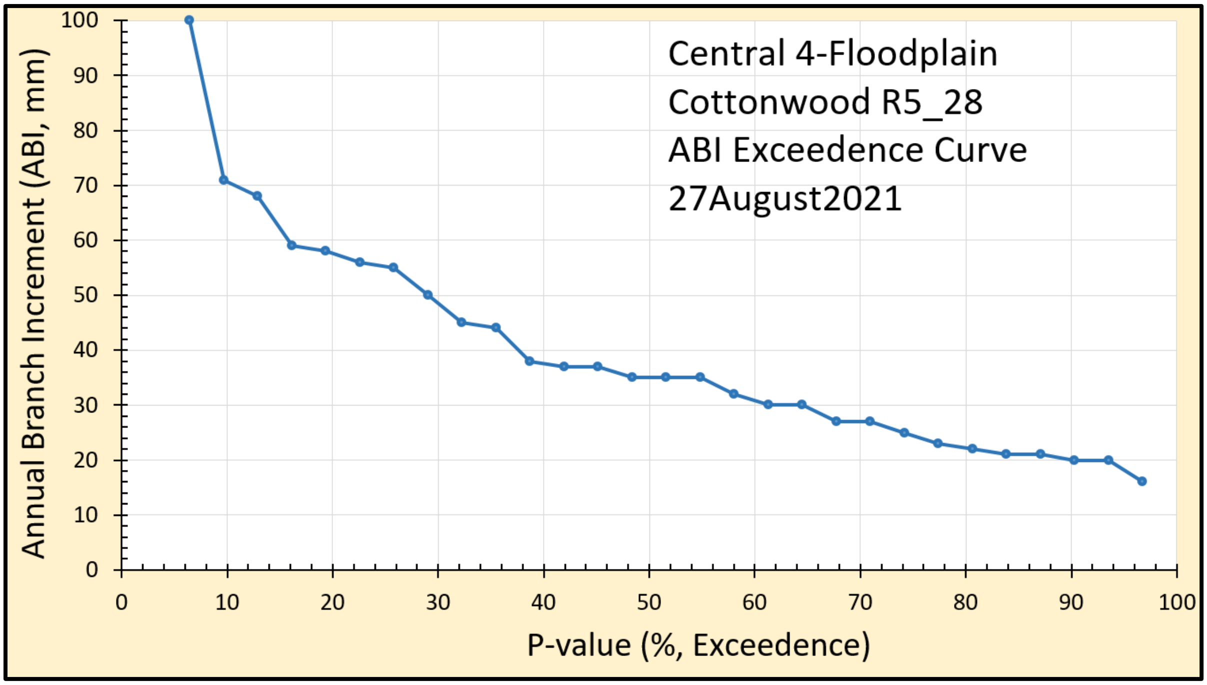


Figure 36. Black cottonwood R5_28 ABI exceedence curve in Central 4-Floodplain in RY2021.

There was one additional baseline this WY2021 R5_28 ABI exceedence curve could be compared. Figure 37 was the outcome of a RY2015 pilot survey of annual branch increments taken from 27 cottonwoods throughout the Lower Rush Creek 3-Floodplain, 4-Floodplain, and 8-Floodplain. Branches were taken back to Humboldt State University (now Cal Poly Humboldt) to estimate ABIs of each stem dating back to RY2009 (refer to LADWP Annual Report 2019 for methodology). RY2014 and

RY2015, both dry water years, had ABI exceedence curves very similar to the RY2021 curve.

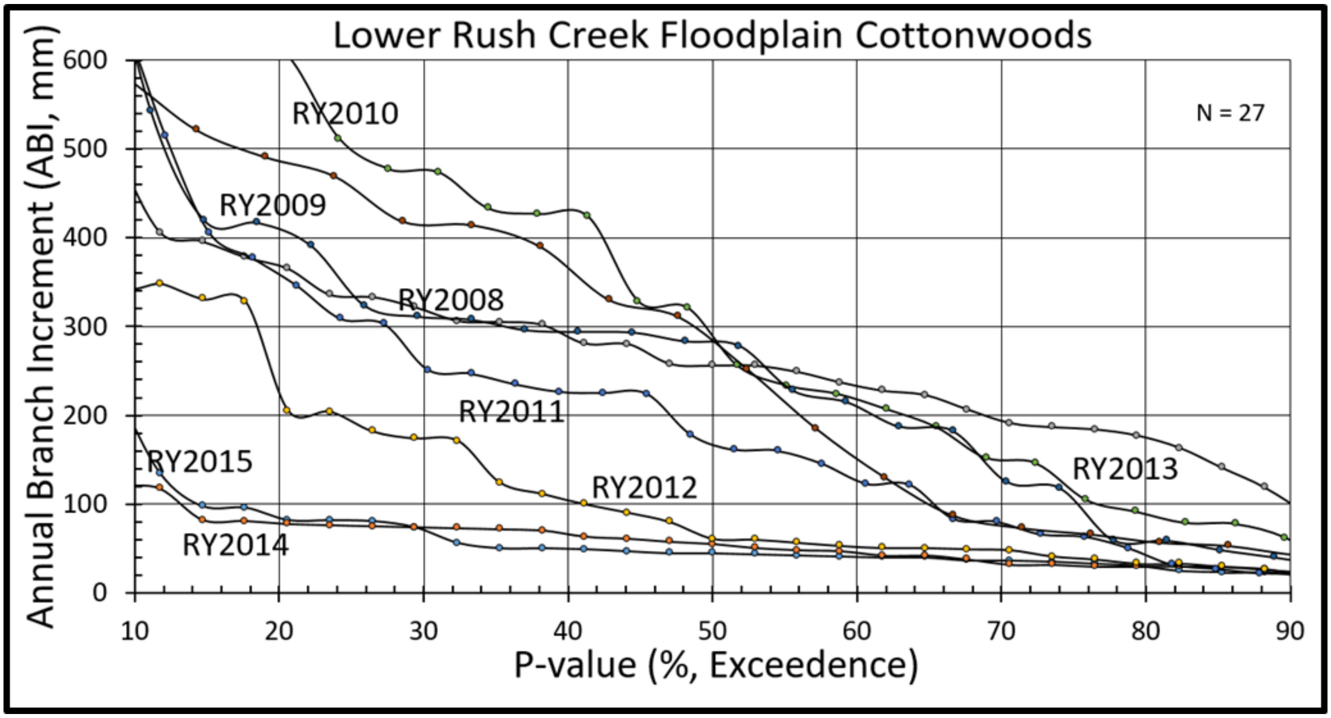


Figure 37. RY2015 pilot cottonwood survey of annual branch increments throughout Lower Rush Creek.

RY2021 Yellow Willow ABI Vigor in Central 4-Floodplain

Yellow Willow R5_26 is located in the central 4-Floodplain (Figure 34). Its relatively arid surroundings (Figure 35) contrast sharply with the recent-evolving wetland in the 8-Floodplain. The median ABI for R4_06 (in the 8-Floodplain) and R5_26 in the 4-Flodplain

(Figure 36) for RY2021 was 139 mm and 61 mm respectively. Their ABI 20% P-values were 290 mm and 85 mm respectively. The range in yellow willow ABI between runoff years and between the two floodplains was considerably less than the range among cottonwoods, even with the sharply contrasting water availability between the two floodplains. Targeting a smaller percentage change in ABI (at P-values of 20% and 50%) would constitute a significant improvement in vigor … more than expected for cottonwoods.

Figure 35. Yellow Willow R5_26 in Lower Rush Creek central 4-Floodplain. August 27, 2021.

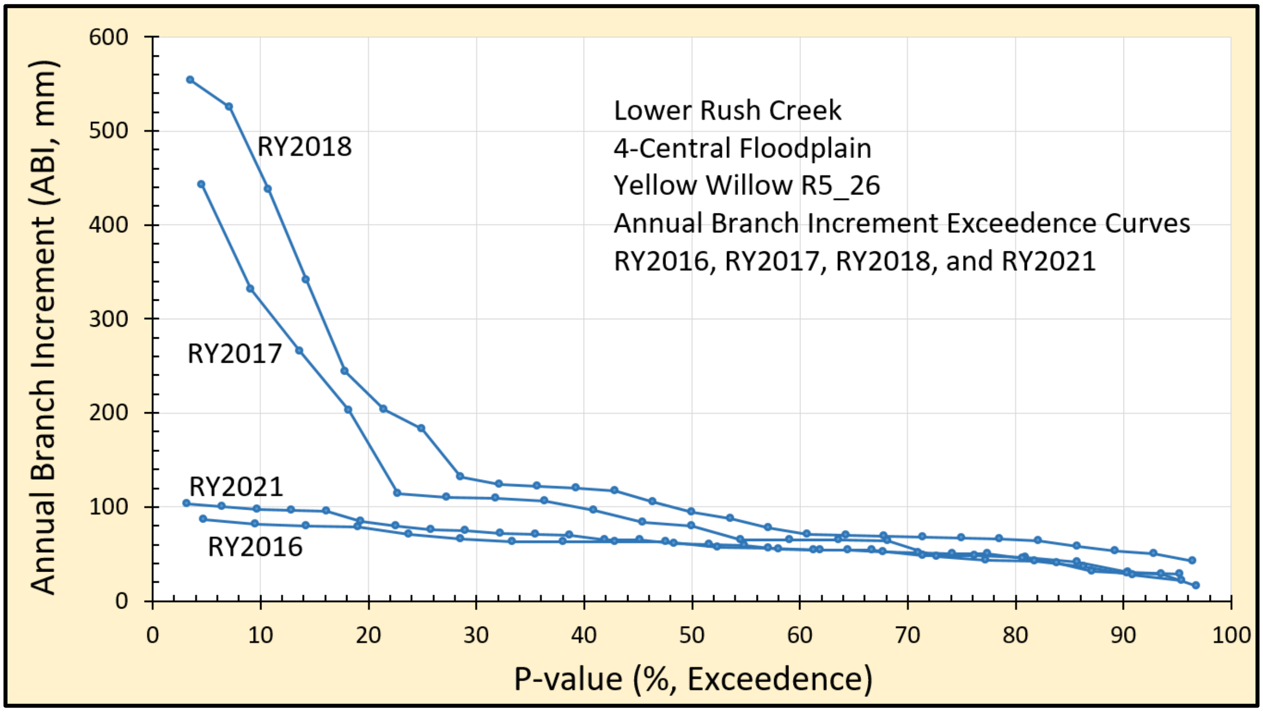


Figure 36. ABI exceedence curves for cottonwood R5_26 for RY2016, RY2017, RY2018, and RY2021.

The four ABI exceedence curves for Yellow Willow R5_26 (Figure 36) were segregated into two general responses. RY2016 and RY2021 resulted in poor vigor and RY2017 and RY2018 resulted in considerably better vigor. Why the differences between RY2017/2018 and RY2016/2021? RY2016 was a dry-normal water year with a peak snowmelt release of 260+ cfs; in RY2021 peak release was 240+ cfs. Streamflow did not enter the 4-Side Channel in RY2021 and only barely entered in RY2016. R5_26 is located along the pathway of the 4-Side Channel as it passes through the center of the 4-Floodplain. In contrast, a peak flood of 900+ cfs in RY2017 provided ample, sustained flow in the side-channel. RY2018 had a peak flood release 450+ cfs providing significant side-channel flow. After observing riparian floodplain responses in many RYs, this one 'given' seems solid: groundwater recharge from the top-down (i.e., by flowing side-channels) is much faster than lateral groundwater recharge from mainstem streamflows. This is particularly important in RYs with modest/poor snowmelt runoff of relatively short duration.

RY2022
Mainstem and Riparian Monitoring Recommendations

(1) Convene a full 1-day workshop mid- to late-summer 2022 on mainstem channel monitoring and riparian vigor assessment (morning classroom and afternoon fieldsites), stressing adaptive management opportunities.
(2) Expand residual depth mainstem reaches to:
    (a) Test Station Road upstream to Ford
    (b) First HU above the 8-Channel Entrance upstream to Grand Old Cottonwood HU
    (c) Lee Vining mainstem reach.
(3) Extend Grand Old Cottonwood Cross-Section through RB and LB floodplains and forensically survey water surface elevation of the RY2017 peak flood.
(4) Locate the 5 most strategic, long-term continuously-recording piezometer locations at/below the Narrows and budget annual cost. Include ABI measurement at these 5 locations to help define floodplain connectivity.
(5) Construct 10 RCT-Q rating curves for each of the three Lower Rush Creek residual pool/run mainstem reaches,

(6) Select ABI measurements 3 times in a single runoff year by re-visiting the same branches of 10 cottonwoods and 10 yellow willows.

Section 4

Mono Basin Waterfowl Habitat Restoration Program
2021 Monitoring Report

Mono Basin Waterfowl Habitat Restoration Program
2021 Monitoring Report

Prepared by:
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Prepared for the State Water Resources Control Board
and Los Angeles Department of Water and Power
April 2022

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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 2021 under the Mono Basin Waterfowl Habitat Restoration Plan (Plan) (LADWP 1996a), as required by Order 98-05.

Mono Lake experienced an overall decrease in lake level in 2021 as compared to 2020. The peak lake level in 2021 of 6,380.9 feet occurred early in spring from February through March, followed by a continuous decline through the remainder of the year. The lake dropped to its lowest level in 2021 of 6,379.4 feet in October and November, rising slightly to 6379.5 feet in December. At the final lake level read in December, Mono Lake was 1.2 feet lower than in December 2020. Runoff during the 2020-2021 Water Year was 45,585 acre-feet, or 38% of the long-term average. The runoff of the 2020-2021 Water Year was the second lowest since 1935, and only 780 acre-feet higher than the record low of 2014-2015. Input from the two major tributaries (Rush and Lee Vining Creeks) in 2021 was 52,204 acre-feet. The combined input of 52% of the long-term average was insufficient to maintain the lake level.

The winter of 2020-21 was not as warm as the winter of 2019-20 however, both the maximum and minimum temperature remained above the long-term averages. The winter of 2020-21 remained dry with only 2.3 inches of precipitation recorded at Cain Ranch, or 69% of the long-term average. The summer of 2021 was the warmest since 1951 and one of the wettest, ranking 6th since 1931.

Clarity of the lake remained below 1 m throughout for the second year in row. Epilimnetic water temperature in 2021 was below the long-term averages while the hypolimnetic water temperature was above the long-term average, mainly due to holomixis at the end of 2020.

The 2021 *Artemia* population peaked in this second post-meromictic year following the end of meromixis in 2020. The mean *Artemia* population almost doubled from 12,991 m^{-2} in 2020 to 23,177 m^{-2} in 2021 as hypolimniotic ammonium was released to the epilimnion at the end of 2020. The 2021 population peak was the lowest among the five previous peaks in spite of higher ammonium accumulation leading up to the peak. The *Artemia* population centroid was 195 days, falling below 200 days for the first time since 2015.

Lake transparency remained below 1 m all year. Algal food sources appear to be readily available but are not being controlled by *Artemia*, based on secchi readings. In 2021, the *Artemia* population mean (23,177 m^{-2}) was the highest since 2013. *Artemia* populations are currently much lower than during the late 1970s to early 1990s when lake level was lower, and salinity higher, approaching 100 g/L.

The lake-fringing wetland transects demonstrated impacts from feral horse grazing at Warm Springs in terms of significant reductions in live cover accompanied by increases in bare ground. These same changes were not observed at Simon Springs where horse grazing has been more localized, or at DeChambeau Embayment, where horse activity has not been observed.

Waterfowl habitat conditions were fairly similar to 2020, with some notable exceptions. Waterfowl habitat conditions continued to be good at Wilson Creek, and at Simons Springs in 2021, at least in the area west of the fault line. Conditions in the Rush Creek delta deteriorated as flow into the fresh water ponds was reduced, and the ponds became algae-covered. Conditions in the South Shore Lagoons area were poor as very few ponds were present and the shoreline was dry and steeply-eroded along much of its length. At Mill Creek, beaver activity continues, creating open water ponds used by waterfowl. The Warm Springs area continued to be very wet, with multiple shallow, open water ponds, attracting waterbirds to feed and shorebird nesting in places previously unavailable because of dense cover, but currently more open due to intense grazing by feral horses.

The dry year also resulted in low water levels at Bridgeport and Crowley Reservoirs, and conditions similar to 2020.

The saltcedar eradication program being conducted by California State Parks has been very effective. The six sites treated in 2021 represents a small number compared to previous years.

Breeding activity at Mono Lake in 2021 was high with an estimated breeding waterfowl population of 364, or approximately 182 pairs. The 2021 breeding population was significantly above the long-term mean. The 74 dabbling duck broods in 2021 was also significantly higher than long-term average. In 2021, breeding activity was concentrated along the northwest shore at DeChambeau, Mill and Wilson Creeks, and at Simons Spring where conditions were most favorable. Most dabbling duck activity was concentrated in and around nearshore water features, primarily freshwater ponds, freshwater outflow areas around the lake (=”ria”), and brackish ponds. At the Restoration Ponds, waterfowl totals were still below the long-term average, yet brood counts were slightly above. The County Ponds continued to be dry, thus reducing available habitat as compared to previous years.

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. Annual brood numbers have been strongly influenced by the June lake level, above a threshold of 6,382 feet.   Below 6,382 feet, there has been no significant effect of lake level.

Waterfowl use at Mono Lake in fall 2021 was significantly above 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* however, as migrating waterfowl seek to meet the energetic demands of migration.

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 2021, waterfowl use of Bridgeport Reservoir was significantly lower than the long-term mean. Totals at Crowley Reservoir and Mono Lake were significantly higher than their respective long-term means.

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.

We recommend that the second year of the waterfowl time budget study, as required by Order 98-05, be completed. 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. Due to the expanding range of feral horses in the Mono Basin, it is recommended that the wetland and riparian vegetation monitoring program be reinstituted, and that transects in Rush Creek and Lee Vining Creek be conducted in 2022.

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 2021 under Restoration Order 98-05 prior to the license amendments.

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 1996). 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 License 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 Water Rights 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 water rights 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 2021, monitoring conducted under the Program included lake elevation, stream flows, lake limnology and secondary producers, vegetation monitoring in lake-fringing wetlands, saltcedar eradication, waterfowl population surveys and aerial photography of waterfowl habitats. The remainder of this report provides a summary and discussion on the 2021 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. LADWP will evaluate the need to continue this program in 2014 and present findings to SWRCB.	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 2021 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. Decision 1631 and Order 98-05 dictate the instream flows (base flows) and channel maintenance flows (peak flows) for these four tributaries, based on "Runoff Year" type. Runoff Year is the period from April 1-March 31. Runoff year type (Table 3-3) 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 per SWRCB Order 98-05

Runoff Year Type	April 1 Runoff Forecast
Dry	<68.5% of average runoff*
Dry/Normal	between 68.5% and 82.5% of average runoff
Normal	between 82.5% and 107% of average runoff
Wet/Normal	between 107% and 136.5% of average runoff
Wet	between 136.5% and 160% of average runoff
Extreme Wet	> 160% of average runoff

*average runoff based on 1941-1990 average runoff of 122,124 acre-feet

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 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 2021, and for the time period 1990-2021. 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 2021, Mono Lake experienced a period of decreasing lake level (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 2021 of 6,380.9 feet in late winter to early spring (February through April). Due to extremely low runoff in the Mono Basin in 2021, the lake level steadily decreased thereafter, falling below 6,380 feet in September. In December, the lake level was 6379.5 feet for a net decline in lake elevation in calendar year 2021 of 1.3 feet.

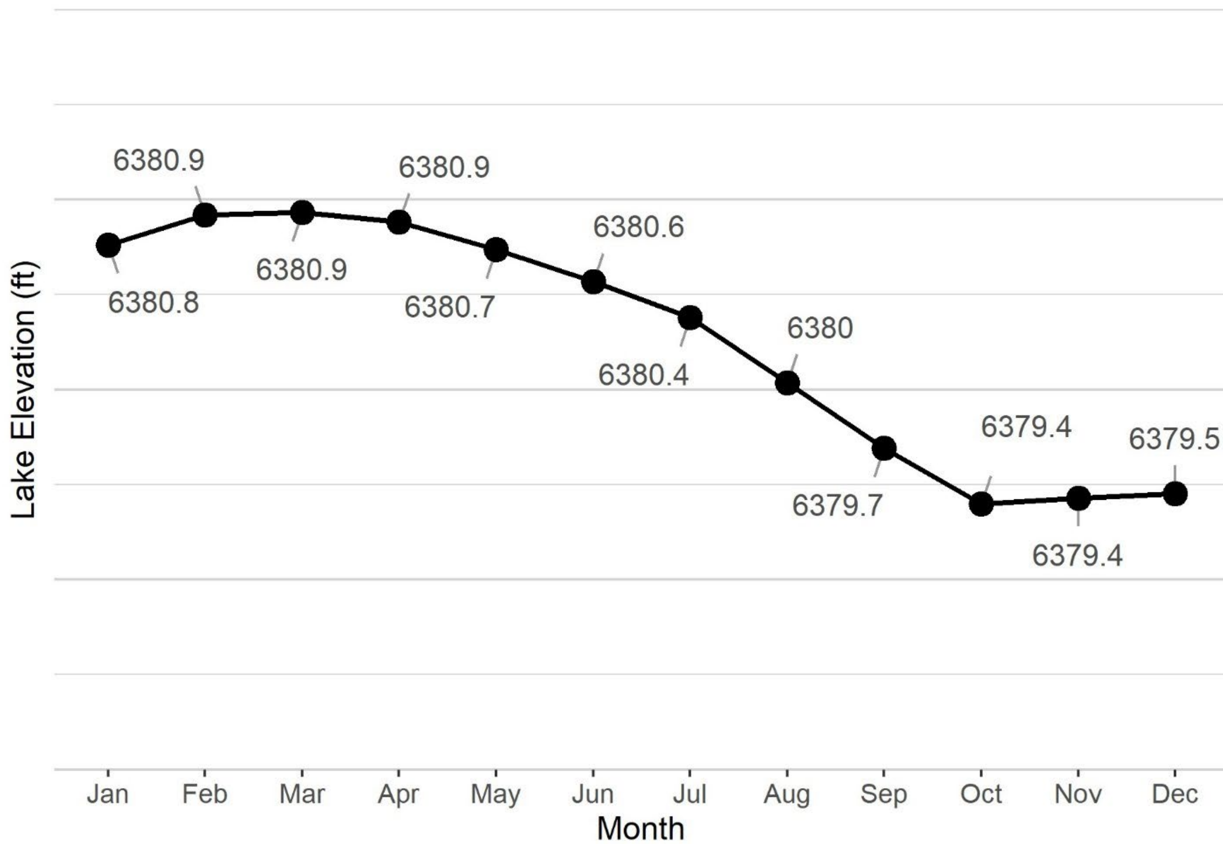


Figure 1. Mono Lake Monthly Elevation - 2021

Runoff during the 2020-2021 Water Year was 45,585 acre-feet, or 38% of the long-term average. The 2020-21 Water Year was the second driest since 1935. The 2020-2021 Water Year was second only to the driest year on record which occurred in 2014-15 with 44,804 acre-feet of runoff (37% of Normal). Starting with the 2011-2012 Water Year (which marked the beginning of an extreme 5-year drought), half of years since then rank within the bottom ten in terms of runoff (2011-12, 2013-14, 2014-15, 2019-20, and 2020-21).

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 146% of normal; the second wet period only lasted two years (2005 to 2006) and averaged 153% of normal; the third wet period also lasted two years (2010 to 2011) and averaged 130% 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 195% 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).

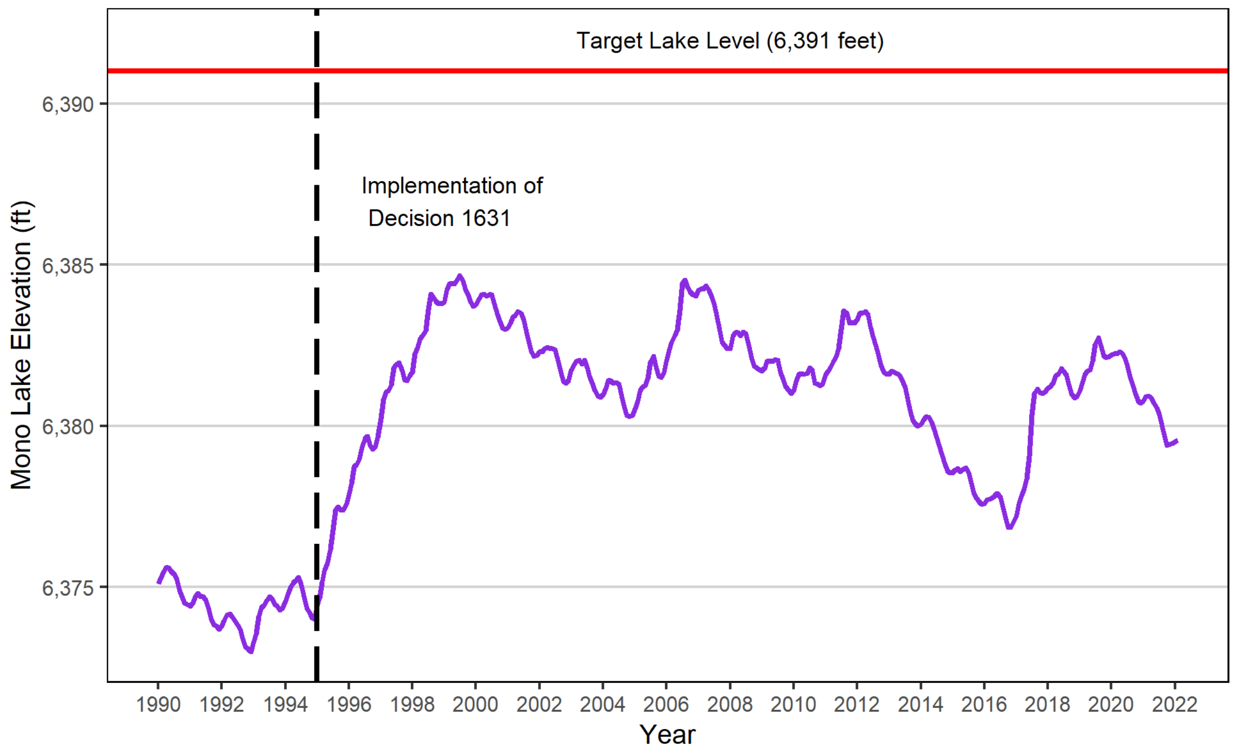


Figure 2. Mono Lake Elevation Between 1990 and 2021

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

Stream Flows

In 2021, the input from Rush Creek was 35,451 acre-feet or approximately 57% of the long-term average since 1990. Additionally, the 2021 input from Rush Creek was 40% of the 2019 runoff and 15% of the 2020 runoff (Table 3-4). Since 1990, Rush Creek has provided the largest inputs to Mono Lake averaging 61,574-acre-foot discharge, with a peak input over this time period in 2017 of 145,349 acre-feet. The input from Lee Vining Creek in 2021 was 16,753 acre-feet, or approximately 44% of the long-term average of 38,290 acre-foot. As was the case with Rush

Creek, the highest input in this time period was 91,133 acre-feet in 2017. Input from the two major tributaries (Rush and Lee Vining Creeks) in 2021 was 52,204 acre-feet, or 52% of the long-term average since 1990. The input from Dechambeau Creek in 2021 was 426 acre-feet, 50% of the long-term mean. DeChambeau Creek has averaged 852 acre-feet since 1944 and has contributed less than 1% of total annual input since 1990. We were unable to get the 2021 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	35,451	16,752	425	NA

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 2021 (6379.5 feet), Mono Lake was 1.3 feet lower than in December 2020. 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 appears to have 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 believed to be largely driven by climate and runoff. An updated lake level model will help determine the influence of various factors currently influencing Mono Lake elevation, including climate change.

Stream Flows

The 2021 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 2021. 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 blue-green algae (*Oscillatoria* spp.) and filamentous green algae (*Ctenocladus circinnatus*) and the diatom *Nitzchia frustulum* dominant the benthic algal community.

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 (Jones and Stokes Associates, Inc. 1993).

Within the hydrographically closed basin, the particular 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. The Marine Science Institute (MSI), University of California, Santa Barbara served as the principle investigator, and Sierra Nevada Aquatic Research Laboratory (SNARL) provided field sampling and laboratory analysis technicians until July 2012. 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 2021, 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 parameters 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 Paoha Island measuring 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 casing with the radiation shield dislodged, resulting no wind data and erratic readings in relative humidity and temperature. Daily precipitation and 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 dating from October 1950 were obtained from the Western Regional Climate Center (www.wrcc.dri.edu) and analyzed to gain better insight into climatic trends. Winter temperature was calculated by averaging the monthly average maximum (or minimum) temperature from December of the previous year and January and February of the subsequent year. For example, the monthly average from December 2018 was combined with the monthly average from January and February 2019 to obtain the winter average for 2019. Summer temperature was calculated as the average monthly temperature between June and August.

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 3-5. 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 persisting snow cover impeding the lake access, no sampling was conducted in February and December 2021. CTD malfunction prevented CTD monitoring September-November 2021. 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 2021

Month	Sampling Dates			
	NH4, CHLA	DO	CTD	Artemia, Secchi
Feb	-	-	-	-
Mar	3/16/2021	3/16/2021	3/16/2021	3/16/2021
Apr	4/22/2021	4/22/2021	4/29/2021	4/29/2021
May	5/18/2021	5/18/2021	5/18/2021	5/18/2021
Jun	6/18/2021	6/18/2021	6/17/2021	6/17/2021
Jul	7/20/2021	7/20/2021	7/20/2021	7/20/2021
Aug	8/25/2021	8/25/2021	8/20/2021	8/20/2021
Sep	9/21/2021	-	-	9/21/2021
Oct	10/21/2021	10/21/2021	-	10/21/2021
Nov	11/17/2021	11/17/2021	-	11/17/2021
Dec	-	-	-	-

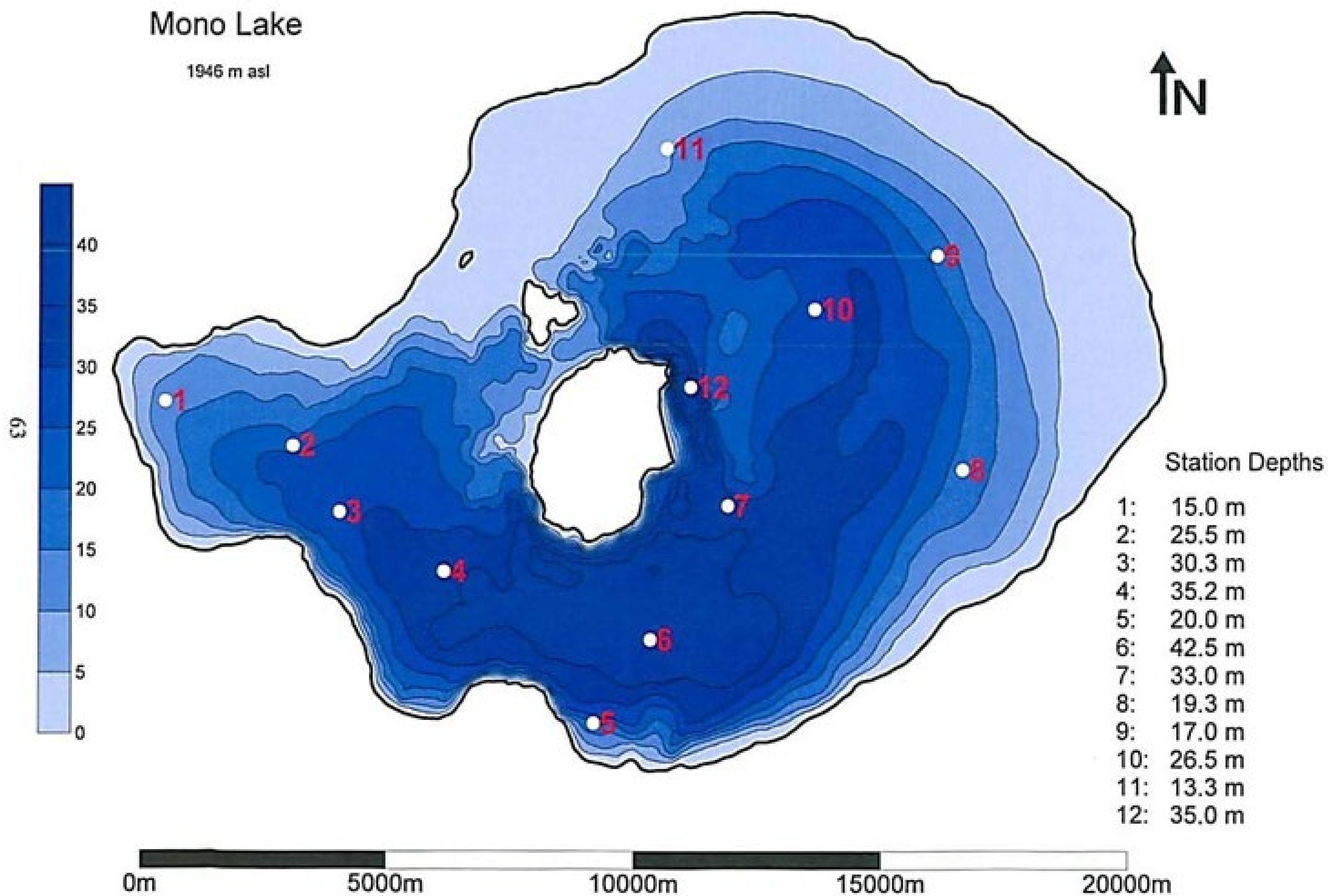


Figure 3. Sampling Stations at Mono Lake and Associated Station Depths

Physical and Chemical

Transparency

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

Temperature, Conductivity, and Salinity

A Sea-Bird high-precision conductivity temperature-depth (CTD) profiler 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 presented by Jellison in past compliance reports (LADWP 2004).

A formula to calculate the conductivity adjusted 25°C had been copied with a minor error, resulting in artificially elevated adjusted conductivity and salinity values. 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 2012-2020. Discrepancies between the errored and correct formulas exceeded 10g/L in the hypolimnion. Current salinity levels are higher than the long-term average but no higher than the levels seen in early 90s when the lake level was a few feet lower.

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.

Ammonium Sampling

Monitoring of ammonium in the epilimnion was conducted using a 9-m integrated sampler at stations 1, 2, 5, 6, 7, 8, and 11. Ammonium was sampled at eight discrete depths (2, 8, 12, 16, 20, 24, 28, and 35 meters) at Station 6 using a vertical Van Dorn sampler. Samples were immediately sent to BSK Associates for analysis. A change in the consultant and laboratory unfortunately resulted in the ammonium analysis not being conducted as requested by LADWP.

The procedure used was not sensitive enough to detect ammonium levels of Mono Lake water, and the results are therefore not presented in this report.

Chlorophyll *a* Sampling

Monitoring of chlorophyll *a* in the epilimnion was conducted using a 9-m integrated sampler at stations 1, 2, 5, 6, 7, 8, and 11. Chlorophyll was sampled at Station 6 at seven discrete depths (2, 8, 12, 16, 20, 24, and 28 meters) using a vertical Van Dorn sampler. Water samples were immediately sent to Brelje and Race Laboratories, Inc. for analysis. A change in the consultant and laboratory, unfortunately resulted in the chlorophyll analysis not being conducted as requested by LADWP, and thus the results are not presented in this report as they would not be meaningful.

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* analyses. 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).

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 time period 1979-2019.

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.

3.2.2 Limnology Data Analysis

Salinity and Mono Lake Elevation

The salinity of Mono Lake is directly influenced by water inputs and lake elevation due to the hydrographically-closed nature of the basin. 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 the tolerance level, which ranges from 159 g/L to 179 g/L (Dana and Lenz 1986). Long-term relationships between lake levels and salinity at three different depths (between 0 and 10 m, between 11 and 20 m, and deeper than 21 m) were examined in this section. Lake elevation data collected as part of the hydrologic monitoring program (Section 3.1.1) was used for this analysis.

Artemia Population Peak

Meromixis has been demonstrated to affect the *Artemia* population in Mono Lake as stratification prevents the release of hypolimnetic ammonium during meromixis. During periods of meromixis, ammonium accumulates in the hypolimnion. With a weakening chemocline, ammonium supply to the epilimnion or mixolimnion increases. This process also allows oxygenation of the hypolimnion, which remains suboxic to anoxic during meromixis. Usually one year after the breakdown of meromixis, the *Artemia* population booms. The salinity gradient as determined by the preceding salinity, and lake inputs, are important aspects affecting the strength and duration of a chemocline, which, in turn, dictates the magnitude of ammonium accumulation. Meromictic events were characterized by salinity gradient and ammonium (NH4) accumulation, in order to evaluate post meromictic *Artemia* population peaks.

Ammonium levels recorded at the two deepest monitoring depths (28 and 35 m) generally show trends similar to *Artemia* population peaks, except 2012. Previously, the peak accumulation during meromixis was reported; however, the meromixis breaks down one or two years before the peak and some accumulated ammonium gets released prior to the peak. It is, therefore, more representative to report accumulated ammonium just before holomixis. Further, ammonium accumulation in the fall just prior to the peak was available for the 1989 peak, and a comparison of all five recorded post meromictic *Artemia* population peaks became possible.

A Temporal Shift in Monthly Artemia Population

A temporal shift in peak *Artemia* population or centroid has been noted by Jellison in previous years' compliance reports. LADWP also has reported a continuation of this trend in the *Artemia* instar population (LADWP 2017). Two water parameters - chlorophyll *a* and temperature - have been demonstrated to affect development of *Artemia*. For instance, spring generation *Artemia* raised at high food densities develop more quickly and begin reproducing earlier. In addition, the abundance of algae may likely affect year-to-year changes in *Artemia* abundance (Jellison and Melack 1993). Cysts of Mono Lake brine shrimp require three months of dormancy in cold water (<5°C) to hatch (Dana 1981, Thun and Starrett 1986) and the summer generation of *Artemia* grows much more quickly than the spring counterpart because of warmer epilimnetic water temperature. For adult development, summer epilimnetic water temperature could affect *Artemia* abundance even though other factors such as food availability confounds growth rate (Jones and Stokes Associates 1994).

In this section, monthly *Artemia* abundance (adult and instar) was quantitatively and qualitatively compared to monthly readings of chlorophyll *a* and temperature in order to understand the mechanisms associated with the temporal shifts in *Artemia* population abundance. All analyses were performed using the statistical software, R (The R Project for Statistical Computing).

3.2.3 Limnology Results

Meteorology

Air Temperature

Daily maximum air temperature ranged from -2.9°C to 36.7°C (Figure 4). The daily maximum of 36.7°C was recorded on June 17, July 9, and August 16 and the minimum daily maximum of -2.9°C was recorded on January 27. The daily minimum temperature ranged from -17.8°C to 13.9°C. The daily minimum was recorded on January 26 and December 17 while the maximum daily minimum of 13.9°C was recorded on July 7. The average winter temperature (January through February) was 0.5°C, or 1.7°C higher than the long-term average since 1990, while the average maximum summer temperature (June through August) was 19.8°C, or almost 3°C higher than the long-term average.

Precipitation

The total precipitation between January 1 and December 31 measured at LADWP Cain Ranch was 12.4 inches. Precipitation events were most frequent (10) in July in 2021 and the largest single day total precipitation of 2.14 inch was recorded on October 25 (Figure 5). Monthly precipitation in 2021 did not follow the long-term seasonal pattern especially during spring and fall during which very little precipitation was recorded. In January, 2.66 inches of precipitation was recorded but between February and April monthly precipitation did not exceed 0.5 inches. July precipitation was 2.15 inches followed by almost no precipitation in August and September. The October snow storm brought much needed early winter precipitation, but November was dry and warm. A series of snow storms in December brought 1.95 inches of precipitation.

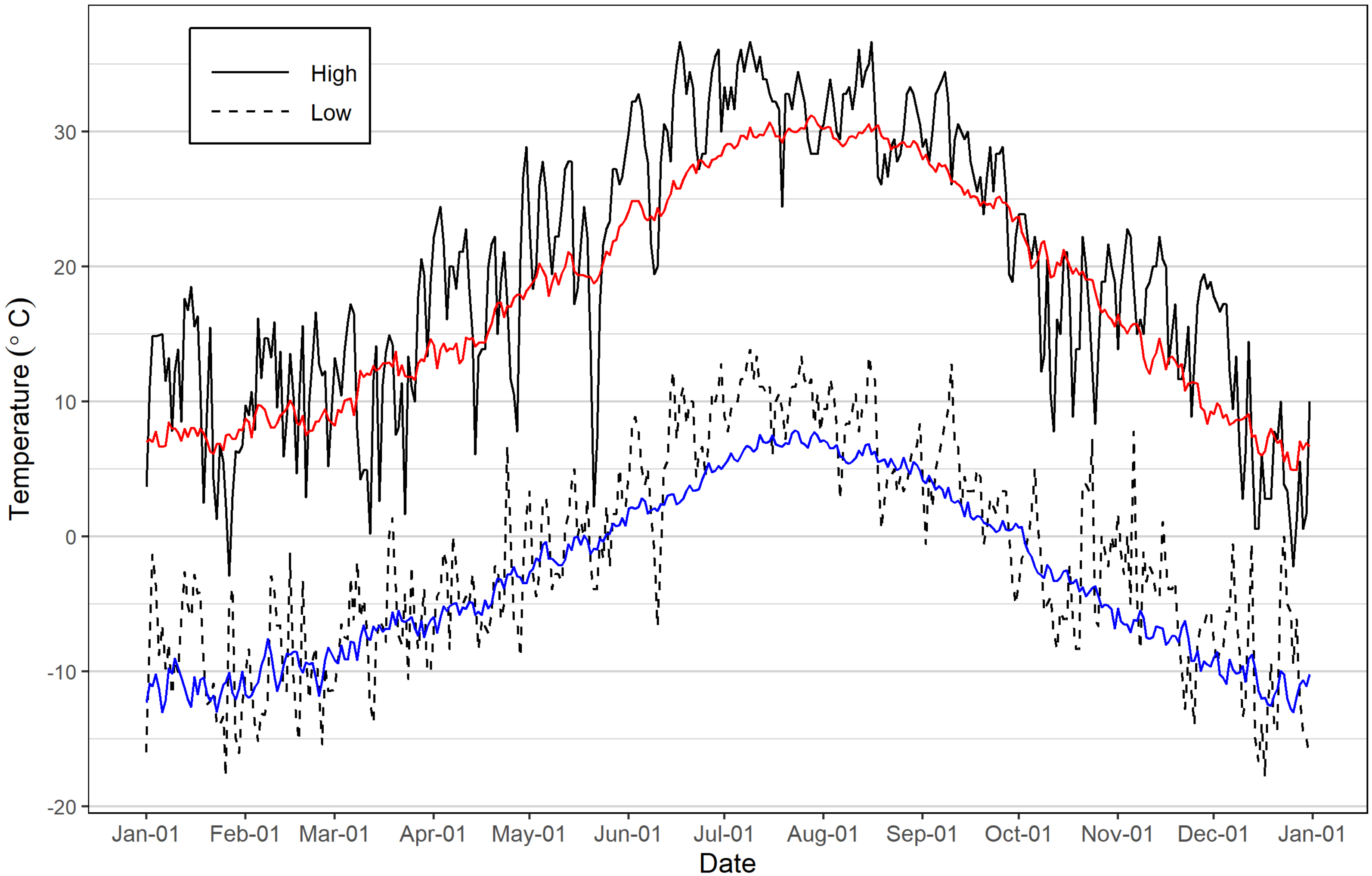
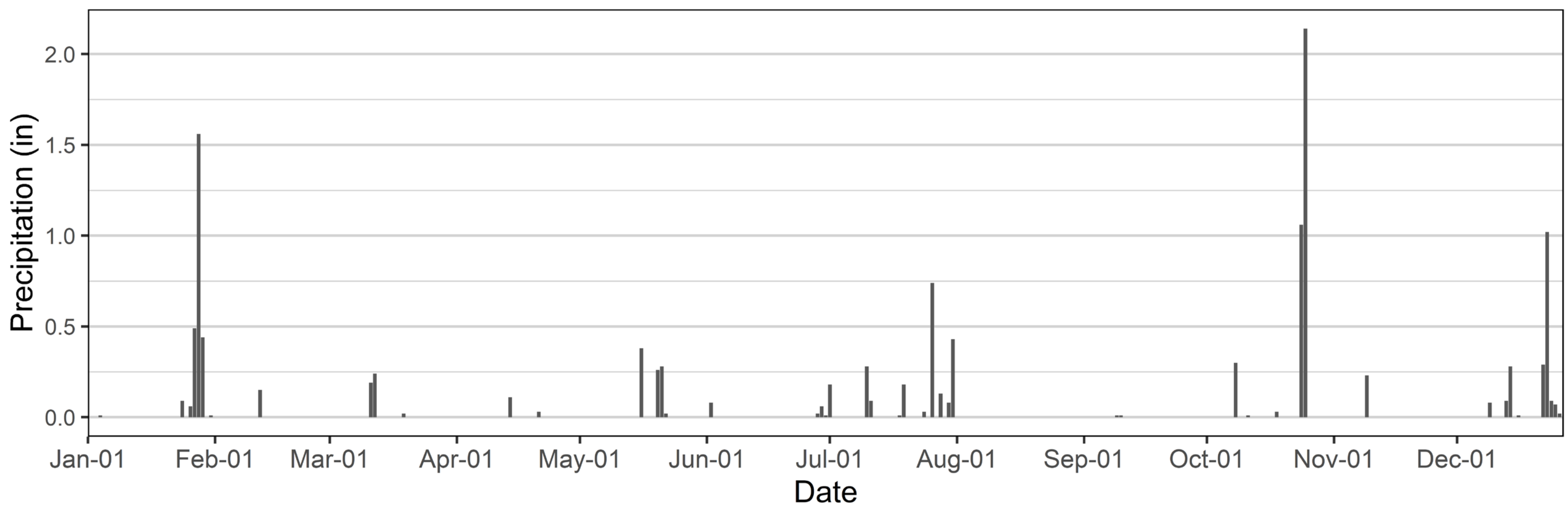


Figure 4. Minimum and Maximum Daily Air Temperature (°C)

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.



Daily Precipitation Summary

Month	Total	Freq
Jan	2.66	7
Feb	0.15	1
Mar	0.45	3
Apr	0.14	2
May	0.94	4
Jun	0.17	4
Jul	2.15	10
Aug	0.00	0
Sep	0.02	2
Oct	3.54	5
Nov	0.23	1
Dec	1.95	9
Total	12.40	48

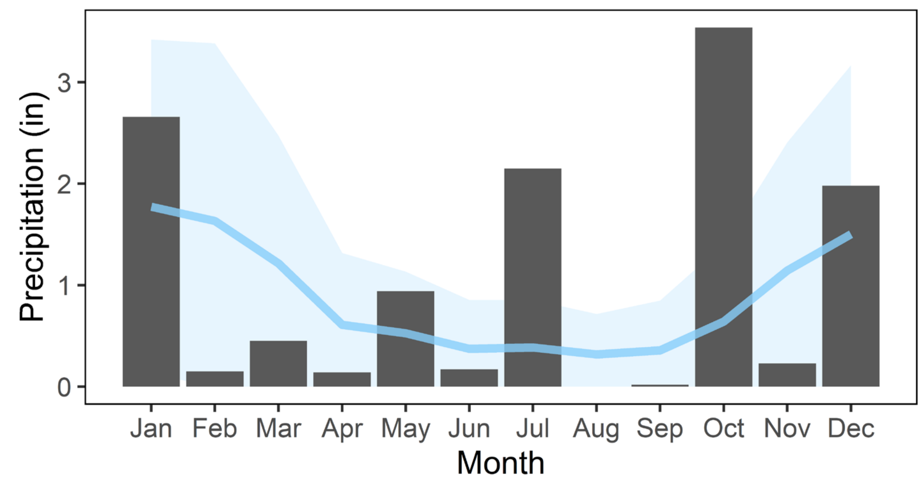


Figure 5. Total Daily Precipitation (in) at the top, Daily Precipitation at the bottom left, and Monthly Precipitation at Bottom Right

Precipitation was recorded at LADWP Cain Ranch Weather Station.

Long Term Trends in Temperature and Precipitation

The year 2021 started with a slightly warmer January followed by colder February and March (Figure 6). Between April and September, the monthly average temperature remained above the long-term average (LTA). It was particularly warmer in June and July when the monthly average temperature was 3.5°C and 2.4°C higher than LTA of the respective month. October was much colder (2.1°C below LTA) but November was much warmer (2.6°C above LTA). December was right around the long-term average.

The winter of 2020-21 followed the recent trend of above LTA for both maximum and minimum monthly average temperature (Figure 7). The maximum monthly average was 0.4°C above LTA and ranked 33rd while the minimum monthly average was -5.9°C above LTA and ranked 32nd. The summer of 2021 was warmest on record with maximum and minimum monthly averages of 2.4°C and 2.5°C above the respective LTA (Figure 8). Winter precipitation in 2020-21 (3.39 in) remained below LTA for the second winter in row and was ranked 56th in 90 years and 69% of LTA (Figure 9). Summer precipitation exceeded 200% of LTA with 2.32 inches recorded in 2021 (Figure 10). Summer precipitation was ranked 6th and 215% of LTA. The summer of 2021 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 11). This trend is much stronger since 1973 ($r=0.80$, $p<0.0001$) indicating there has been a very strong warming trend in summer minimum temperature from the beginning of the limnology monitoring in 1979. Correlation coefficients increased from 0.03 to 0.28 for winter maximum, 0.05 to 0.51 for winter minimum, and 0.25 to 0.54 for summer maximum compared to the entire data set (>1951) to the data set starting in the early to mid-80s. This increasing positive trend was found still continuously occurring in last 15 years or so as correlation coefficients increased from 0.28 to 0.46 for winter maximum, 0.51 to 0.54 for winter minimum, and 0.54 to 0.64 for summer maximum.

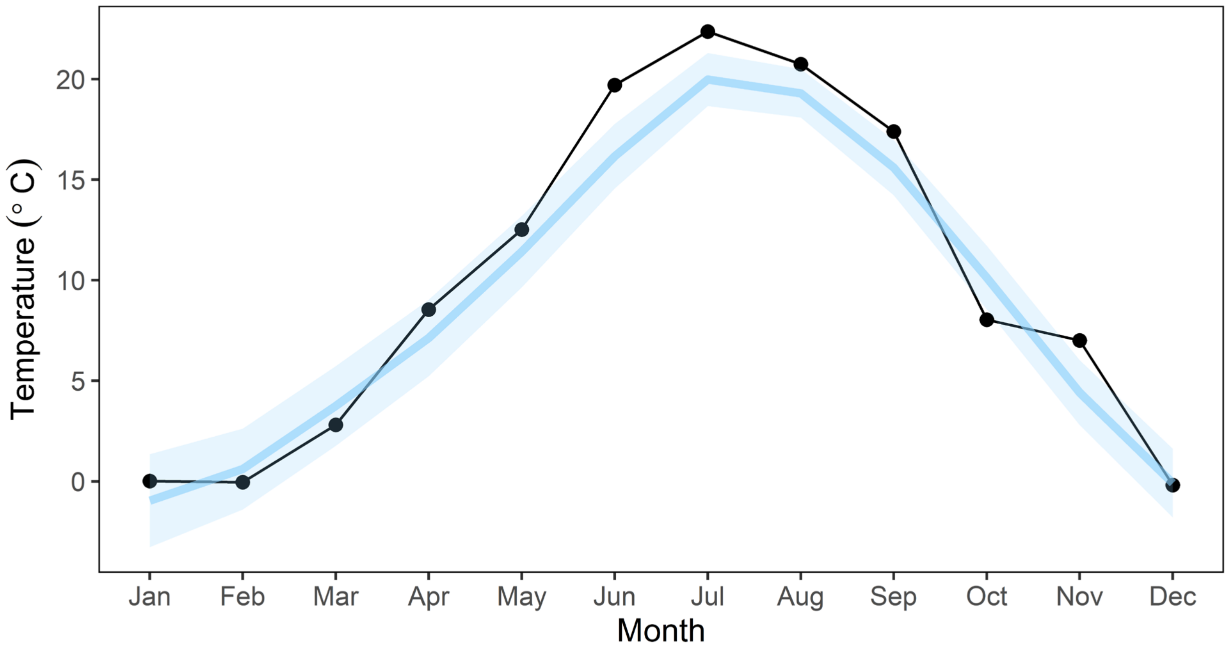


Figure 6. Monthly Temperature in 2021 Compared to the Long-term Averages

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.

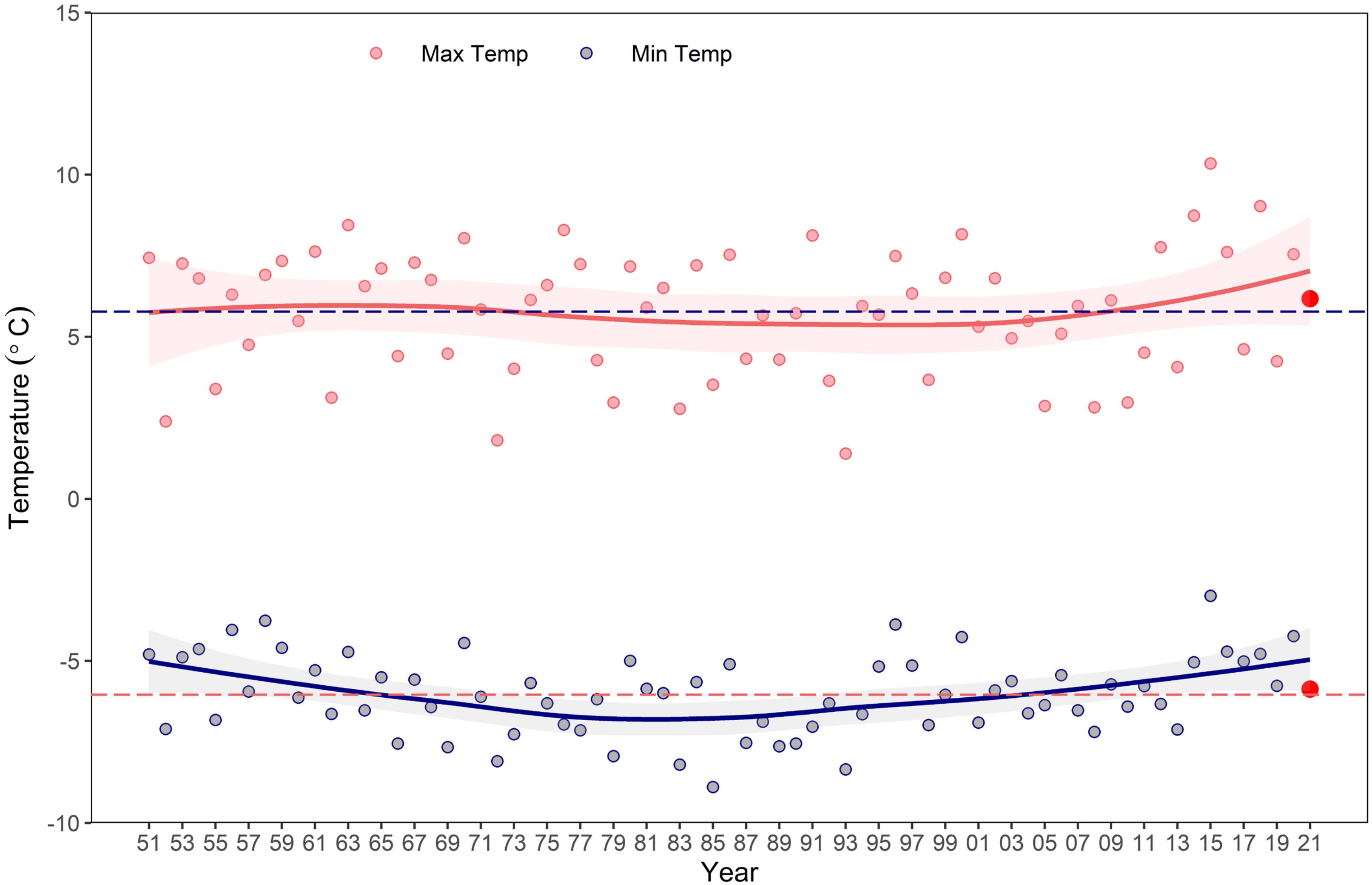


Figure 7. Average Temperature during Winter Months (December through February)

Temperature was recorded at Mono Lake (Station Number 045779-3 obtained) 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.

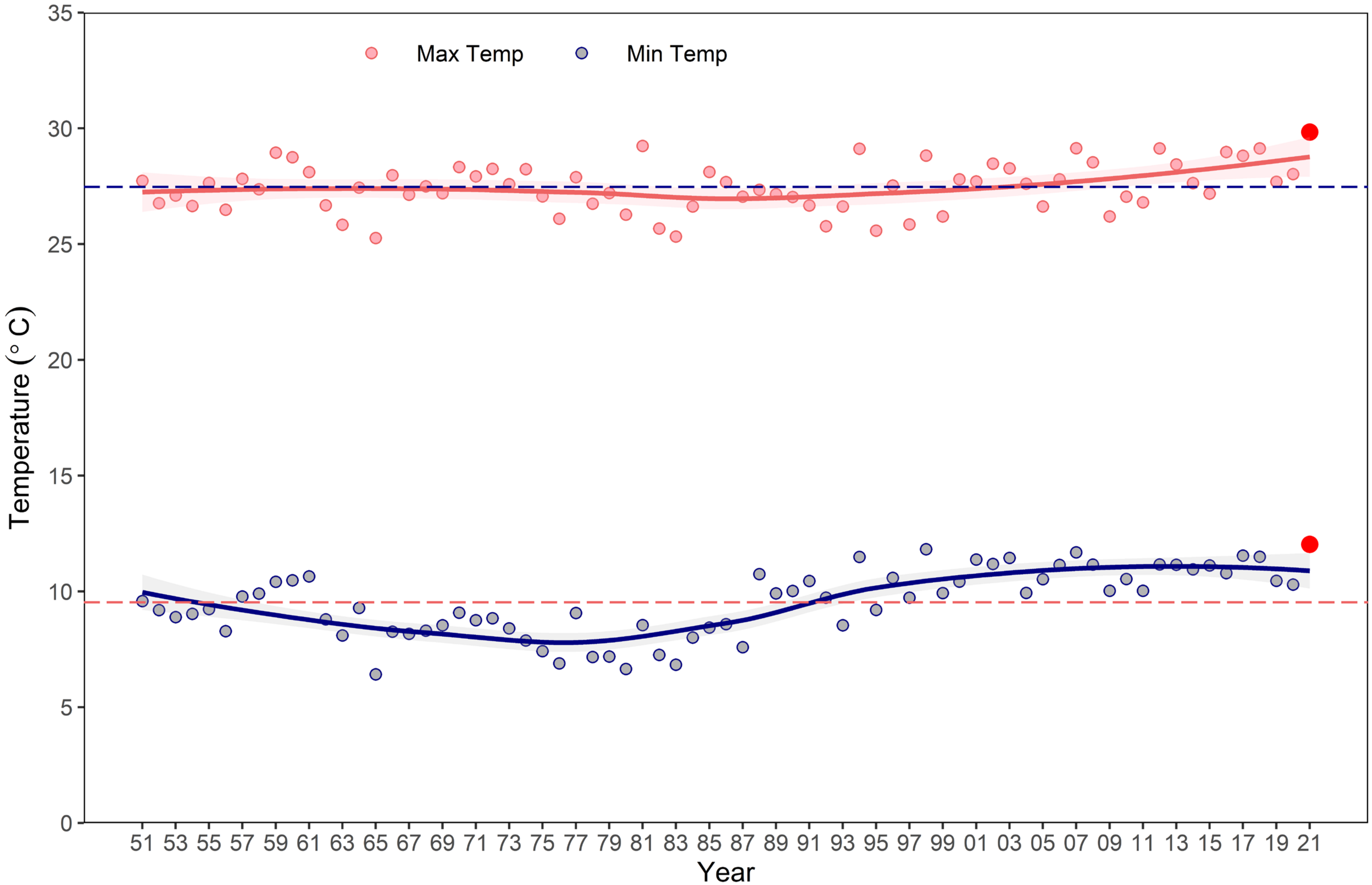


Figure 8. Average Temperature during Summer Months (June through August)

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.

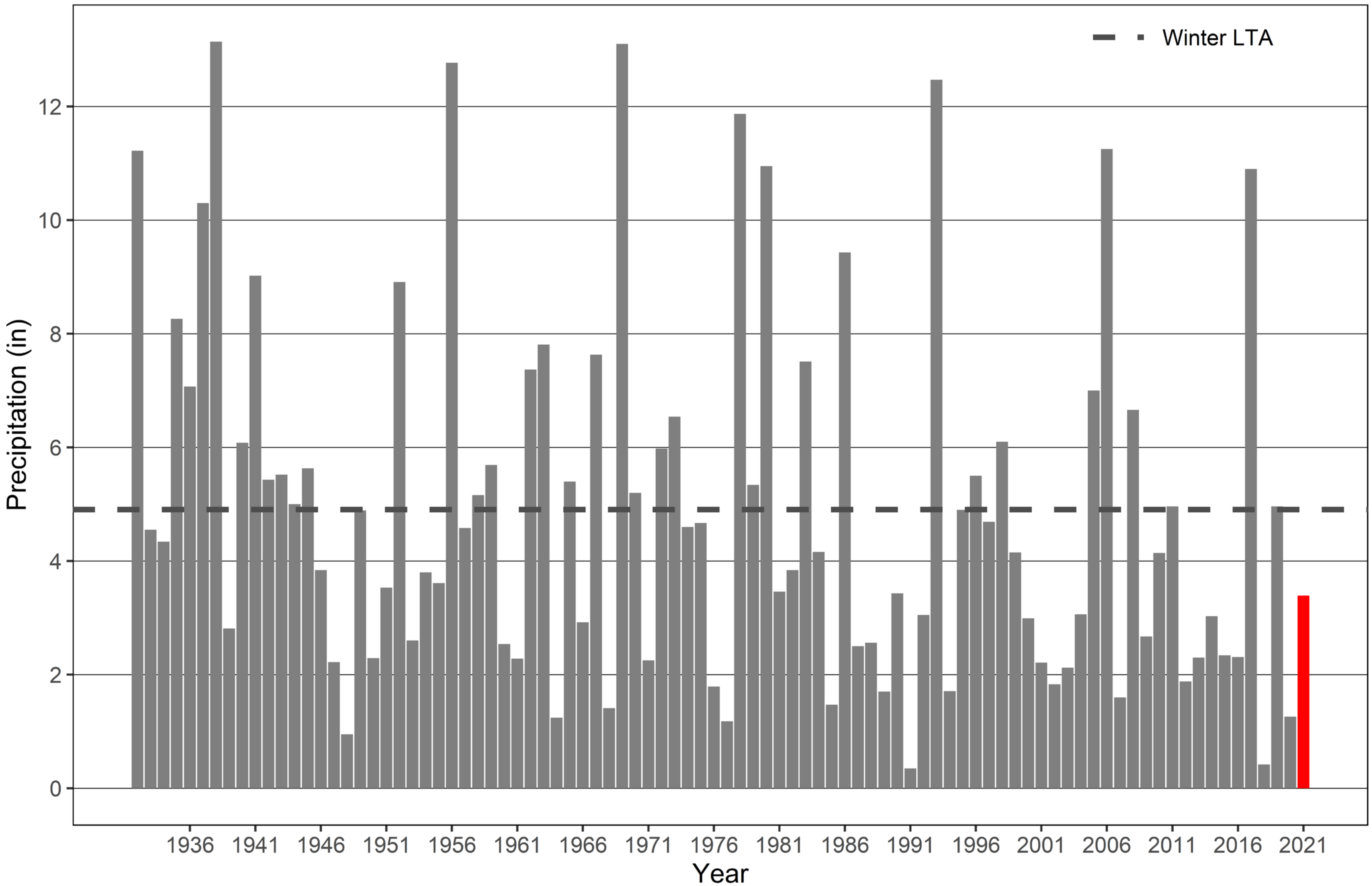


Figure 9. Total Winter Precipitation (December through February)

Precipitation recorded at LADWP Cain Ranch since 1932.

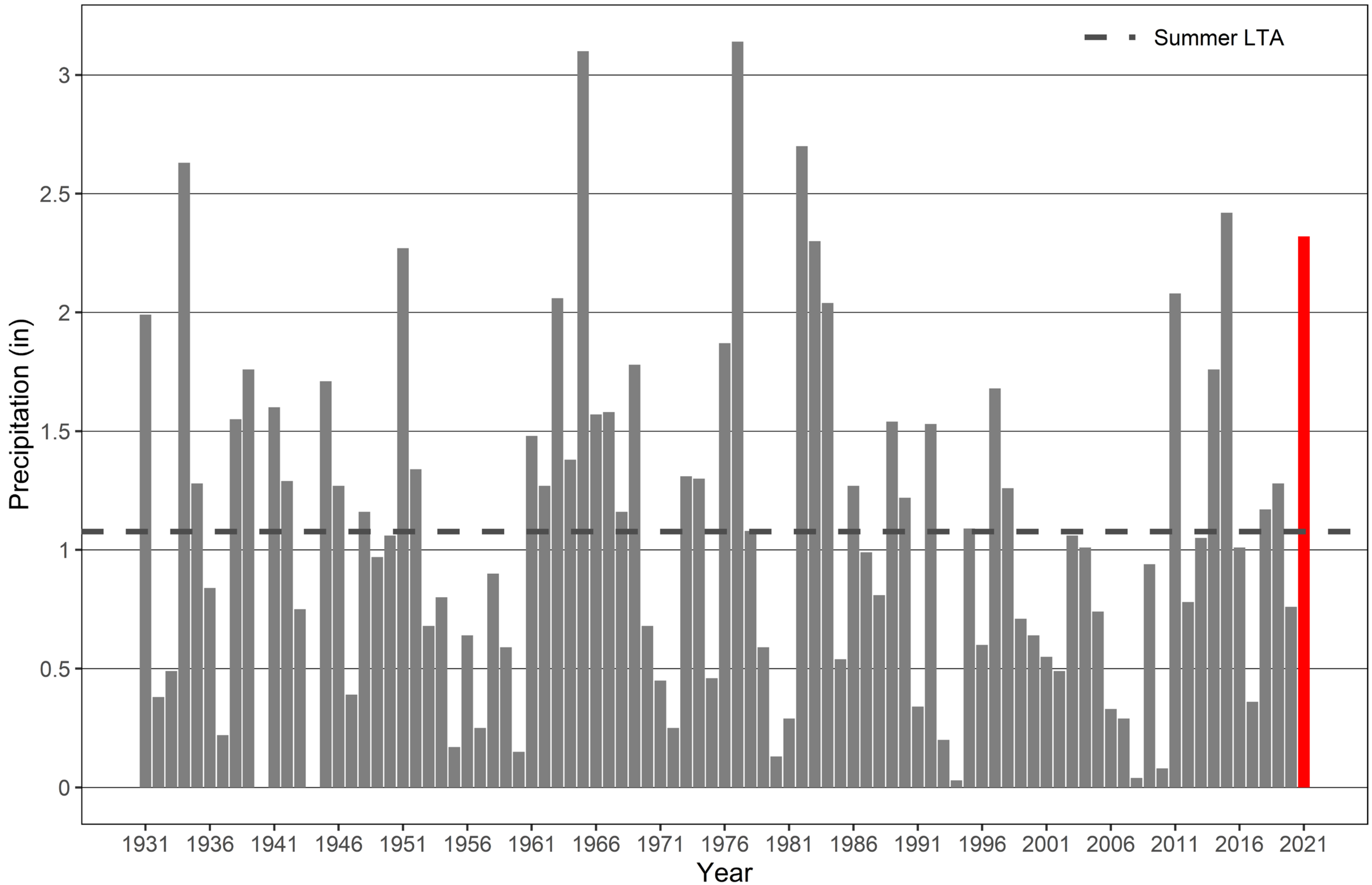


Figure 10. Total Summer Precipitation (June through August)

Precipitation recorded at LADWP Cain Ranch since 1932.

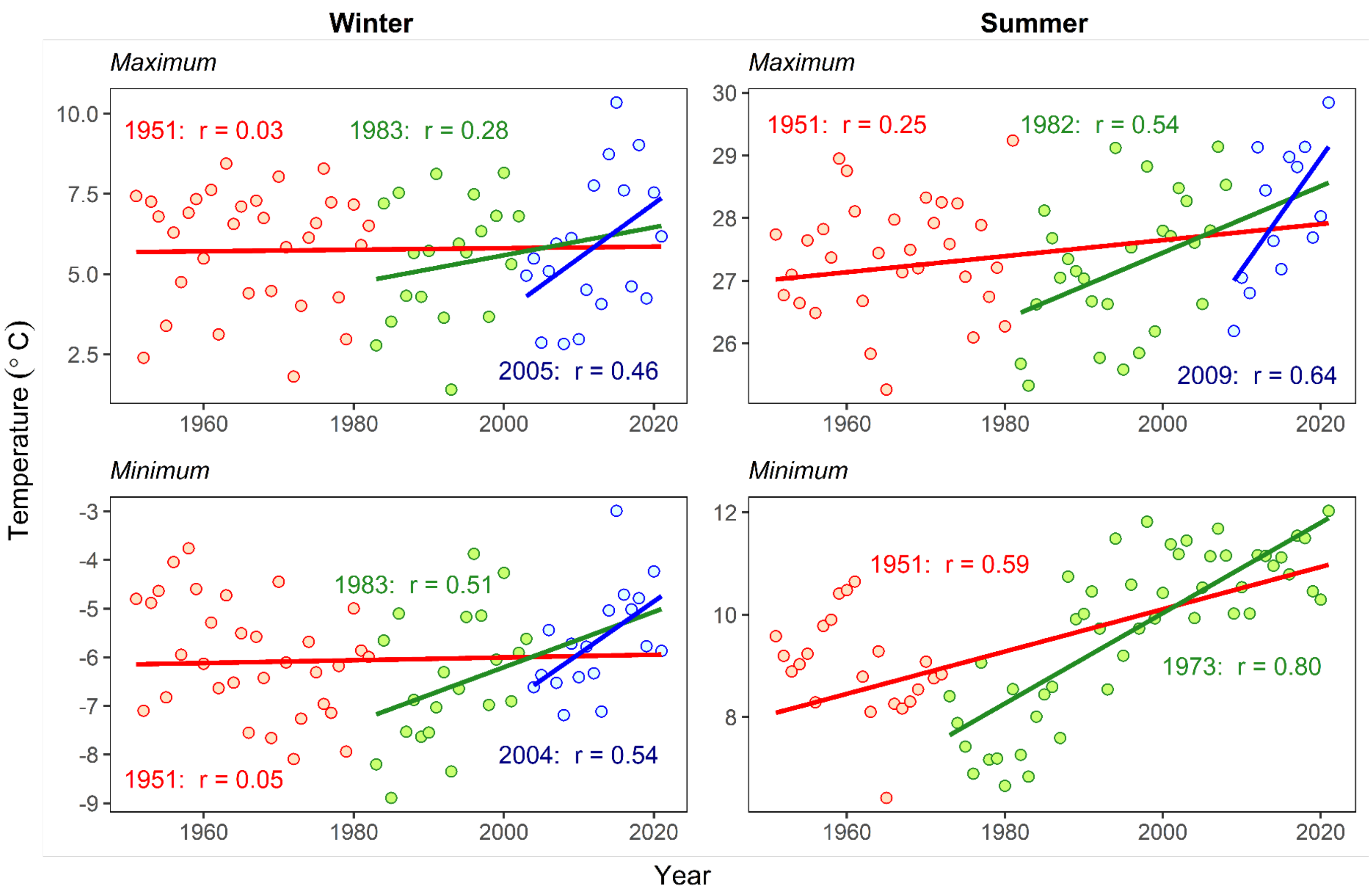


Figure 11. Correlation Coefficients between Seasonal Average Temperature and Different Sets of Years

Years on the x axis indicate a starting year of the set of years. For instance, in the case of 1990, a correlation coefficient was calculated based on years between 1990 and 2021.

Physical and Chemical

Mono Lake Surface Elevation

The average monthly surface elevation of Mono Lake in January 2021 was 6380.8 feet or 1.5 feet lower than the January lake elevation in 2019 (Figure 12). Water Year 2020-21 produced 45,585 acre-feet of runoff in Mono Basin, 38% of the long-term average and ranked 86th since 1935. Input from the two major tributaries (Rush and Lee Vining Creeks) was 52,204 acre-feet, or 52% of the long-term average since re-watering in 1990. The lake level dropped 1.3 feet to 6379.5 feet by December 2021. The input of 52% of normal was insufficient to maintain the lake level at 6380 feet.

Transparency

Average lake-wide transparency remained below 1 m except for a single station read in May at Station 5 throughout 2021, and the maximum single station reading was 1.0 m (Table 3-6, Figure 13). Transparency of Mono Lake during the summer improved from 0.40 m in May to only 0.82 m in June, even though *Artemia* grazing reduced midsummer phytoplankton. A year-round lake-wide transparency below 1 m was last observed in 2015 and 2016, the last two years of the driest five-year period on record, and again in 2020 (Figure 14). Beginning in 2014, annual lake-wide mean transparency has progressively worsened each year; 1.5 m in 2014, 0.9 m in 2015 and 0.6 m in 2016; however, this trend was finally reversed in 2017 even though it still lagged behind historical values. In 2020, however, transparency degraded to the levels observed in 2015 and 2016 and remained so in 2021. In 2021, the input flow of Rush and Lee Vining Creeks combined peaked on June 3 with estimated combined flow of 218 cfs, which corresponded to an approximate 0.9 exceeding probability and 1.1-year recurrence interval based on daily flow data available since 1990. A peak inflow below 220 cfs has been observed only six times since 1990: 1990, 2007, and 2012 to 2015. The influx of freshwater combined with lake stratification helped transparency to improve considerably in 2017. Beginning in 2008, annual peak transparency started to deteriorate with an average decline of 0.5m per year (Figure 15). Maximum transparency was 10.9m in 2007 decreased to 4.9m in 2008 and to 0.63m in 2016. Currently the maximum transparency is 0.81m.

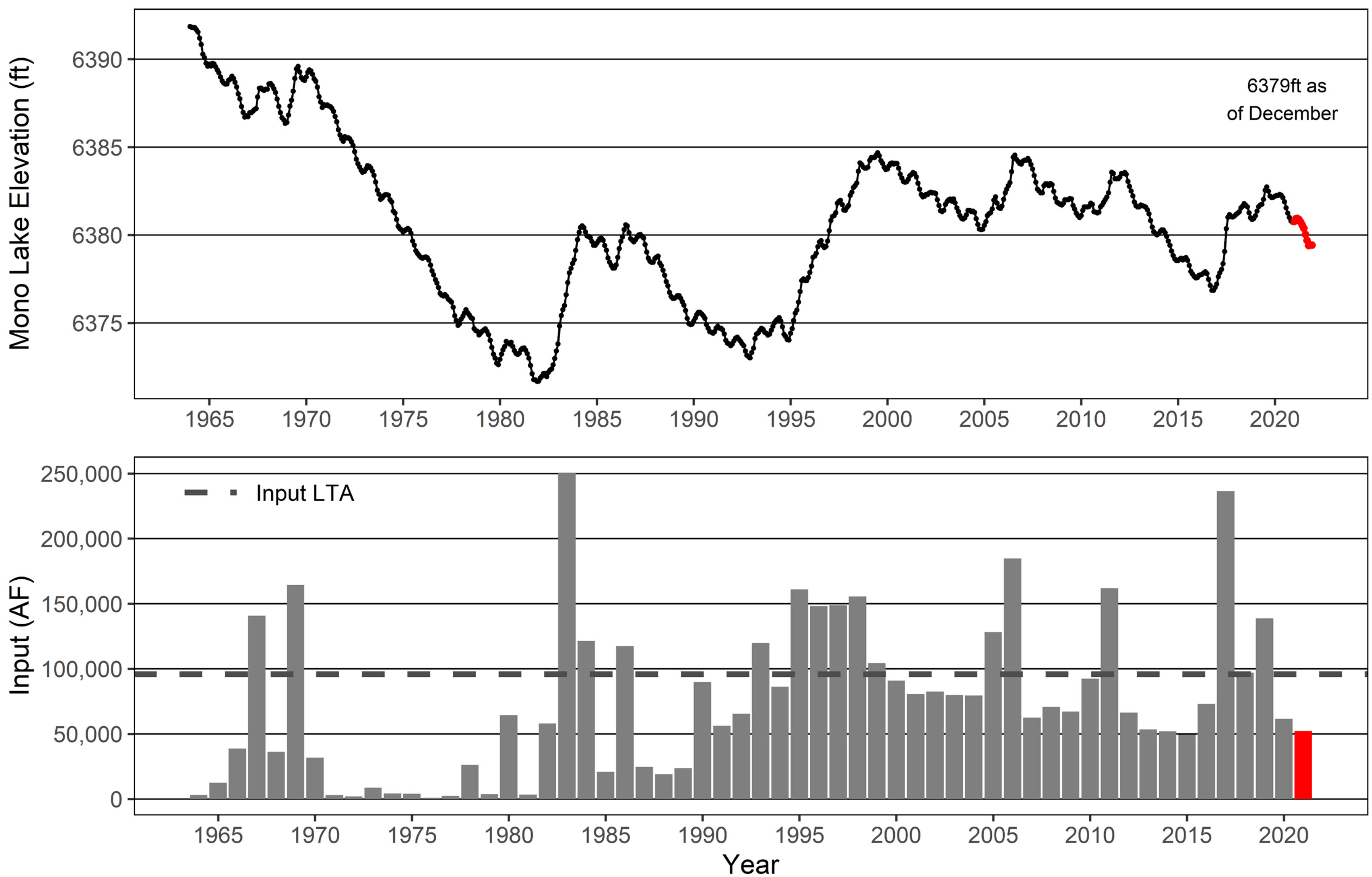


Figure 12. Mono Lake Surface Elevation (top) and Combined Inflow of Rush and Lee Vining Creeks (bottom)

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 2021.

Table 3.5. Secchi Depths (m) between February and December in 2021

	Sampling Month										
Station	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Western Sector											
1		0.5	0.25	0.4	0.7	0.6	0.5	0.5	0.6	0.5	
2		0.5	0.4	0.4	0.6	0.6	0.6	0.6	0.6	0.5	
3		0.4	0.5	0.6	0.8	0.6	0.9	0.5	0.6	0.5	
4		0.5	0.5	0.3	0.95	0.6	0.5	0.5	0.6	0.6	
5		0.3	0.3	0.4	1	0.6	0.6	0.6	0.6	0.4	
6		0.4	0.3	0.4	0.9	0.7	0.65	0.5	0.5	0.5	
AVG		0.43	0.38	0.42	0.83	0.62	0.63	0.53	0.58	0.50	
SE		0.08	0.11	0.10	0.15	0.04	0.15	0.05	0.04	0.06	
Eastern Sector											
7		0.4	0.3	0.3	0.7	0.7	0.6	0.6	0.5	0.5	
8		0.3	0.6	0.45	0.95	0.6	0.6	0.6	0.5	0.4	
9		0.4	0.5	0.4	0.95	0.5	0.7	0.6	0.7	0.5	
10		0.4	0.3	0.4	0.6	0.6	0.6	0.5	0.6	0.5	
11		0.4	0.5	0.4	0.95	0.8	0.6	0.6	0.6	0.4	
12		0.45	0.3	0.3	0.7	0.6	0.5	0.5	0.5	0.5	
AVG		0.39	0.42	0.38	0.81	0.63	0.60	0.57	0.57	0.47	
SE		0.05	0.13	0.06	0.16	0.10	0.06	0.05	0.08	0.05	
Total Lakewide											
AVG		0.41	0.40	0.40	0.82	0.63	0.61	0.55	0.58	0.48	
SE		0.41	0.40	0.40	0.82	0.63	0.61	0.55	0.58	0.48	

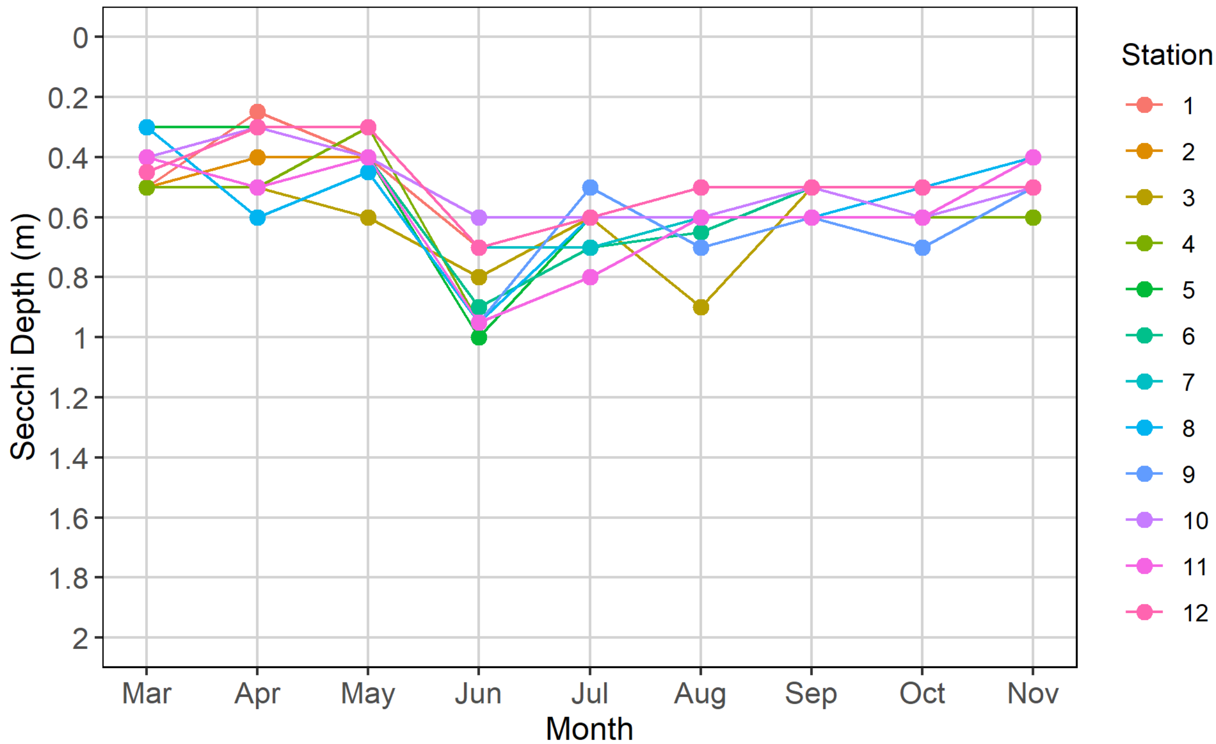


Figure 13. Lake-wide Secchi Depths in 2021 by Station

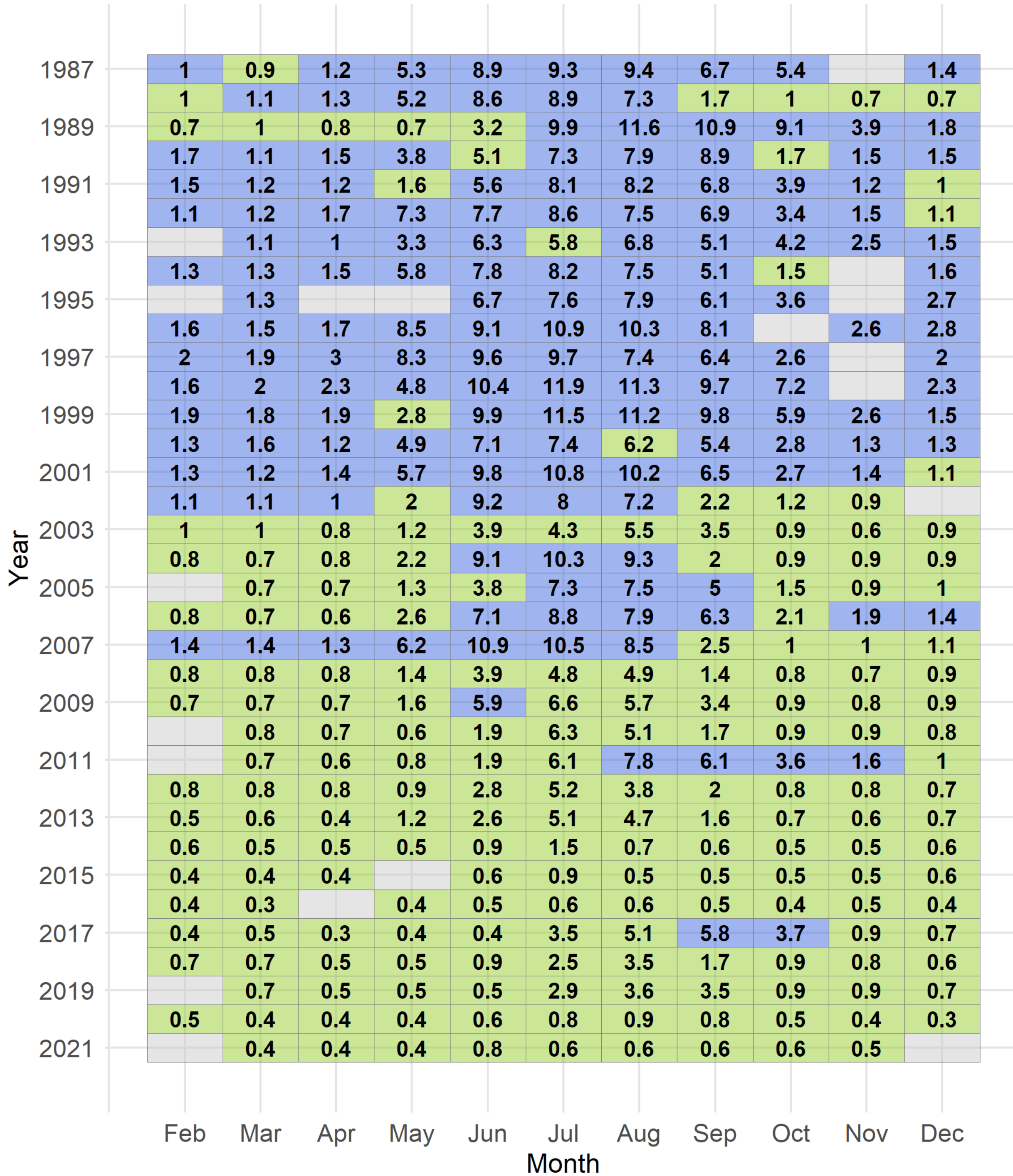


Figure 14. Long-term Lake-wide Average Secchi Depths (m)

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.

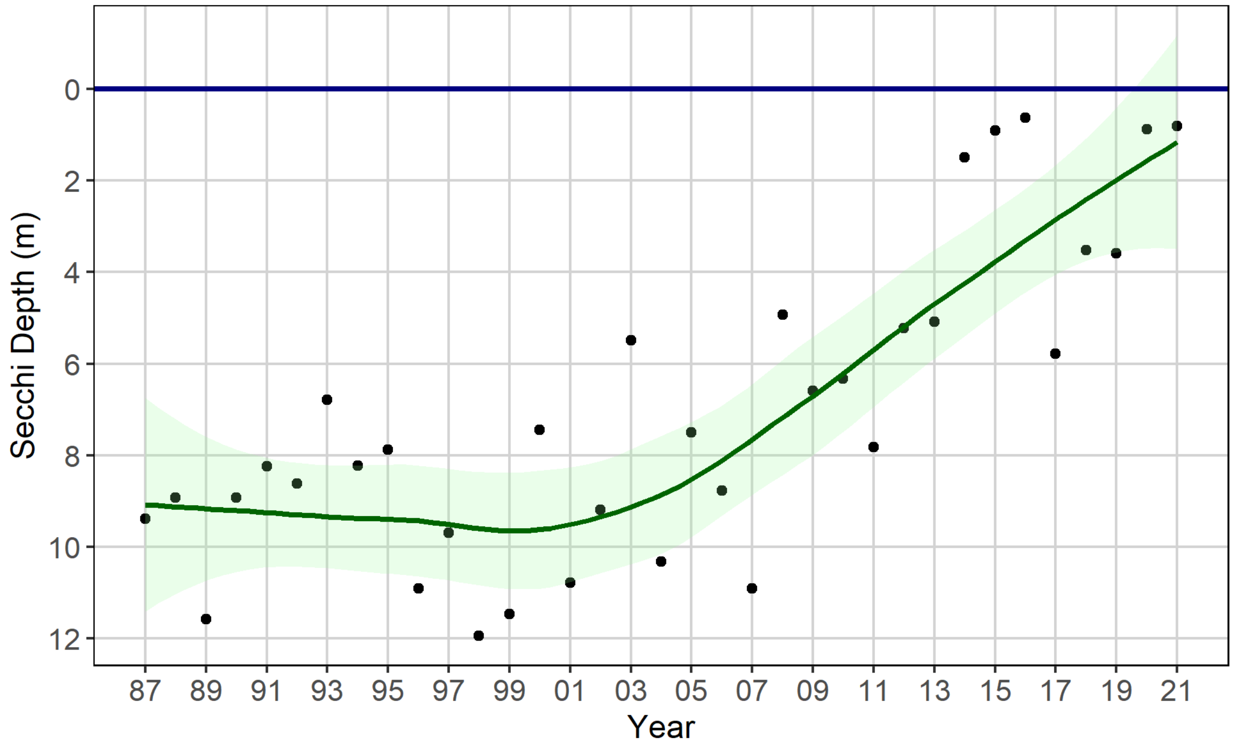


Figure 15. Trend in Annual Peak Lake-wide Secchi Depth Readings (m)

Water Temperature

Mono Lake started the year with isothermal conditions in 2021 as there was only 0.3°C of thermal gradient in March (Table 3-7, Figure 16). The thermocline started to develop between 5m and 6m of depth in May and migrated downward to between 9m and 10m by August. No data were available for the rest of the year; however, it is likely that Mono Lake has become isothermal again at the end of 2021.

Average water temperature in the epilimnion (<=10m) remained mostly below normal throughout 2021 except in April while average water temperature in the hypolimnion was found mostly above normal (Figure 17, Figure 18). Water temperature in the epilimnion remained below average in summer in spite of unusually warm summer. It is possible that higher than normal summer precipitation helped offset a rise in water temperature. Water temperature in the hypolimnion during monomictic period tends to be lower in spring and winter, and higher in summer and fall. Higher than normal hypolimnetic might have been following this trend.

Conductivity

Epilimnetic specific conductivity began to decrease in March, the first month of monitoring in 2021, and there was very little to no difference along the water column and across monitoring months (Table 3-8, Figure 19). Due to lower influx of freshwater, a gradient in conductivity did not develop in summer.

Table 3.6. Water Temperature (°C) Depth Profile at Station 6 in 2021

Depth	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
0	-	3.2	9.7	15.2	18.6	21.1	19.9	-	-	-	-
1	-	3.2	9.6	14.8	18.2	20.9	19.9	-	-	-	-
2	-	3.2	9.2	14.3	17.6	20.9	20	-	-	-	-
3	-	3.2	9	13.7	16.5	20.8	20	-	-	-	-
4	-	3.2	9	13.1	14.4	20.8	20	-	-	-	-
5	-	3.2	9	11.7	14	20.8	20	-	-	-	-
6	-	3.2	9	8.6	13.7	20.7	20	-	-	-	-
7	-	3.2	8.9	8	13.3	19.2	20	-	-	-	-
8	-	3.2	7.6	7	12.9	13.7	20	-	-	-	-
9	-	3.2	6.7	6.6	12.3	12.2	20	-	-	-	-
10	-	3.2	6.4	6.2	10	11	13.4	-	-	-	-
11	-	3.2	6.2	6	8.4	10	10.9	-	-	-	-
12	-	3.2	6	5.8	8	8.9	9.6	-	-	-	-
13	-	3.2	5.6	5.6	7.6	8.5	8.6	-	-	-	-
14	-	3.2	5.4	5.4	7.1	8.1	8.3	-	-	-	-
15	-	3.2	5.3	5.2	6.2	7.6	8.2	-	-	-	-
16	-	3.2	5.1	5.1	5.9	7.1	8.2	-	-	-	-
17	-	3.2	4.8	4.9	5.8	6.8	7.8	-	-	-	-
18	-	3.2	4.6	4.9	5.6	6.7	7.2	-	-	-	-
19	-	3.2	4.4	4.7	5.5	6.5	7.1	-	-	-	-
20	-	3.2	4.3	4.6	5.4	6.2	6.7	-	-	-	-
21	-	3.2	4.2	4.5	5.3	6.1	6.6	-	-	-	-
22	-	3.2	4.2	4.4	5.3	6	6.5	-	-	-	-
23	-	3.1	4.1	4.4	5.2	5.9	6.4	-	-	-	-
24	-	3	4	4.3	5.1	5.8	6.3	-	-	-	-
25	-	3	3.9	4.3	5.1	5.7	6.1	-	-	-	-
26	-	2.9	3.9	4.2	5	5.6	6	-	-	-	-
27	-	2.9	3.8	4.1	4.9	5.5	5.9	-	-	-	-
28	-	2.9	3.8	4	4.8	5.4	5.8	-	-	-	-
29	-	2.9	3.7	4	4.8	5.3	5.8	-	-	-	-
30	-	2.9	3.7	4	4.7	5.3	5.7	-	-	-	-
31	-	2.8	3.7	4	4.6	5.2	5.6	-	-	-	-
32	-	2.8	3.6	3.9	4.5	5.1	5.5	-	-	-	-
33	-	2.8	3.6	3.9	4.5	5	5.5	-	-	-	-
34	-	2.8	3.6	3.9	4.5	4.9	5.4	-	-	-	-
35	-	2.8	3.6	3.8	4.4	4.9	5.4	-	-	-	-
36	-	2.8	3.5	3.8	4.4	4.9	5.3	-	-	-	-
37	-	2.8	3.5	3.8	4.4	4.8	5.3	-	-	-	-
38	-	2.8	3.5	3.8	4.4	4.8	5.3	-	-	-	-
39	-	2.8	3.5	3.8	-	-	-	-	-	-	-

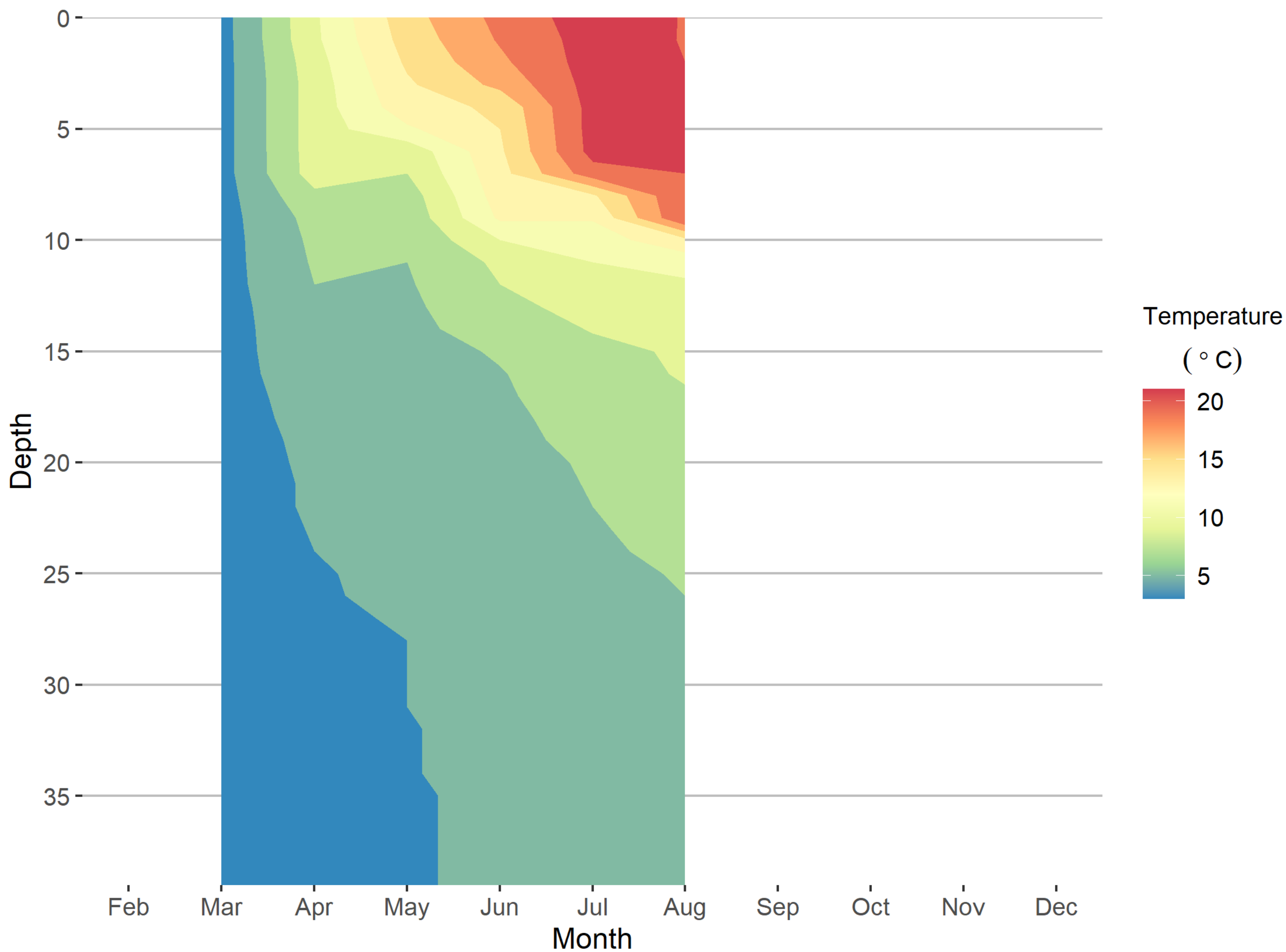


Figure 16. Water Temperature (°C) Depth Profile at Station 6 in 2021

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

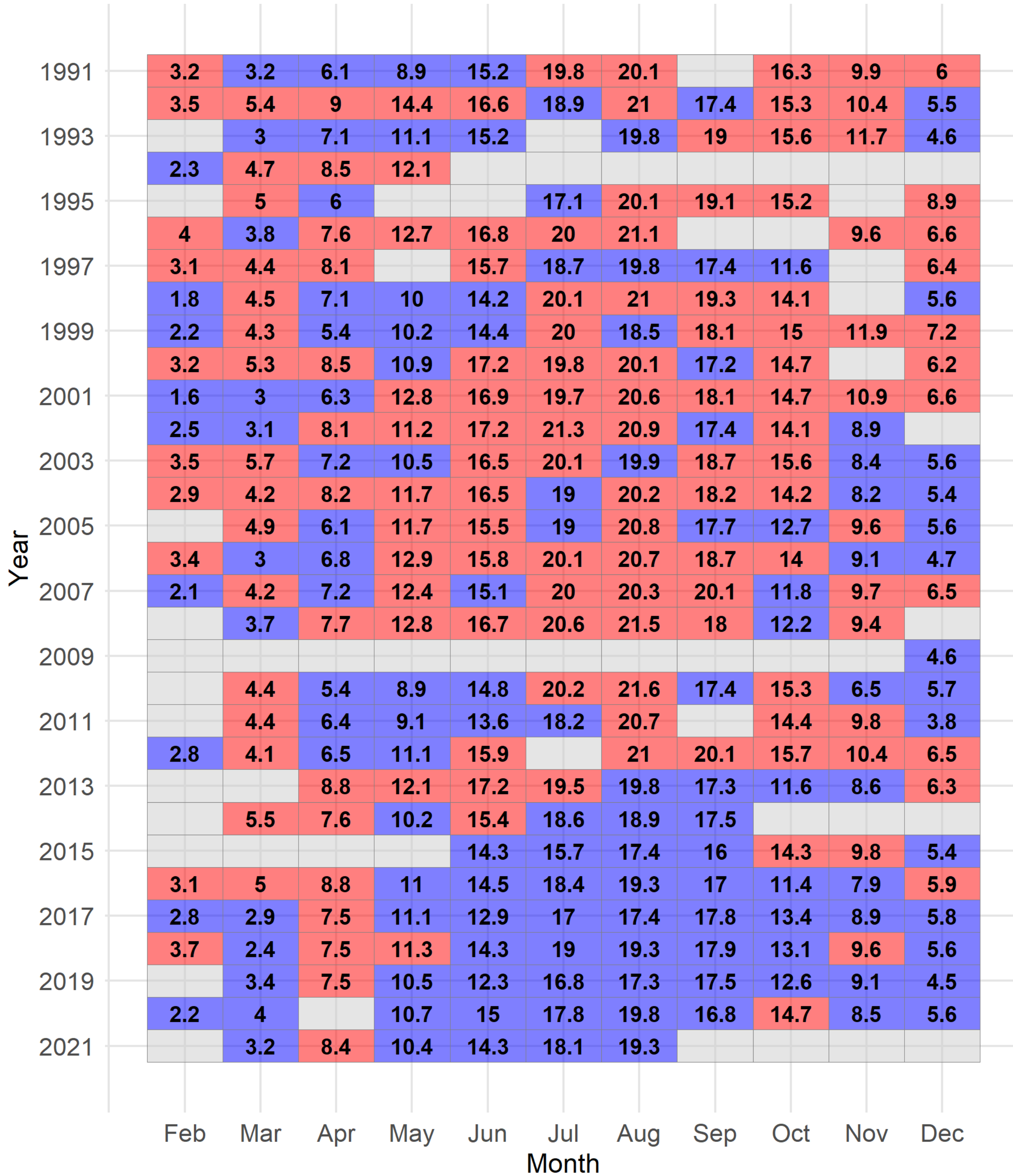


Figure 17. Average Water Temperature (°C) between 1 and 10 m at Station 6

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.

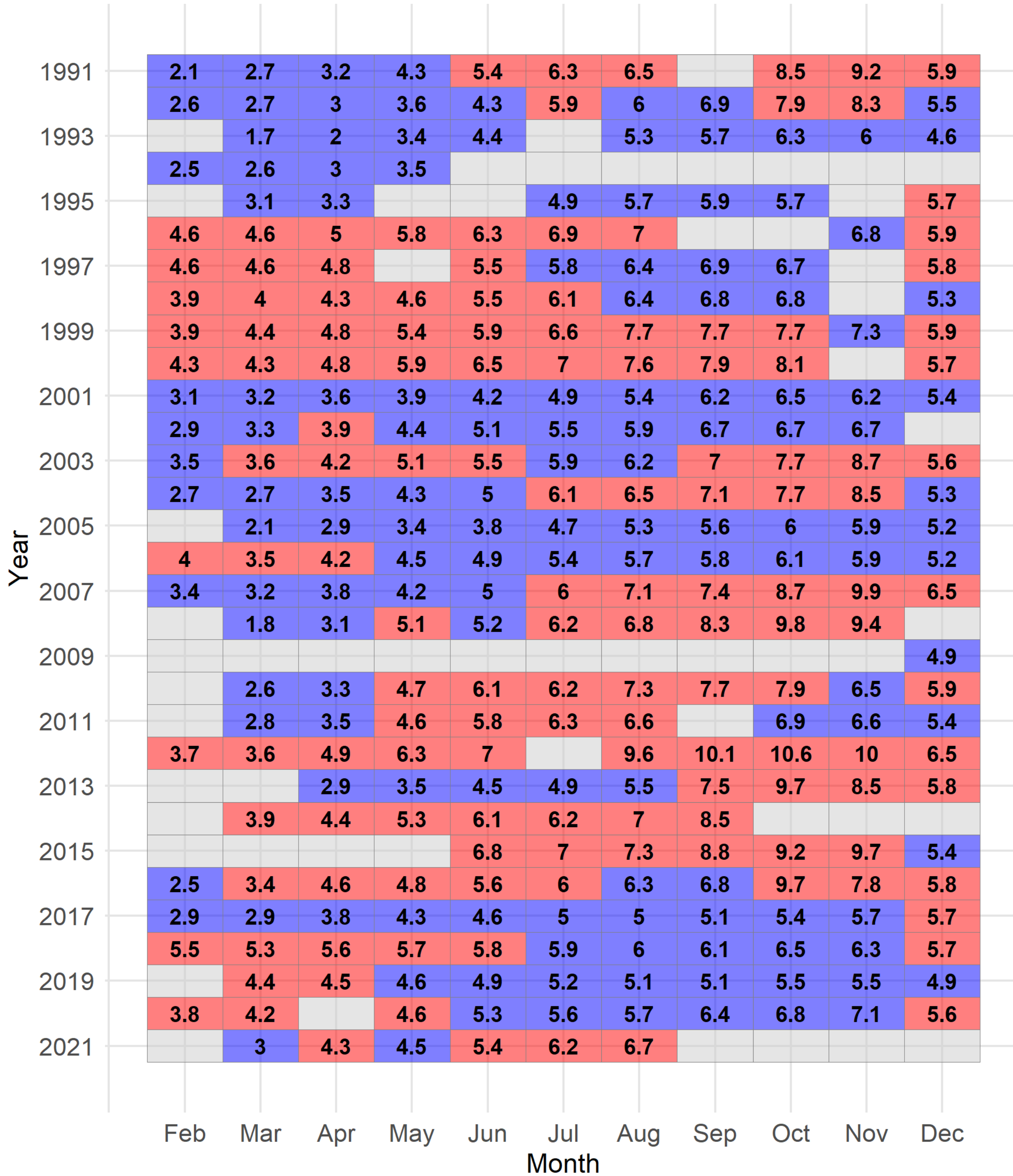


Figure 18. Average Water Temperature (°C) between 11 and 38 m at Station 6

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 3.7. Conductivity (mS/cm at 25°C) Depth Profile at Station 6 in 2021

Depth	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
0	-	82.3	83.3	82.5	83.1	83.3	83.5	-	-	-	-
1	-	82.3	83.3	82.7	83	83.4	83.8	-	-	-	-
2	-	82.3	82.8	82.9	83.1	83.4	83.9	-	-	-	-
3	-	82.3	82.7	83	82.3	83.4	83.9	-	-	-	-
4	-	82.3	82.7	83.3	82.5	83.4	83.9	-	-	-	-
5	-	82.3	82.6	84	82.8	83.4	83.9	-	-	-	-
6	-	82.3	82.6	82.4	82.6	83.2	84	-	-	-	-
7	-	82.3	82.6	82.1	82.6	80.4	83.9	-	-	-	-
8	-	82.3	82.1	82.3	82.8	82.2	83.9	-	-	-	-
9	-	82.3	82.1	82.4	82.4	82.2	83.9	-	-	-	-
10	-	82.3	82.4	82.4	80.8	82.5	82.4	-	-	-	-
11	-	82.3	82.4	82.5	82.2	82.3	82.7	-	-	-	-
12	-	82.3	82.2	82.3	82.5	82.4	82.9	-	-	-	-
13	-	82.4	82.3	82.4	82.4	82.5	82.9	-	-	-	-
14	-	82.4	82.4	82.4	82.1	82.5	82.8	-	-	-	-
15	-	82.4	82.4	82.4	82.4	82.3	82.8	-	-	-	-
16	-	82.4	82.3	82.4	82.5	82.3	82.7	-	-	-	-
17	-	82.4	82.3	82.4	82.5	82.5	82.3	-	-	-	-
18	-	82.4	82.3	82.4	82.5	82.4	82.7	-	-	-	-
19	-	82.4	82.5	82.4	82.5	82.4	82.5	-	-	-	-
20	-	82.4	82.4	82.4	82.4	82.4	82.5	-	-	-	-
21	-	82.4	82.4	82.4	82.5	82.5	82.6	-	-	-	-
22	-	82.4	82.4	82.4	82.5	82.4	82.5	-	-	-	-
23	-	82.4	82.4	82.4	82.5	82.5	82.5	-	-	-	-
24	-	82.4	82.4	82.4	82.5	82.4	82.5	-	-	-	-
25	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
26	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
27	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
28	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
29	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
30	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
31	-	82.5	82.4	82.4	82.4	82.4	82.4	-	-	-	-
32	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
33	-	82.5	82.4	82.4	82.5	82.4	82.4	-	-	-	-
34	-	82.5	82.4	82.4	82.4	82.4	82.5	-	-	-	-
35	-	82.5	82.4	82.4	82.4	82.4	82.4	-	-	-	-
36	-	82.5	82.4	82.4	82.5	82.4	82.5	-	-	-	-
37	-	82.5	82.4	82.4	82.5	82.4	82.5	-	-	-	-
38	-	82.5	82.4	82.4	82.5	82.4	82.5	-	-	-	-
39	-	82.5	82.4	82.4	-	-	-	-	-	-	-

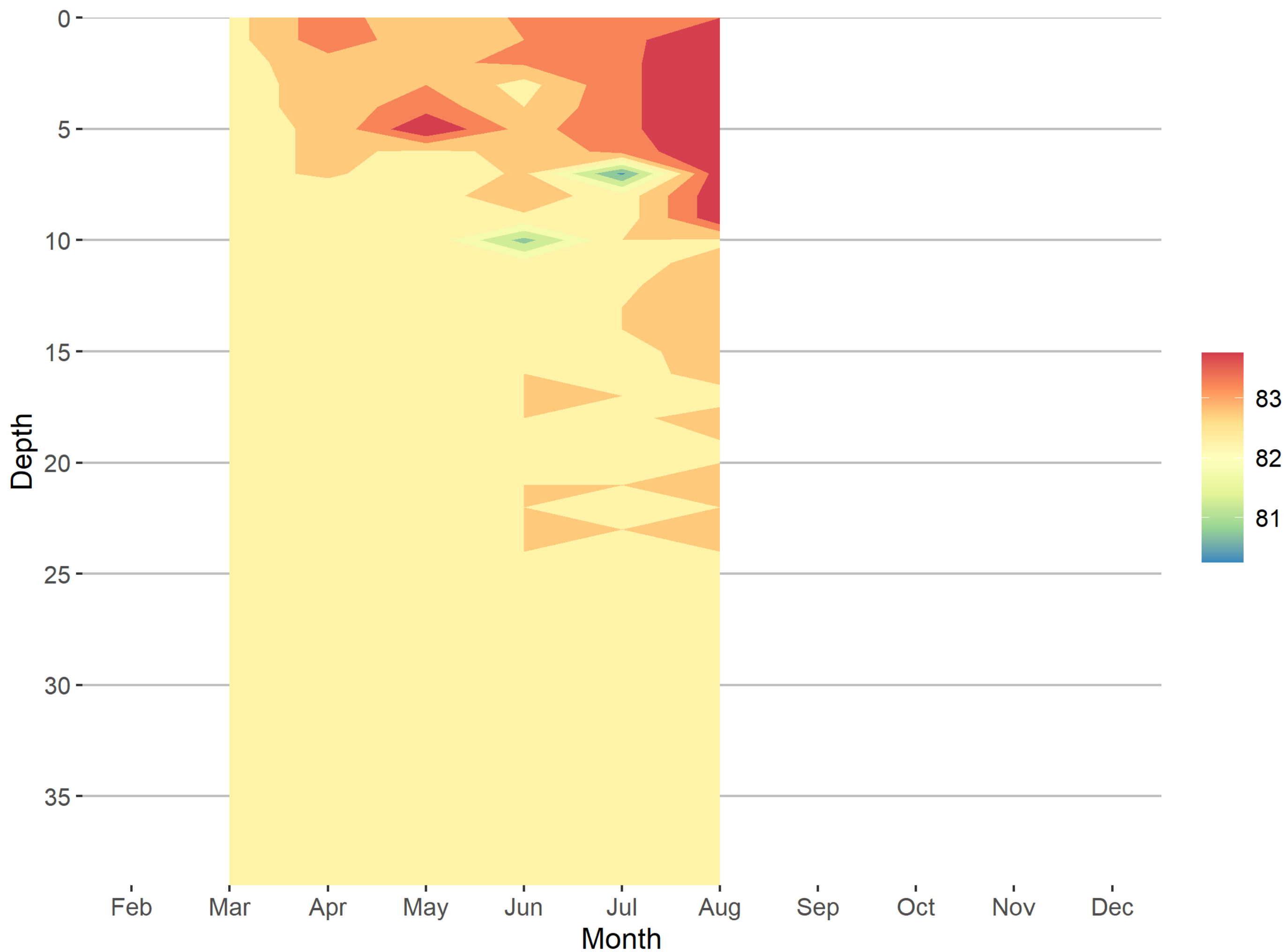


Figure 19. Conductivity (mS/cm) Depth Profile at Station 6 in 2021

Salinity

Salinity expressed in g/L at two different depth classes (between 1 and 10 m and below 10 m) is presented in Figure 20 and Figure 21. Salinity in the epilimnion started at above 81.5 g/L in March, and increased steadily to 85.5 g/L in August. The salinity values between May and August were higher than the previous three years during which Mono Lake became and remained stratified, and also higher than the long-term average of the respective months. Because of 52% of inflow from Rush and Lee Vining Creeks in 2021, it is expected that the salinity in the epilimnion would have increased steadily throughout the rest of the year, although these data are not available. This in turn, would have resulted in higher hypolimnetic salinity at the end of 2021 due to holomixis. Salinity in both the epilimnion and hypolimnion are on the rise after the latest meromixis when epilimnetic and hypolimnetic salinities reached their respective lows in 2019 and 2020. Epilimnetic salinity was higher than the long-term average of the respective month between May and August while hypolimnetic salinity remained below the long-term average throughout 2021.

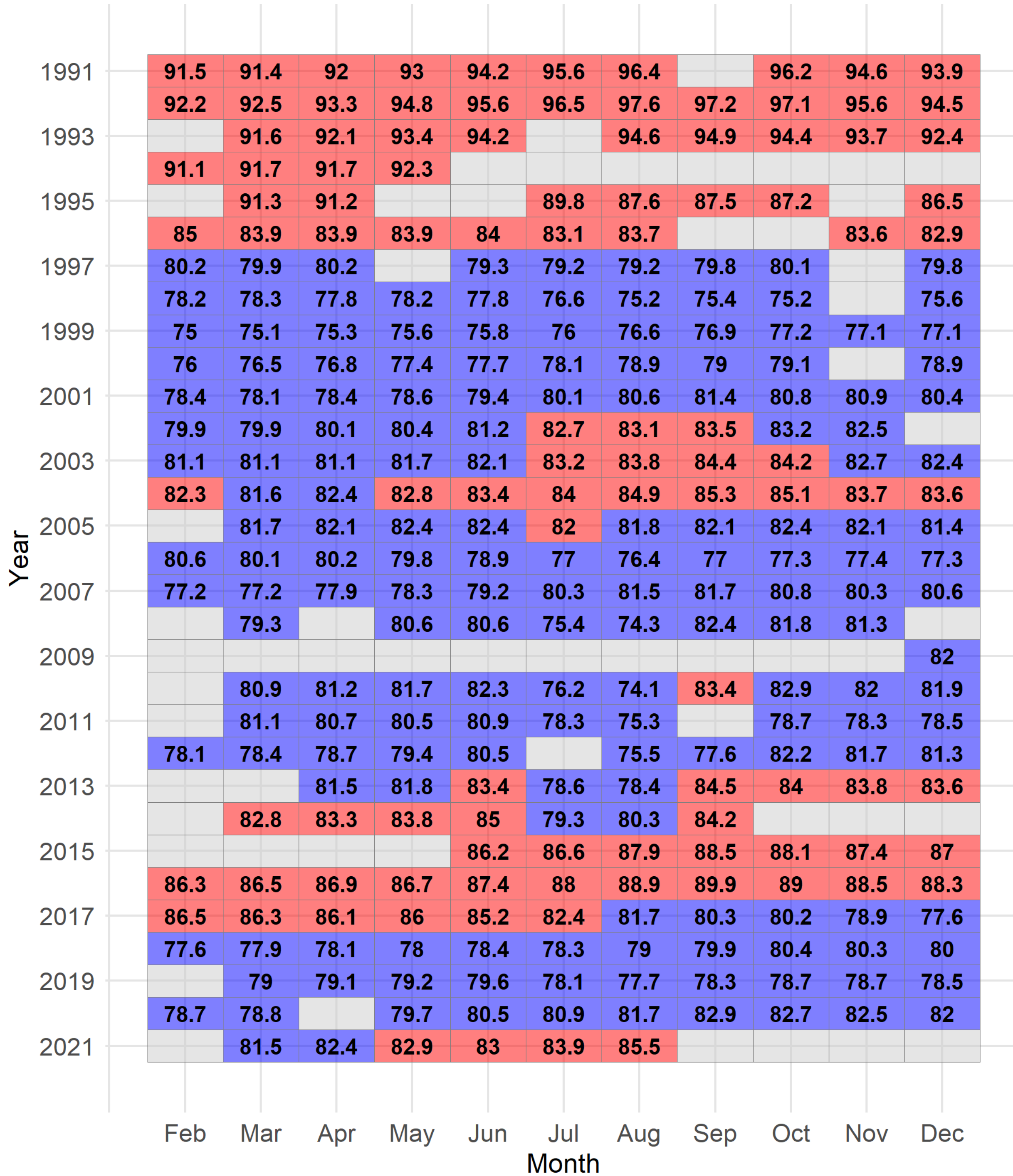


Figure 20. Average Salinity (g/L) between 1 and 10 m at Station 6

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.

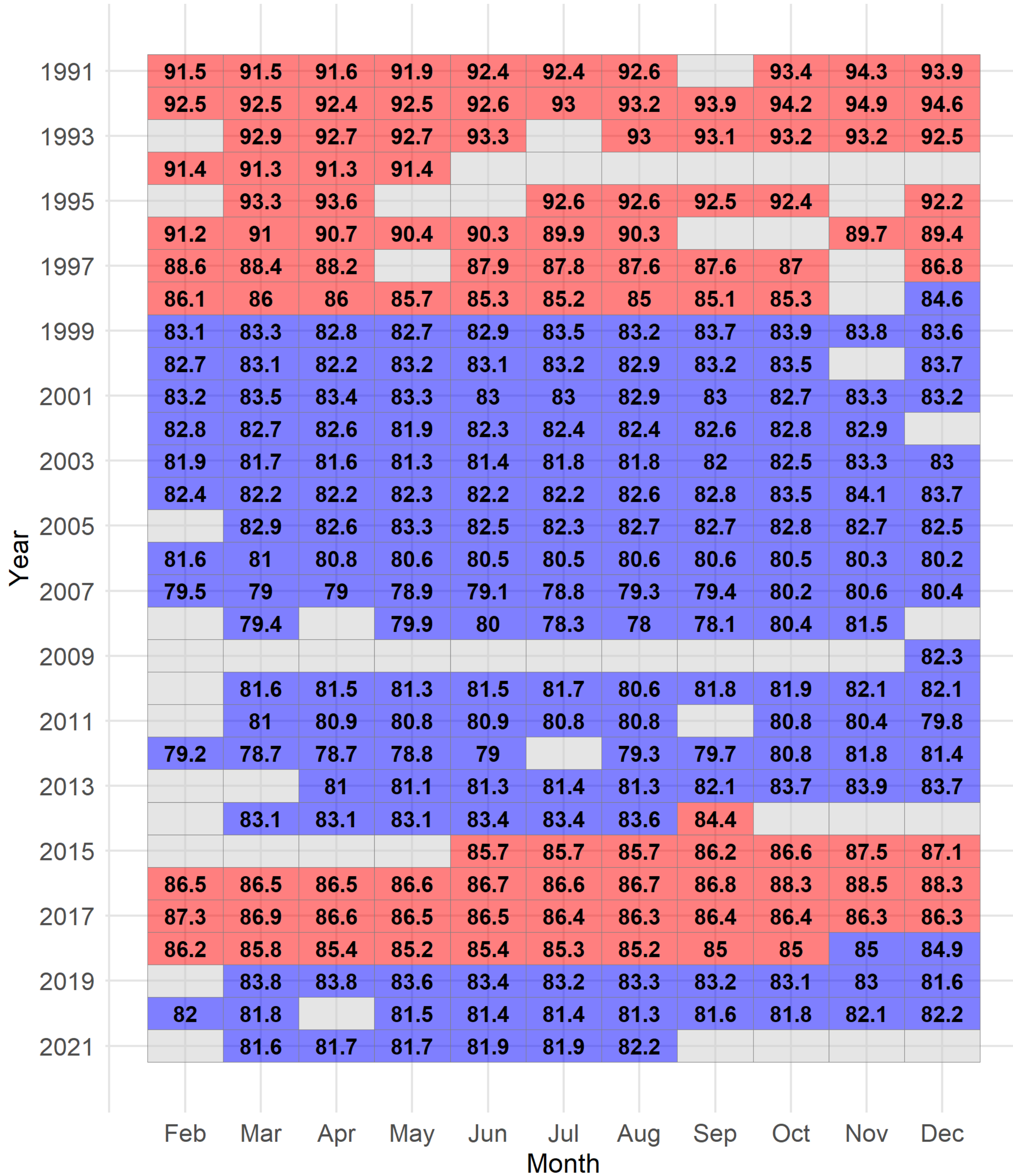


Figure 21. Average Salinity (g/L) between 11 and 38 m at Station 6

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.

Dissolved Oxygen

Dissolved oxygen (DO) levels in the upper mixed layer (< 15 m) started around 6.0 mg/L in March , peaked in April, and remained relatively stable around 5 mg/L throughout the year until October when the levels dipped under 1 mg/L before rebounding in November (Table 3-9, Figure 22). A steady decline of DO levels from spring through fall due to *Artemia* grazing pressure on phytoplankton populations is usually followed by DO level recovery in winter with disappearance of *Artemia*. In 2021, however, DO in the upper mixed layer was depleted in October potentially due to unusually warm October. DO in the lower mixing layer was depleted by June and started to recover in November. In October DO below 30m increased above 1 mg/L in spite of the water column above with being suboxic or anoxic. Average DO concentrations in the upper mixing layer in spring and early summer have been above LTA since the onset of the last meromixis in 2017 meanwhile average DO concentrations in the lower mixing layer remained mostly suboxic to anoxic and below LTA since the onset of the latest meromixis except 2021 (Figure 23, Figure 24).

Table 3.8. Dissolved Oxygen* (mg/L) Depth Profile at Station 6 in 2021

Depth	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
0	-	6.37	10.44	8.87	5.51	5.06	5.56	-	1.3	6.07	-
1	-	6.77	11.53	9.65	5.57	5.2	5.56	-	1.38	5.55	-
2	-	6.69	11.83	9.82	5.565	5.06	5.56	-	0.86	5.2	-
3	-	6.36	11.55	9.62	5.56	4.78	5	-	0.64	4.96	-
4	-	6.28	11.12	8.77	5.71	4.46	4.83	-	0.53	4.97	-
5	-	6.25	10.92	7.85	6.19	4.34	4.73	-	0.47	4.77	-
6	-	6.22	10.7	7.15	6.7	4.31	4.58	-	0.5	4.8	-
7	-	6.21	9.57	6.51	5.87	3.26	4.46	-	0.6	4.62	-
8	-	6.2	7.71	6.05	4.89	0.29	4.35	-	0.55	4.55	-
9	-	6.19	6.74	5.46	4.63	0.19	4.3	-	0.56	4.42	-
10	-	6.19	6.32	4.69	4.69	0.14	4.29	-	0.56	3.96	-
11	-	6.19	5.27	4.24	4.64	0.13	4.31	-	0.57	3.66	-
12	-	6.18	5.04	3.74	3.11	0.12	3.93	-	0.49	3.55	-
13	-	6.18	4.67	3.44	2.71	0.12	2.8	-	0.6	3.37	-
14	-	6.17	4.42	3.18	1.47	0.11	1.61	-	0.61	3.32	-
15	-	6.17	4.46	2.92	1.51	0.11	1.27	-	0.56	3.14	-
16	-	6.17	4.42	2.49	1.13	0.11	1.14	-	0.66	3.11	-
17	-	6.15	4.32	2.57	0.21	0.1	1.07	-	0.65	3.24	-
18	-	6.15	4.2	2.37	0.15	0.1	1.06	-	0.66	3.14	-
19	-	6.13	4.08	2.23	0.15	0.1	1.03	-	0.66	3.08	-
20	-	6.11	3.92	2.06	0.13	0.1	1.01	-	0.64	3.21	-
21	-	6.02	3.63	2.04	0.12	0.1	0.92	-	0.62	2.94	-
22	-	5.47	3.62	1.63	0.12	0.1	0.88	-	0.57	3.04	-
23	-	4.38	3.04	1.56	0.12	0.09	0.84	-	0.62	3.06	-
24	-	3.38	2.85	1.51	0.11	0.09	0.82	-	0.66	3.05	-
25	-	3.35	2.92	1.43	0.11	0.09	0.79	-	0.46	3.16	-
26	-	3.33	2.72	1.38	0.11	0.09	0.77	-	0.29	3.12	-
27	-	3.36	2.65	1.27	0.11	0.09	0.75	-	0	3.04	-
28	-	3.22	2.59	1.13	0.11	0.09	0.73	-	0	2.6	-
29	-	3.12	2.57	1.08	0.11	0.09	0.72	-	0	2.5	-
30	-	2.93	2.53	0.75	0.11	0.09	0.71	-	1.2	1.88	-
31	-	2.57	1.93	0.66	0.11	0.09	0.7	-	1.33	1.5	-
32	-	2.34	1.48	0.59	0.11	0.09	0.69	-	1.42	1.29	-
33	-	2.09	1.26	0.6	0.1	0.09	0.68	-	1.34	1.02	-
34	-	2.02	1.11	0.61	0.11	0.09	0.67	-	1.33	0.99	-
35	-	1.91	-	0.56	0.1	0.09	0.66	-	1.3	1.5	-
36	-	1.86	-	0.3	0.1	-	0.66	-	1.39	1.69	-
37	-	-	-	0.28	0.1	-	0.65	-	1.44	1.09	-
38	-	-	-	0.28	0.1	-	0.64	-	1.36	0.77	-
39	-	-	-	0.23	0.1	-	-	-	-	-	-
40	-	-	-	0.17	0.09	-	-	-	-	-	-

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

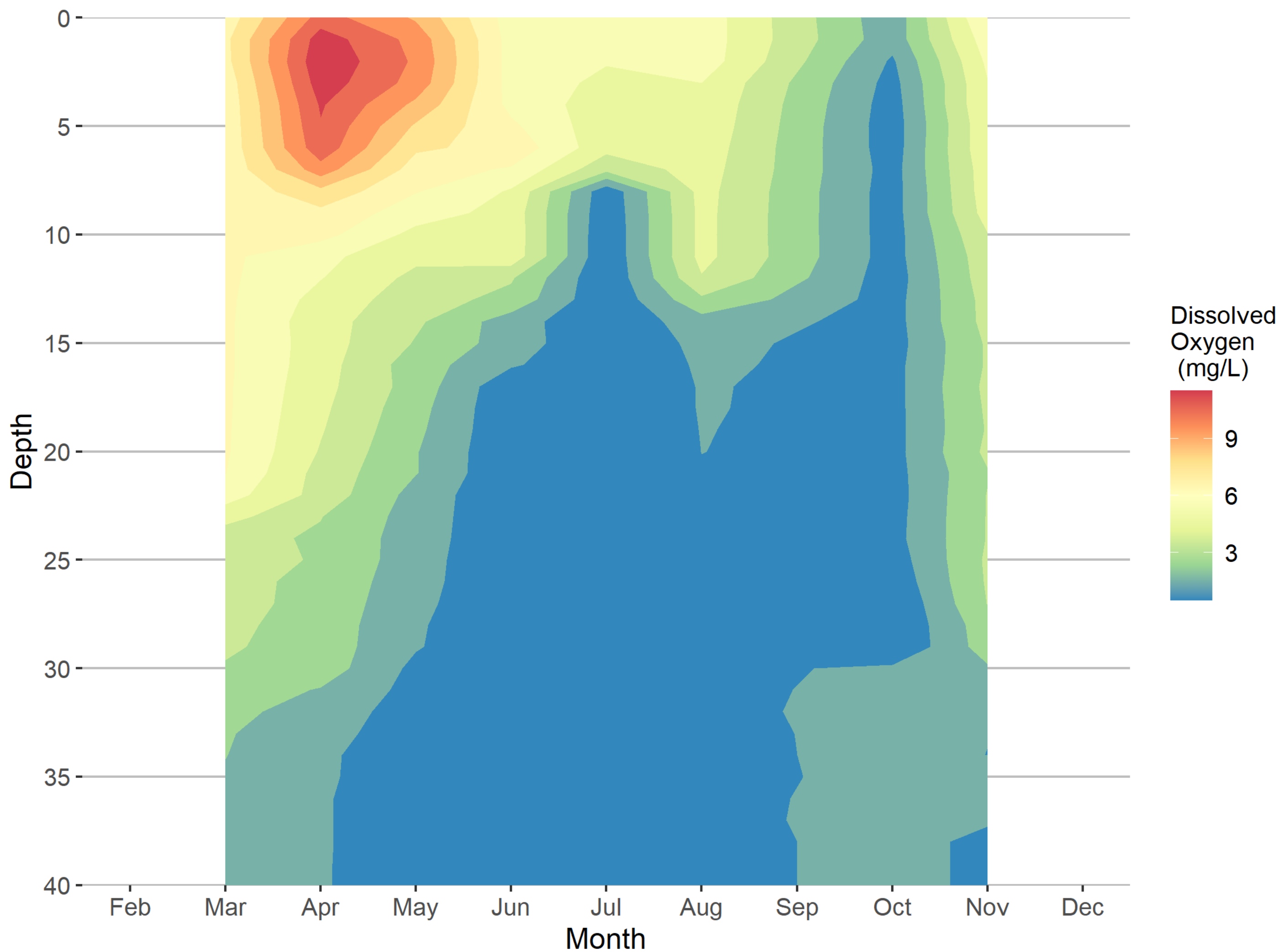


Figure 22. Dissolved Oxygen (mg/L) Depth Profiles at Station 6 in 2021

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

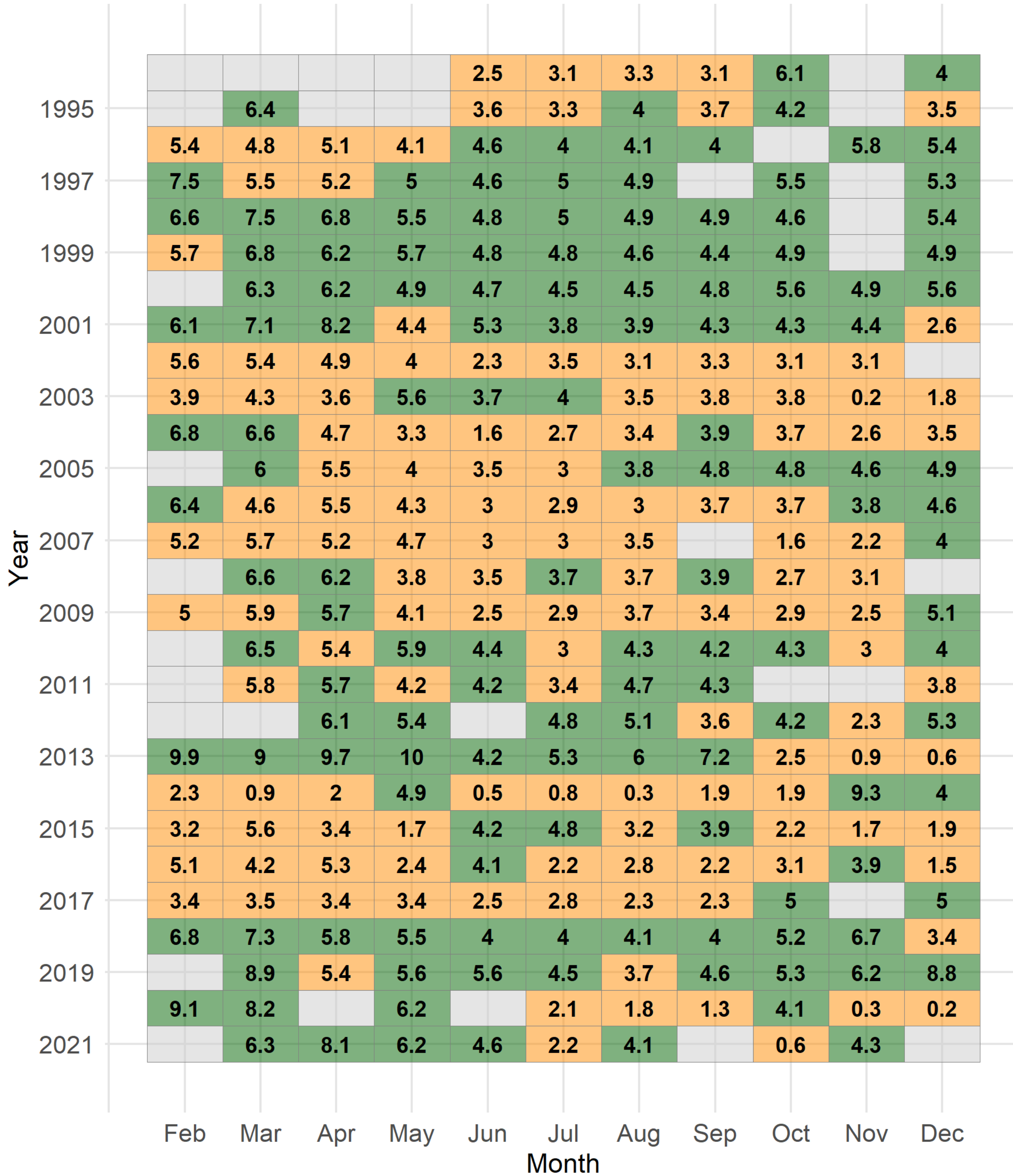


Figure 23. Average Dissolved Oxygen (mg/L) at Station 6 between 1 and 15 m

Orange-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.

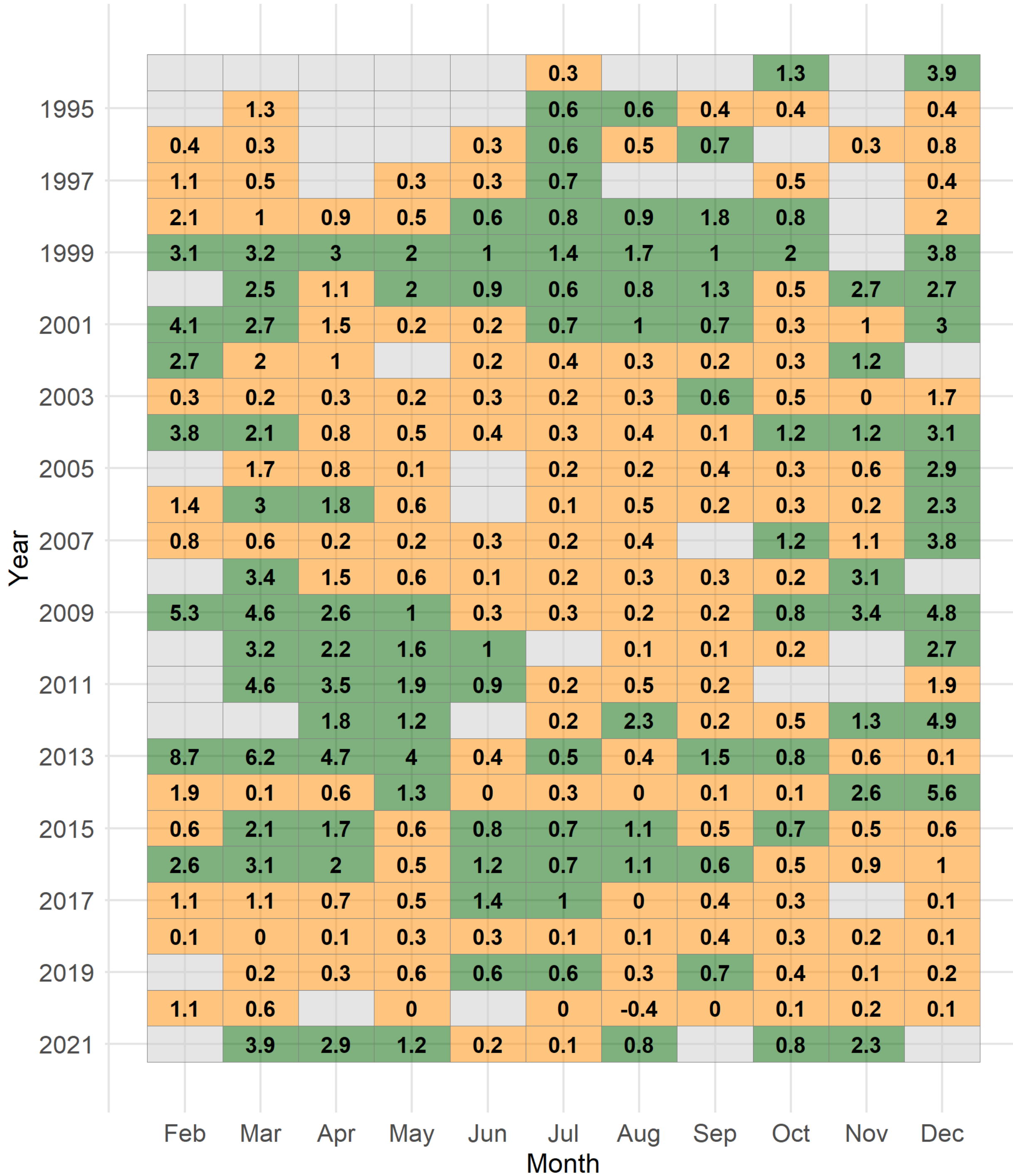


Figure 24. Average Dissolved Oxygen (mg/L) at Station 6 between 16 and 38 m

Orange-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.

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 always consider the standard error associated with *Artemia* counts when making inferences from the data.

Artemia Population

Hatching of overwintering cysts accelerated considerably in March as naupliar instar abundance peaked in March (70,022 +/- 33,764 m^{-2}) with the eastern sector showing much higher abundance (132,413 +/- 24,778 m^{-2}) compared to that of western section (7,631 413 +/- 2,065 m^{-2}), where the peak was observed in April (35,573 +/- 15,374 m^{-2}). All naupliar instars in mid-March were instars 1 and 2. Adult *Artemia* started to appear in April and recorded the monthly peak in the mid-June at 50,731 +/- 7,535 m^{-2}.

Fecund females were first recorded in June. Oviparous females peaked at 13,307 +/- 2,208 m^{-2} in June and declined continuously afterward. Ovoviviparous females were most abundant in June at 376 +/- 293 m^{-2}, declining sharply in July, then showing a slight increase again in September. By July, hatching and growth decreased significantly, with naupliar instars and juveniles comprising only 11% of the population as compared to 65% in May. A sharp adult *Artemia* peak was followed by continuous decline, indicating very small later generations resulting from depletion of food sources by the early summer generation.

Table 3.9. *Artemia* Lake-wide and Sector Population Means (per m^2 or m^{-2}) in 2021

	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	-	-	-	-	-	-	-	-	-	-
Mar	70,022	0	0	0	0	0	0	0	0	70,022
Apr	51,885	2,696	161	161	0	0	0	0	0	54,742
May	13,950	46,144	31,730	31,730	0	0	0	0	0	91,824
Jun	8,880	14,648	50,731	32,622	18,109	4,319	107	13,307	376	74,259
Jul	2,306	1,279	28,377	17,361	11,016	1,531	76	9,283	126	31,963
Aug	1,443	504	24,130	14,853	9,276	693	32	8,451	101	26,077
Sep	1,642	140	13,738	7,572	6,166	113	76	5,804	173	15,496
Oct	1,060	225	5,572	2,837	2,735	93	46	2,494	102	6,858
Nov	1,747	622	2,199	1,141	1,059	35	41	917	66	4,569
Dec	-	-	-	-	-	-	-	-	-	-
Western Sector										
Feb	-	-	-	-	-	-	-	-	-	-
Mar	7,631	0	0	0	0	0	0	0	0	7,631
Apr	35,573	1,905	0	0	0	0	0	0	0	37,478
May	6,331	22,857	14,795	14,795	0	0	0	0	0	43,984
Jun	8,317	17,921	61,596	40,134	21,462	4,829	161	15,882	590	87,834
Jul	1,210	1,323	20,330	12,515	7,814	1,122	0	6,541	151	22,863
Aug	1,462	630	25,031	15,049	9,982	681	0	9,175	126	27,123
Sep	1,405	47	10,587	6,535	4,052	76	63	3,768	145	12,024
Oct	419	98	1,667	838	829	60	22	712	35	2,184
Nov	539	195	1,141	520	621	25	9	545	41	1,875
Dec	-	-	-	-	-	-	-	-	-	-
Eastern Sector										
Feb	-	-	-	-	-	-	-	-	-	-
Mar	132,413	0	0	0	0	0	0	0	0	132,413
Apr	68,196	3,488	322	322	0	0	0	0	0	72,005
May	21,569	69,430	48,665	48,665	0	0	0	0	0	139,665
Jun	9,443	11,375	39,866	25,111	14,755	3,810	54	10,731	161	60,684
Jul	3,403	1,235	36,424	22,208	14,217	1,941	151	12,024	101	41,062
Aug	1,424	378	23,228	14,658	8,570	706	63	7,726	76	25,031
Sep	1,878	202	16,889	8,608	8,281	151	88	7,839	202	18,968
Oct	1,701	353	9,478	4,837	4,641	126	69	4,276	170	11,532
Nov	2,956	1,049	3,258	1,761	1,497	44	72	1,289	91	7,263
Dec	-	-	-	-	-	-	-	-	-	-

Table 3.10. Standard Errors (SE) of *Artemia* Sector Population Means (per m^2 or m^{-2}) from Table 3.9 in 2021

	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	-	-	-	-	-	-	-	-	-	-
Mar	24,778	-	-	-	-	-	-	-	-	24,778
Apr	10,884	741	116	116	0	0	0	0	0	11,509
May	4,252	13,519	9,786	9,786	0	0	0	0	0	26,569
Jun	1,411	2,003	7,535	4,953	2,712	815	61	2,208	293	8,842
Jul	1,066	140	4,165	2,543	1,912	353	54	1,567	55	4,633
Aug	269	114	2,902	1,484	1,517	239	25	1,308	39	3,092
Sep	221	32	1,544	767	868	28	28	826	49	1,688
Oct	242	57	1,640	817	838	20	25	777	31	1,867
Nov	516	195	700	376	328	12	24	296	18	1,364
Dec	-	-	-	-	-	-	-	-	-	-
Western Sector										
Feb	-	-	-	-	-	-	-	-	-	-
Mar	2,065	-	-	-	-	-	-	-	-	2,065
Apr	15,374	1,306	0	0	0	0	0	0	0	16,475
May	455	11,661	7,648	7,648	0	0	0	0	0	19,574
Jun	2,097	2,753	13,254	8,855	4,572	1,456	110	3,776	590	15,658
Jul	110	234	3,710	3,013	1,596	221	0	1,396	55	3,755
Aug	401	193	4,548	2,240	2,548	418	0	2,185	61	4,902
Sep	199	18	2,138	1,306	953	29	29	897	44	2,163
Oct	117	25	606	288	323	19	12	284	20	741
Nov	173	74	423	186	241	13	6	222	19	573
Dec	-	-	-	-	-	-	-	-	-	-
Eastern Sector										
Feb	-	-	-	-	-	-	-	-	-	-
Mar	33,764	-	-	-	-	-	-	-	-	33,764
Apr	13,359	677	220	220	0	0	0	0	0	13,864
May	7,491	21,242	15,753	15,753	0	0	0	0	0	42,515
Jun	2,059	2,407	5,186	2,640	2,638	837	54	2,132	110	5,012
Jul	2,124	176	6,054	3,161	3,072	656	103	2,417	101	6,871
Aug	396	116	4,005	2,160	1,854	278	49	1,594	52	4,194
Sep	392	32	1,398	671	785	44	49	734	91	1,740
Oct	281	85	2,317	1,121	1,236	32	48	1,143	46	2,461
Nov	747	298	1,236	659	581	20	46	529	29	2,225
Dec	-	-	-	-	-	-	-	-	-	-

Table 3.11. Percentage in Different Classes of *Artemia* Population Means from Table 3-10 in 2021

	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	-	-	-	-	-	-	-	-	-	-	-
Mar	100	0	100	0	0	0	0	0	0	0	0
Apr	95	5	100	0.3	0.3	0	0	0	0	0	0
May	15	50	65	35	35	0	0	0	0	0	0
Jun	12	20	32	68	44	24	24	1	96	3	76
Jul	7	4	11	89	54	34	14	1	98	1	86
Aug	6	2	7	93	57	36	7	0.4	98	1	93
Sep	11	1	11	89	49	40	2	1	96	3	98
Oct	15	3	19	81	41	40	3	2	94	4	97
Nov	38	14	52	48	25	23	3	4	90	6	97
Dec	-	-	-	-	-	-	-	-	-	-	-
Western Sector											
Feb	-	-	-	-	-	-	-	-	-	-	-
Mar	100	0	100	0	0	0	0	0	0	0	0
Apr	95	5	100	0	0	0	0	0	0	0	0
May	14	52	66	34	34	0	0	0	0	0	0
Jun	9	20	30	70	46	24	23	1	95	4	78
Jul	5	6	11	89	55	34	14	0	98	2	86
Aug	5	2	8	92	55	37	7	0	99	1	93
Sep	12	0.3	12	88	54	34	2	2	95	4	98
Oct	19	4	24	76	38	38	7	3	93	5	93
Nov	29	10	39	61	28	33	4	2	92	7	96
Dec	-	-	-	-	-	-	-	-	-	-	-
Eastern Sector											
Feb	-	-	-	-	-	-	-	-	-	-	-
Mar	100	0	100	0	0	0	0	0	0	0	0
Apr	95	5	100	0.4	0.4	0	0	0	0	0	0
May	15	50	65	35	35	0	0	0	0	0	0
Jun	16	19	34	66	41	24	26	0.5	98	1	74
Jul	8	3	11	89	54	35	14	1	98	1	86
Aug	6	2	7	93	59	34	8	1	98	1	92
Sep	10	1	11	89	45	44	2	1	96	2	98
Oct	15	3	18	82	42	40	3	2	95	4	97
Nov	41	14	55	45	24	21	3	5	89	6	97
Dec	-	-	-	-	-	-	-	-	-	-	-

<u>Instar Analysis</u>

The instar analysis shows patterns similar to those of the lake-wide and sector analysis, but provides more insight into *Artemia* reproductive cycles occurring at the lake (Figure 25). Instars 2 were proportionally more abundant than Instars 1 in March. By April all age classes (1 through 7) of naupliar instars and juveniles were present and comprised approximately 65% of the *Artemia* population while adults comprised the remainder (35%). The proportion of naupliar instars and juveniles combined fell precipitously beginning in May, and proportions remained low until September and slowly increased afterward. The presence of naupliar instars and juveniles throughout the summer and fall, indicating continuous hatching and naupliar development in spite of much lower proportions of ovoviviparous females compared to oviparous females.

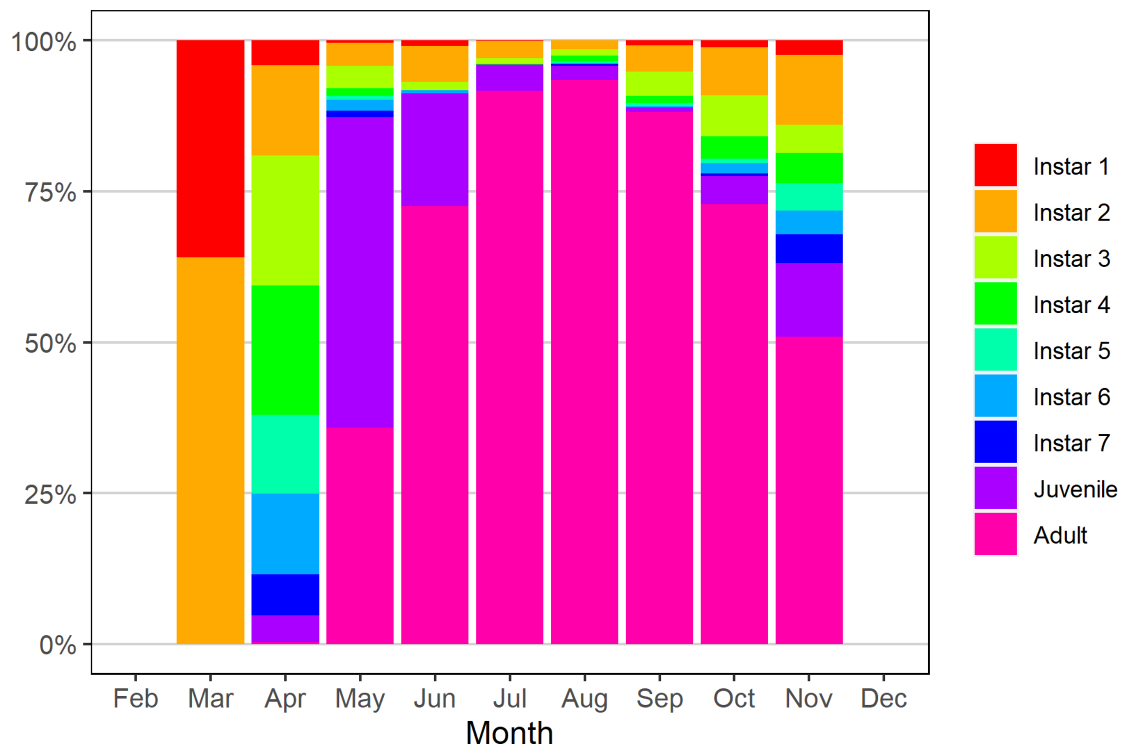


Figure 25. Compositional Changes of *Artemia* Instars and Adults in 2021

Biomass

Mean lake-wide *Artemia* biomass rapidly increased from 2.78 g/m^2 in April to the peak of 41.2 g/m^2 in June (Table 3-13). In July, however, the mean biomass quickly dropped to 19.0 g/m^2 and slowly declined throughout the rest of the year. In the western sector, the peak was recorded in June at 49.3 g/m^2. The peak in the eastern sector of 33.1 g/m^2 was comparatively lower and broader, spreading over May and June. The higher peak in the western sector in 2021 conforms to the pattern recorded prior to 2020 when the peak was higher in the eastern sector. The biomass in the western sector was mostly lower than what observed in the eastern sector, except the June peak and a slight bump in biomass in August, indicating higher *Artemia* productivity in the eastern sector in 2021.

Table 3.12. *Artemia* Mean Biomass (g/m^2) in 2021

Month	Lake-wide	Western Sector	Eastern Sector
Feb	-	-	-
Mar	1.13	0.16	2.10
Apr	2.78	2.41	3.15
May	22.2	11.3	33.1
Jun	41.2	49.3	33.1
Jul	19.0	15.1	23.0
Aug	16.7	17.5	15.9
Sep	12.4	9.23	15.5
Oct	5.68	1.64	9.71
Nov	2.23	1.27	3.19
Dec	-	-	-

Reproductive Parameters and Fecundity Analysis

By June, fecund females were plentiful enough to conduct a fecundity analysis. In mid-June, approximately 24% of total adults were females with 96% oviparous (cyst-bearing), 3% ovoviviparous (naupliar eggs) and 1% undifferentiated eggs (Table 3-10, Table 3-14, Figure 26). From July through November, over 85% of females were ovigerous with the majority (68 to 98%) oviparous.

The lake-wide mean fecundity declined initially in July, but started and continued to increase throughout the rest of the summer and fall. The lake-wide mean fecundity was initially 29.6 +/- 0.8 egg per brood in June, decreased to 19.9 +/- 0.8 eggs per brood by August, and rebounded to 41.2 +/- 3.0 in October. The majority of fecund females were oviparous between July and October. The peak in the eastern and western sections occurred in October. Typically, mean female lengths are positively correlated with mean eggs per brood, and 2021 followed this pattern.

Table 3.13. ***Artemia*** **Fecundity Summary in 2021**

	# of Eggs/Brood				Female Length (mm)		
Month	Mean	SE	% Cyst	% Indented	Mean	SE	n
Lakewide							
Jun	37.0	1.6	98.6	57.1	9.4	0.1	7
Jul	25.8	1.1	98.6	53.5	9.2	0.1	7
Aug	23.3	1.0	97.2	59.2	9.0	0.1	7
Sep	38.1	2.4	96.8	62.9	9.4	0.1	7
Oct	46.2	2.1	100	56.0	9.6	0.1	5
Western Sector							
Jun	39.6	2.4	100	55.0	9.6	0.1	4
Jul	27.2	1.6	100	50.0	9.2	0.1	4
Aug	23.2	1.4	95.1	63.4	9.0	0.1	4
Sep	41.7	3.7	96.8	51.6	9.1	0.2	4
Oct	40.5	2.8	100	60.0	9.2	0.3	2
Eastern Sector							
Jun	33.5	1.8	96.7	60.0	9.1	0.1	3
Jul	23.9	1.3	96.8	58.1	9.1	0.1	3
Aug	23.5	1.2	100	53.3	9.1	0.1	3
Sep	34.5	3.1	96.8	74.2	9.7	0.1	3
Oct	50.1	2.7	100	53.3	9.8	0.1	3

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

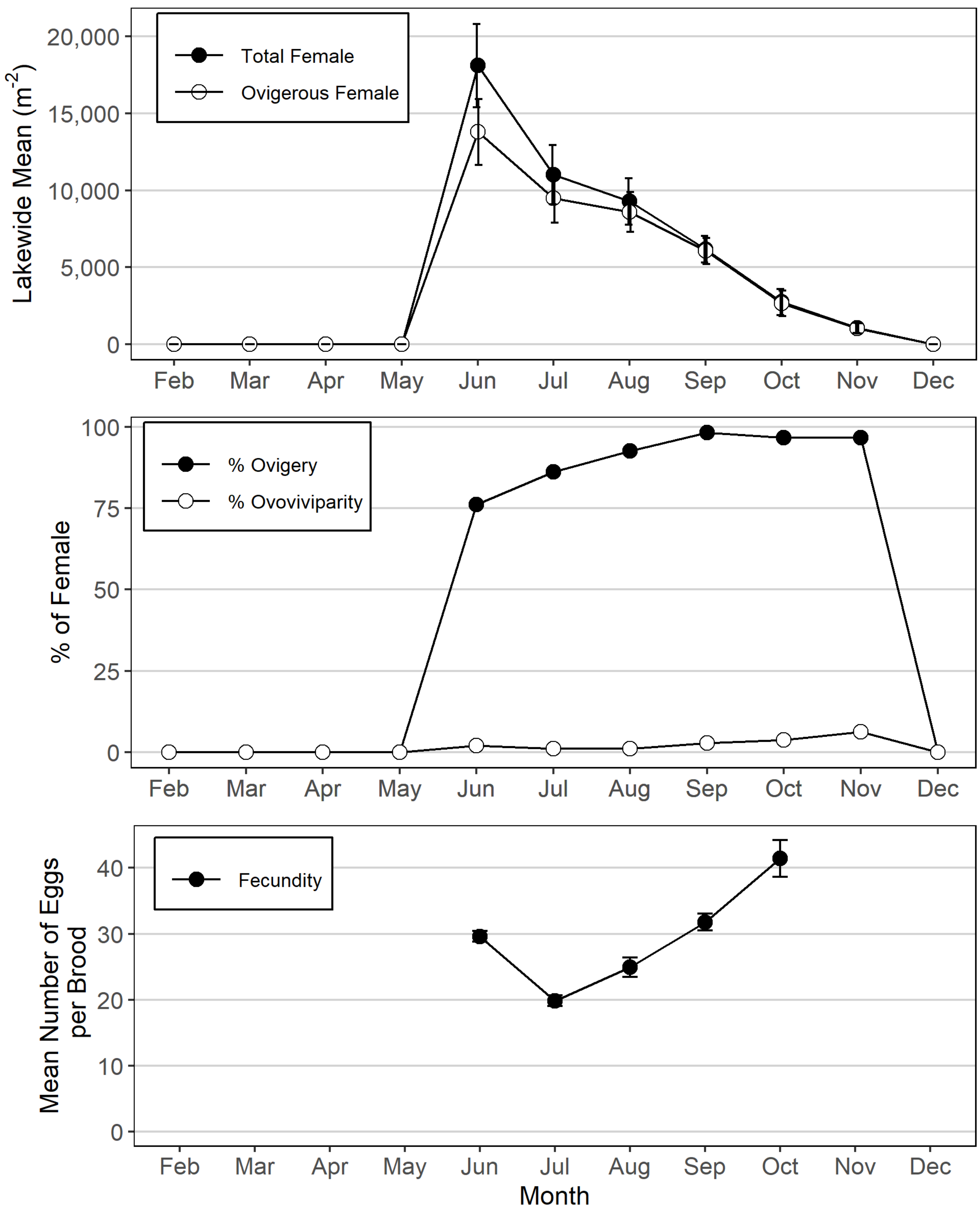


Figure 26. *Artemia* Reproductive Parameters and Fecundity between June and October in 2021

<u>*Artemia* Population Statistics</u>

The 2021 *Artemia* population experienced a post-meromictic peak as the annual mean adult *Artemia* population almost doubled from 12,991 m^{-2} in 2020 to 23,177 m^{-2} in 2021, exceeding the long-term average of 18,626 m^{-2} (Table 3-15). The mean population value of 23,177 m^{-2} for 2021 was, however, lowest among five recorded post meromixis population peaks (1989, 2004, 2009, 2013, and 2020), following the declining trend of the post meromixis population peaks (Figure 27). The centroid decreased to 195 days (July 14th) from 209 days in 2020 and from 221 days in 2019, and dipped below 200 days for the first time since 2015 (Figure 28). Adult *Artemia* population quickly increased to a peak in June, exceeding the long-term average between May and June, but sharply dropped to and followed the long term-average level starting in July (Figure 29).

In 2021, the peak average adult abundance, which occurred in June, was above June LTA and fourth highest for June since 1987 (Figure 30). The May average was fifth highest for May since 1987 as well. Between July and November monthly averages remained below LTA. The monthly average naupliar instar (instars 1 to 7) abundance peaked in March, and the 2021 peak was second highest for March (Figure 31). The April average was also found above LTA, but much smaller than what recorded in 2020 (51,885 m^{-2} compared to 113,491 m^{-2}). Instar abundance declined sharply in May and remained below LTA for the rest of 2021 even though instar abundance rose again in November.

Table 3.14. Summary Statistics of Adult *Artemia* Abundance between May 1 and November 30

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	23,952	22,621	54,278	226
1988	27,639	25,505	71,630	207
1989	36,359	28,962	92,491	249
1990	20,005	16,775	34,930	230
1991	18,129	19,319	34,565	226
1992	19,019	19,595	34,648	215
1993	15,025	16,684	26,906	217
1994	16,602	18,816	29,408	212
1995	15,584	17,215	24,402	210
1996	17,734	17,842	34,616	216
1997	14,389	16,372	27,312	204
1998	19,429	21,235	33,968	226
1999	20,221	21,547	38,439	225
2000	10,550	9,080	22,384	210
2001	20,031	20,037	38,035	209
2002	11,569	9,955	25,533	200
2003	13,778	12,313	29,142	203
2004	32,044	36,909	75,466	180
2005	17,888	15,824	45,419	192
2006	21,518	20,316	55,748	186
2007	18,826	17,652	41,751	186
2008	11,823	12,524	27,606	189
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	16,324	11,302	53,813	179
2013	26,033	31,275	54,347	196
2014	13,467	7,602	42,298	194
2015	7,676	5,786	18,699	185
2016	10,687	10,347	18,498	220
2017	15,158	15,536	26,064	221
2018	12,120	12,024	21,836	216
2019	13,541	12,590	26,531	221
2020	12,991	13,427	24,353	209
2021	23,177	23,480	50,731	195
Mean	18,626	17,462	42,519	210
Min	7,676	5,786	18,498	179
Max	36,643	36,909	105,245	252

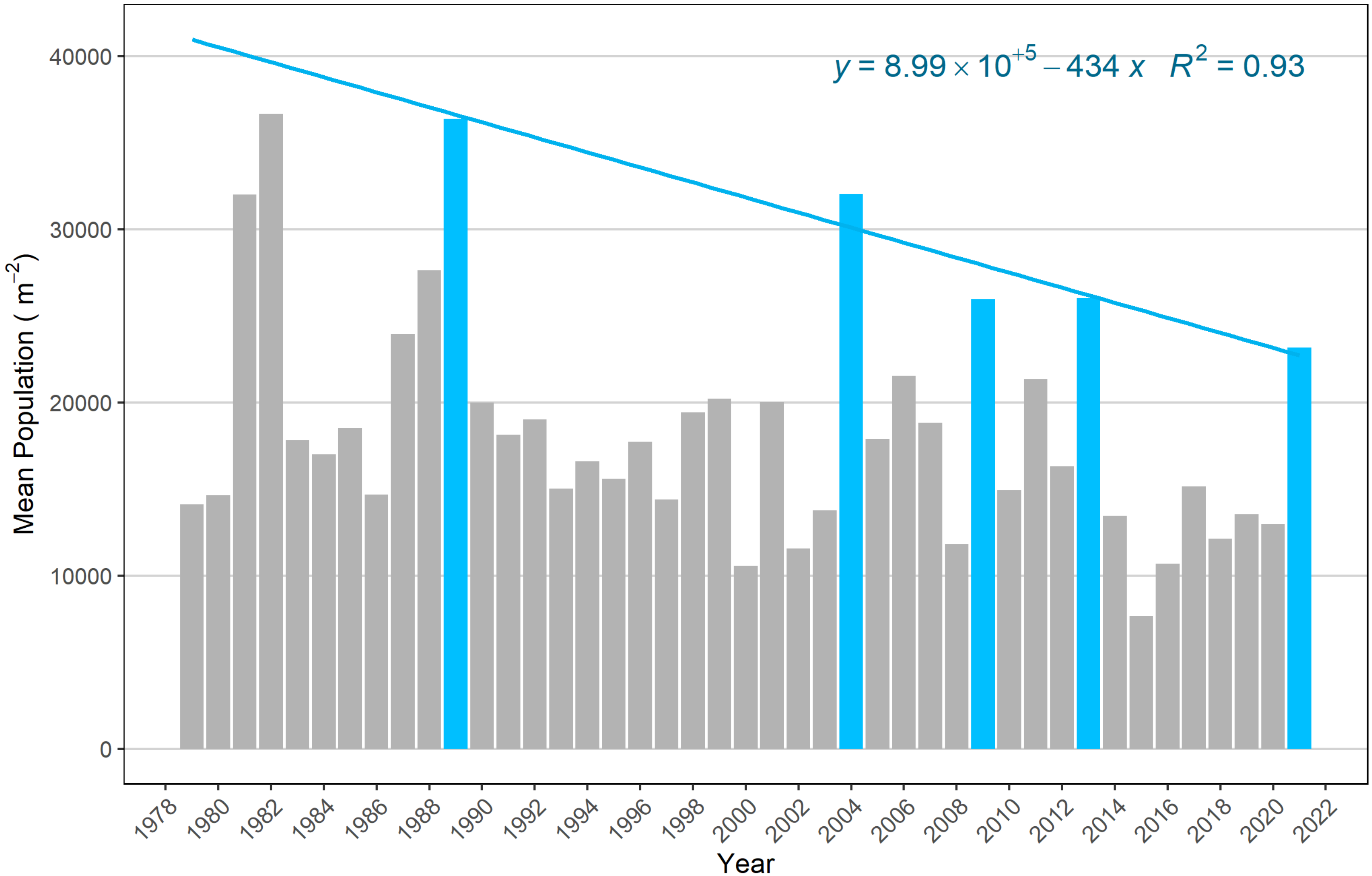


Figure 27. Adult *Artemia* Population Mean since 1979

Bars filled with light blue color indicate years with post meromixis Artemia population peaks.

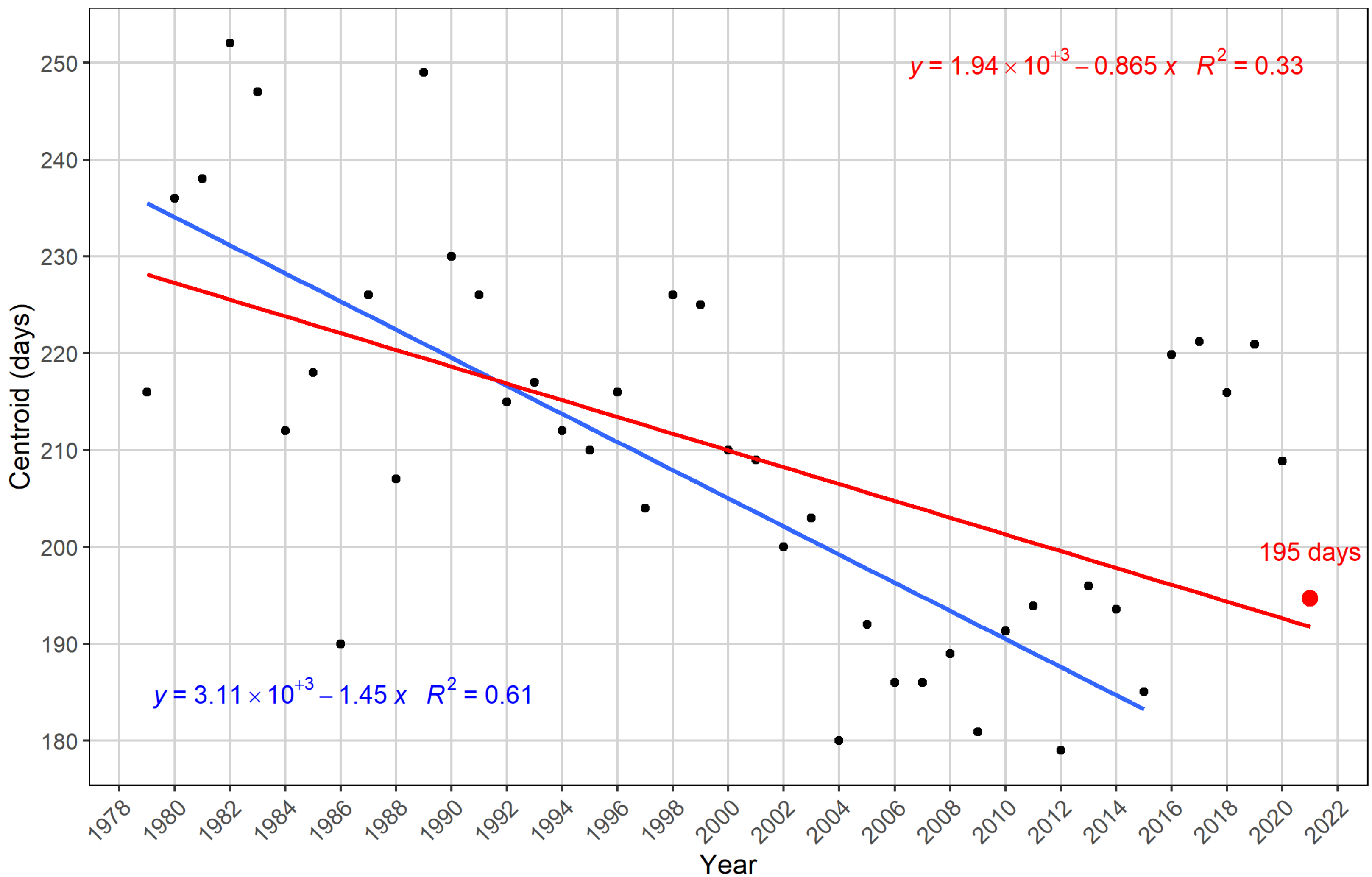


Figure 28. Adult *Artemia* Population Centroid

A red dot indicates a value in 2020. The blue line indicates the linear trend between 1979 and 2015 while the red line indicates the linear trend for all monitoring years.

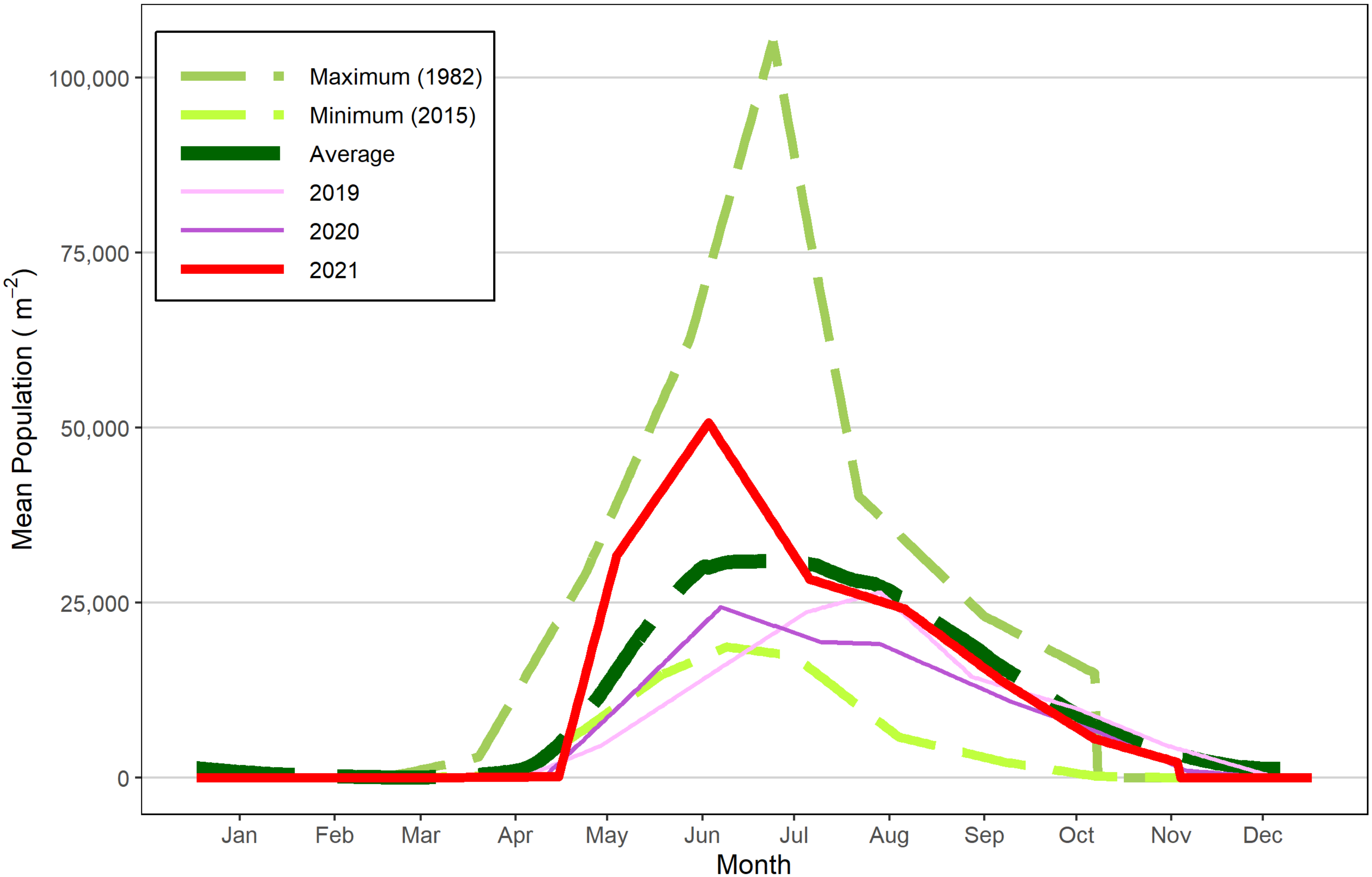


Figure 29. Mean lake-wide Adult *Artemia* Population (m^{-2}) since 1987

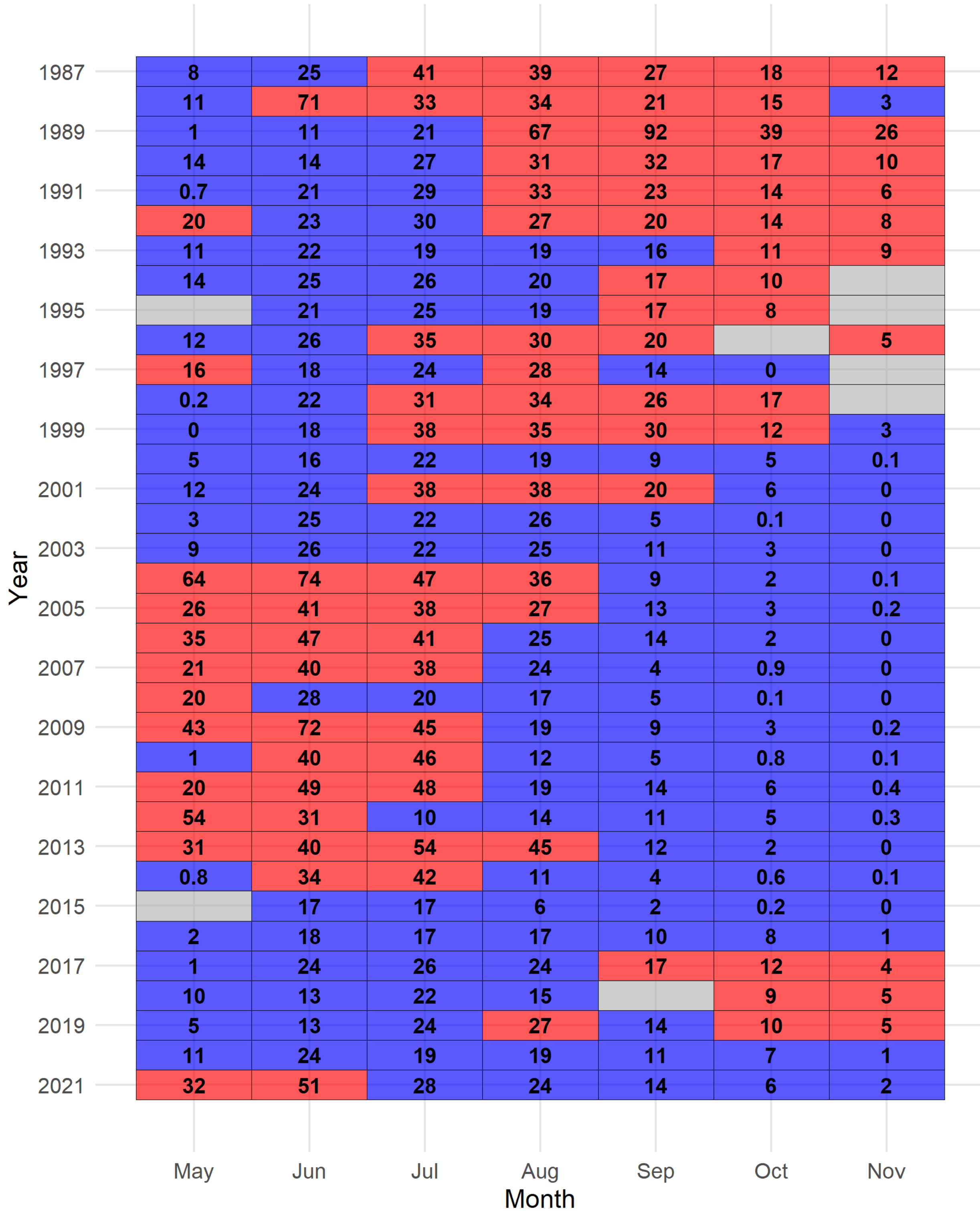


Figure 30. Monthly Average Adult *Artemia* Abundance of 12 Stations

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.

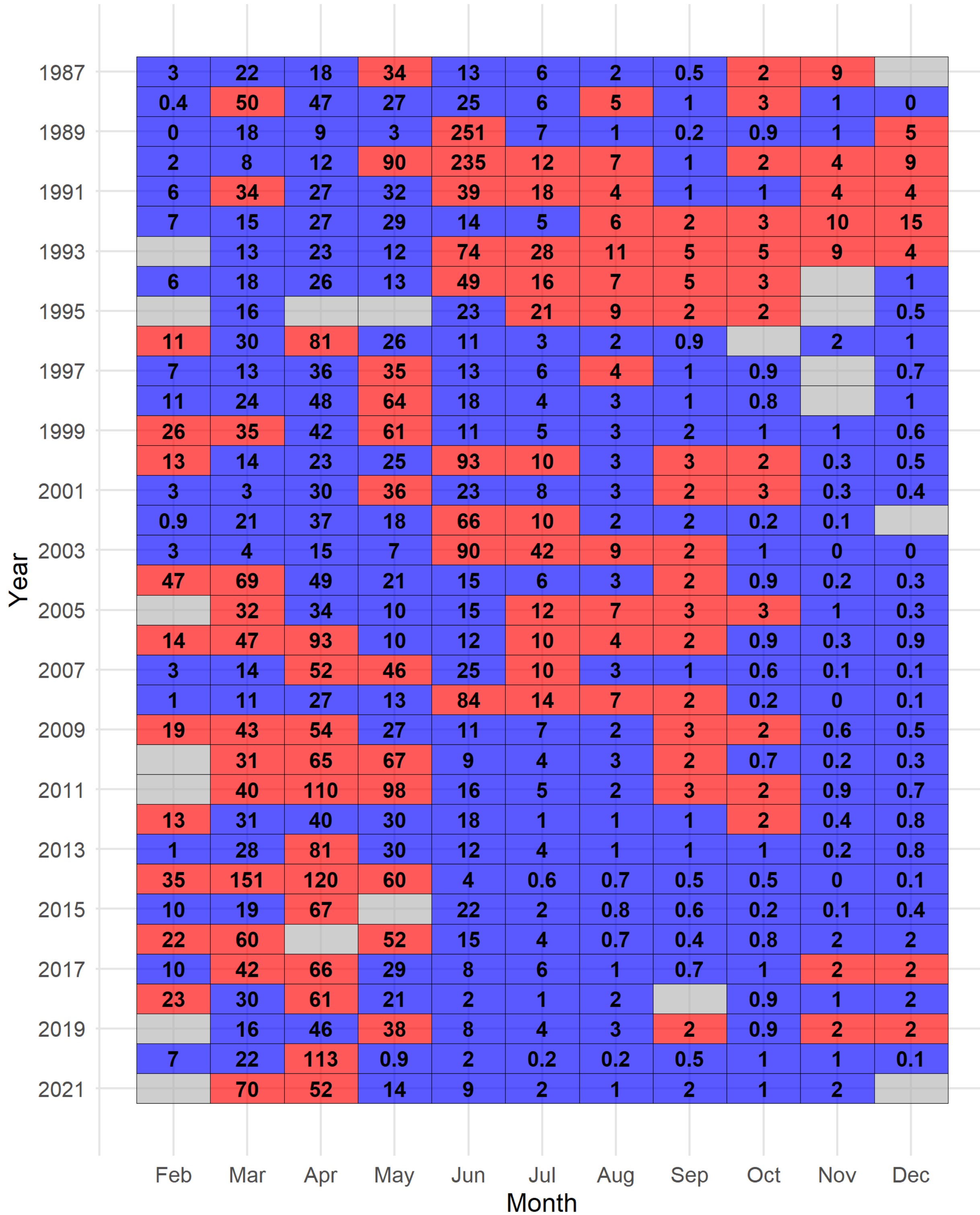


Figure 31. Monthly Average Instars 1-7 *Artemia* Abundance of 12 Stations

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.

Analysis of Long-Term Trends

Salinity and Mono Lake Elevation

The salinity of Mono Lake was closely associated with lake elevation across all monitoring stations, especially at shallower depths (Table 3-16). The strongest correlations were found for all stations with depths reaching 20m when salinity readings between 1 and 20 were averaged. Stations 2, 6, and 7 show correlation coefficients of 0.98, corresponding to r^2 of 0.96 (a coefficient of determination). In 1983, the lake level elevation ranged between 6,373ft and 6,380ft, and Dana and Lenz (1986) reported 88 g/L as the actual Mono Lake salinity at the depth of 8m in July of that year at the lake level of 6,377,3ft (based on LADWP's database). Between August and November of 1995, the monthly lake level was at 6,377,4ft with salinity ranging from 89.8 g/L in August to 87.2 g/L in October[1]. The lake level close to 6,377.3ft was recorded between November of 2015 and February of 2017, salinity recorded during this period ranged from 86.3 g/L to 87.4 g/L. These values from both periods roughly correspond to the value reported by Dana and Lenz. The value of 88 g/L, reported by Dana and Lenz, is very close to 88.2 g/L, calculated using the simple linear regression based on average salinity between 1 and 10m at Station 6 at the lake level of 6,377.3ft (Figure 32).

Simple linear regression between lake elevation and salinity between 1 and 20m at Station 6 explains over 96% of variability of salinity readings (Table 3-17). This regression predicts salinity level of 64.8 g/L between 1 and 20m in depth at the target lake level of 6,390ft. Salinity is expected to exceed 90 g/L when the lake level falls below 6,376.3ft, and 95 g/L and 100 g/L when the lake level falls below 6,373.6ft and 6,370.3ft, respectively. The lowest monthly lake level was recorded at 6,371.7ft in December of 1981, which would have resulted in a salinity value of approximately 97.6 g/L. Conversely, prior to exports from the Mono Basin, the lake level exceeded 6,400ft, corresponding to an approximate salinity value of 50 g/L even though the simple linear extrapolation may not be appropriate for values far outside of the data range.

Predicting salinity below 20m of depths is more difficult due to lake stratifications, during which salinity continues to decline even after the lake level stops to rise. This situation results in a no-linear, highly variable relationship and significant interactions between meromictic and monomictic periods (Figure 33, Table 3-17). Salinity below 20m during monomictic periods, however, can be predicted with much higher accuracy (r^2 = 0.97 compared to r^2 = 0.30 during meromictic periods). Based on this regression, salinity level at the target lake level is predicted to be 68.8 g/L - slightly higher than the predicted salinity above 20m in depth. Salinity below 20m should exceed 90 g/L at the lake level below 6,375.9ft, 95 g/L at the lake level below 6,372.5ft, and 100 g/L at the lake level below 6,369.2ft. As mentioned previously, the lake level

[1]No salinity data is available for November.

was at its lowest on record at 6,371.7ft in 1981, and Mono Lake was most likely in a monomictic state due to the prolonged decline of the lake level preceding the recent minimum. Based on the regression during monomictic years, the estimated salinity below 20m of depth at that time was 96.2 g/L (Figure 34).

Between 1970 and 1981, Mono Basin exports exceeded 100,000 acre-feet per year nine times, and Mono Lake came closest to exceeding the salinity level of 100 g/L. With Mono Lake elevation fluctuates around 6,380ft, Mono Basin export will be limited to 4,500 acre-feet per year or will not be allowed if the lake level is projected to, or falls below 6,377 ft. Consequently, Mono Lake elevation is now mainly driven by the climatic pattern. For past two years the Mono Basin has experienced very dry conditions, 52% and 37% of Normal in 2020 and 2021 water years, respectively (6th and 2nd driest on record). The summer of 2021 was the warmest on record since 1955. A combination of drier and warmer conditions decreases the lake level, which, in turn, increases salinity of the lake. As of December 2021, the lake level was at 6,379.5 ft. The salinity in December 2021 was estimated to be 84.9 g/L between 1 and 20m and 84.6 g/L[2] below 20m, up from 82.1 g/L and 82.2 g/L in December 2020 and 79.4 g/L and 83.6 g/L in December 2019 for the respective depths The current salinity conditions are still much lower than the historical highs; however, since the onset of the driest five-year period on record between 2012 and 2016, five of top ten driest years since 1935 have occurred. Mono Lake could become more saline quickly under much drier and warmer conditions. It should be noted, however, that *Artemia* population exceeded 30,000 m^{-2} in 1981, 1982, and 1989 and 25,000 m^{-2} in 1988 in spite of salinity exceeding 90 g/L. In the case of 1981 and 1981, salinity throughout the water column most likely exceeded 95 g/L. The *Artemia* population in Mono Lake was able to thrive during years when Mono Lake had its highest salinity levels recorded in the last 100 years.

[2] The salinity value for August was 81.9 g/L, relatively low for the given lake level: thus, the actual salinity in December should have been lower than 84.6 g/L which was estimated based on the monomictic deep water regression.

Table 3.15. Relationships between Salinity and Lake Elevation for Different Depth Classes

	Depth						
Station	**1 to 10m**	**1 to 15m**	**1 to 20m**	**11 to 20m**	**21 to 30m**	**21 to 38m**	**31 to 38m**
1	-0.81	-0.83	-	-	-	-	-
2	***-0.96***	***-0.97***	***-0.98***	***-0.96***	-	-	-
3	***-0.90***	***-0.94***	***-0.96***	***-0.93***	-0.53	-	-
4	***-0.92***	***-0.95***	***-0.97***	***-0.94***	-0.49	-0.48	-0.38
5	-0.89	***-0.94***	***-0.94***	-0.90	-	-	-
6	***-0.95***	***-0.97***	***-0.98***	***-0.97***	-0.72	-0.69	-0.61
7	***-0.95***	***-0.97***	***-0.98***	***-0.97***	-0.75	-	-
8	-0.88	***-0.91***	***-0.92***	-0.85	-	-	-
9	-0.81	-0.86	-	-	-	-	-
10	-0.84	***-0.90***	***-0.93***	***-0.92***	-0.82	-	-
11	-0.74	-	-	-	-	-	-
12	-0.87	***-0.92***	***-0.95***	***-0.93***	-0.48	-0.47	-0.47

Monthly average lake elevations were used. Stations 1 and 9 were not included due to a lack of long-term data, and Station 11 was not included because of its shallow depth.

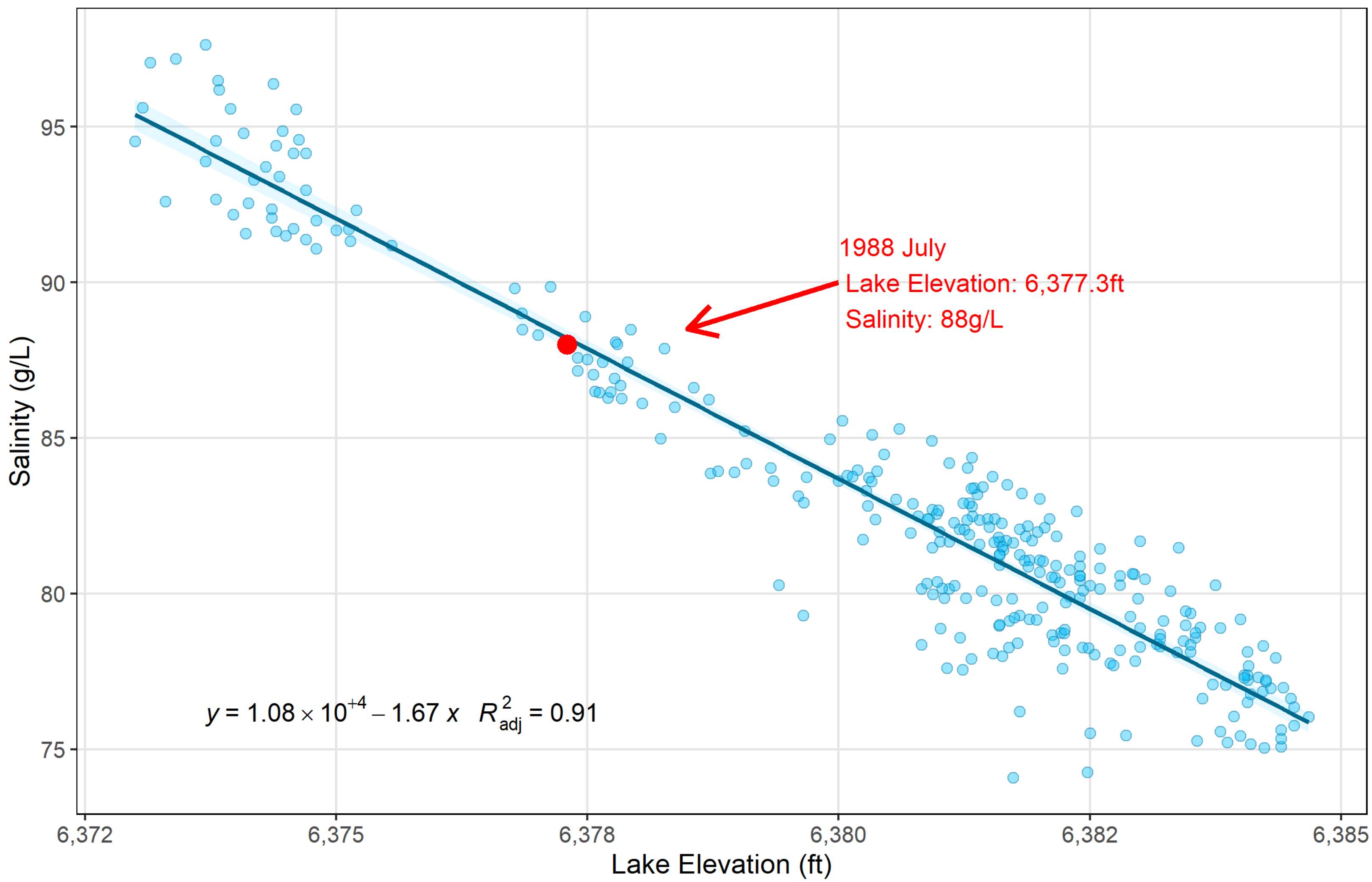


Figure 32. Relationships between Lake Elevation and Salinity averaged between 1 and 10m at Station 6

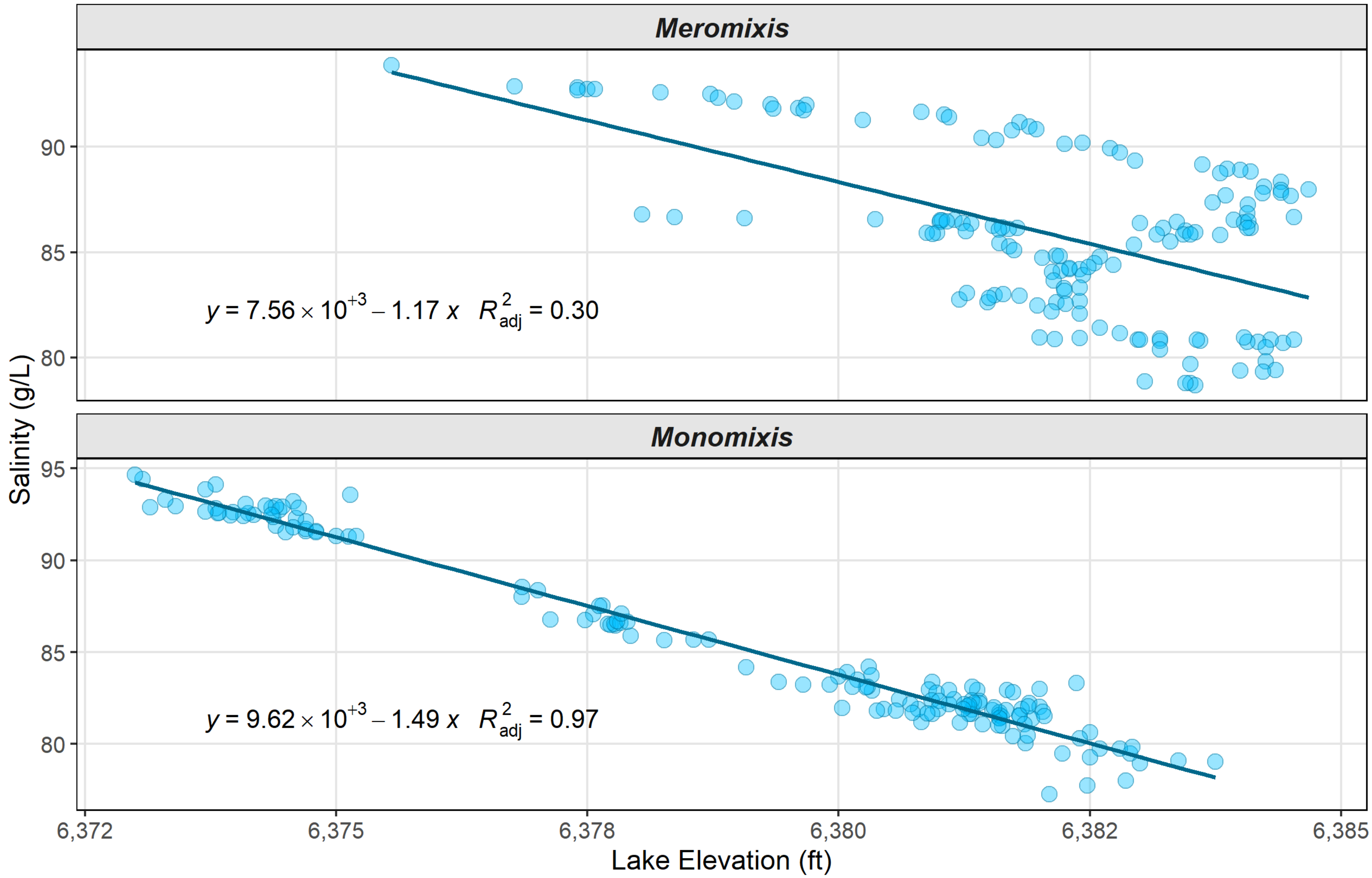


Figure 33. Relationships between Salinity and Lake Elevation below 21m separated for Meromictic and Monomictic Periods at Station 6

Table 3.16. Correlation Coefficients and Simple Linear Regression Results for Overall, Meromixis and Monomixis Time Periods

Depth	Group	r	r^2	Interaction	Regression
1 to 10m	Overall	-0.95	0.91	-	*y = -1.67 x (Elevation) + 10753*
1 to 20m	Overall	-0.98	0.96	-	y = -1.6 x (Elevation) + 10295
21 to 30m	Overall	-0.72	0.73	0.0487	-
	Meromixis	-0.59	0.35	-	*y = -1.27 x (Elevation) + 8163*
	Monomixis	-0.99	0.98	-	*y = -1.5 x (Elevation) + 9677*
21 to 38m	Overall	-0.69	0.71	0.0092	-
	Meromixis	-0.55	0.30	-	*y = -1.17 x (Elevation) + 7560*
	Monomixis	-0.99	0.97	-	*y = -1.49 x (Elevation) + 9616*
31 to 38m	Overall	-0.61	0.66	0.0005	-
	Meromixis	-0.46	0.21	-	*y = -1 x (Elevation) + 6448*
	Monomixis	-0.98	0.96	-	*y = -1.48 x (Elevation) + 9513*

Interaction indicates probabilities associated with the interaction term of the regression.
Statistics for “1 to 10m” and “1 to 20m” do not include an interaction term.
Correlation coefficients (r) were calculated without an interaction term.

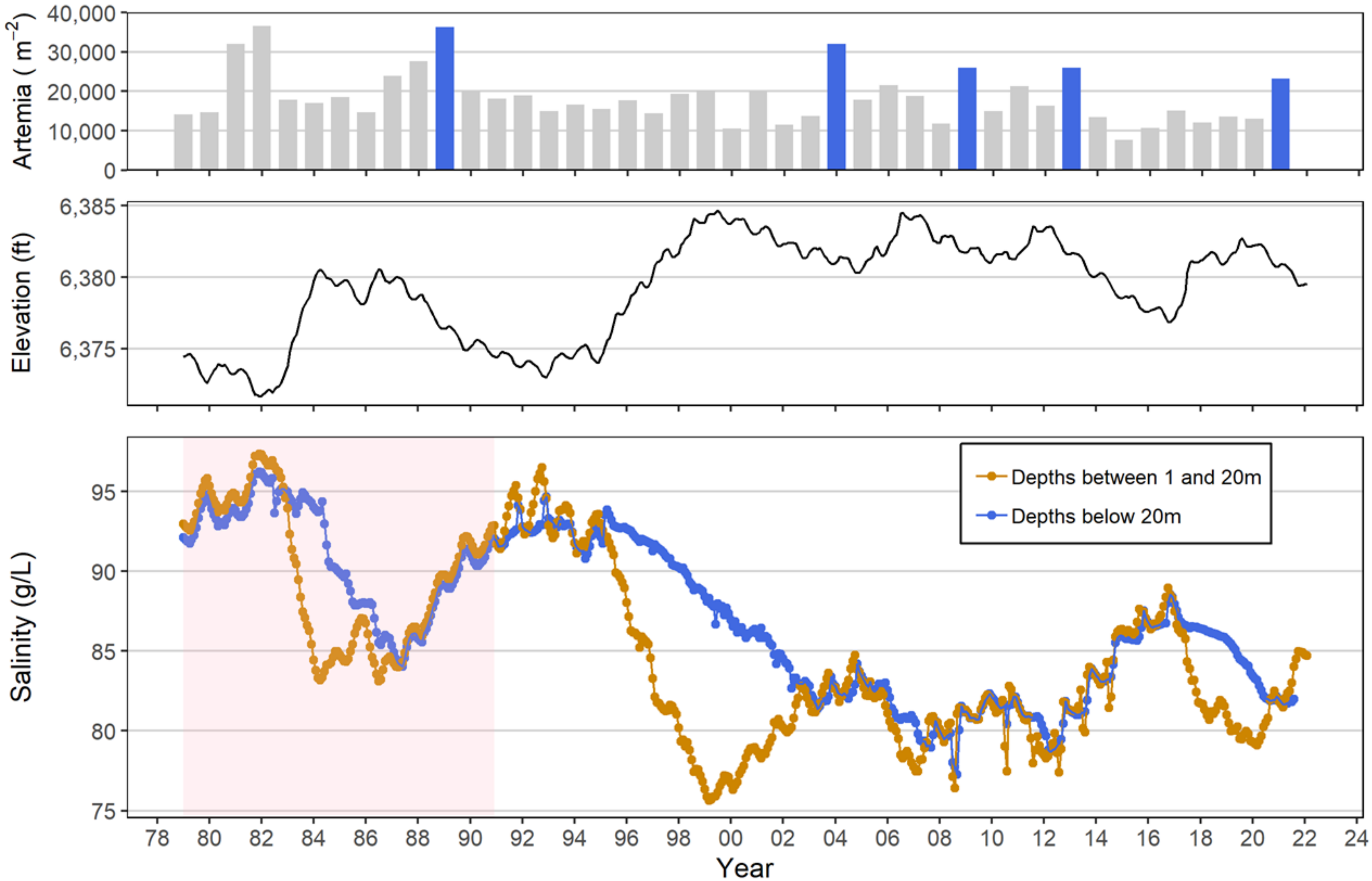


Figure 34. Monthly Time Series of Salinity for Depths between 1 and 10m and below 20 with Estimated Salinity Values between 1980 and 1990

<u>*Artemia* Population Peak</u>

Post-meromictic *Artemia* population peaks have been recorded one or two years after the end of meromictic periods. The latest population peak was observed in 2021, the fifth such event recorded. The previous four peaks occurred in 1989, 2004, 2009, and 2013. *Artemia* abundance and ammonium accumulation during each meromictic event are summarized in Table 3-18, Figure 35, and Figure 36.

Mean *Artemia* population has been declining since 1978, and the pattern is much stronger for post-meromictic population peaks (Figure 35 A). The strong linear decline correlates very strongly with the three-year average of mean *Artemia* population prior to the peak ($r = 0.97$), indicating lower abundance prior the population peak tended to result in lower population peaks (Figure 35 B). Secchi readings, especially annual maximum readings, also show strong negative correlations with *Artemia* population peaks (Figure 35 C), indicating more intense grazing during peak years. Salinity shows much more complicated relationships with *Artemia* population peaks (Figure 36 A and B). The population peak in 1989, highest among five known peaks, occurred when salinity in shallower depths (A) and deeper depths (B) exceeded 90 g/L while all other peaks showed corresponding salinity levels around 82 g/L; thus, there was no correlation between population peaks and salinity when the 1989 peak was removed. Ammonium accumulations prior to the peaks show a strong correlation with *Artemia* population peaks, except the 2021 peak, which should have been much closer to 30,000 m^{-2} given the ammonium accumulation (Figure 36 C). Deviation from the trend in 2021 indicates that other factors may have exerted stronger influences on the most recent *Artemia* population peak.

With regard to lake level, Mono Lake was at its most recent high stand of 6,382.7 ft in August 2019 and the 2021 *Artemia* peak occurred when the lake level was slightly above 6,380ft (Figure 37 A). Lake levels preceding the 2021 peak were lower than during the previous three *Artemia* peaks, but higher than the first peak. Water year 2021 runoff was much lower than other years, ranking the second driest on record (Figure 37 B). The cumulative runoff during the formation of the meromixis in 2021 was lower than the first two peaks but higher than the third and fourth peaks (Figure 37 C). This resulted in a much stronger chemocline than the third and fourth meromictic periods, which, in turn, lead to increased accumulation of ammonium during the 2017-20 meromixis. Summer temperature in 2021 was much higher than the long-term average and also all other years, resulting in a relatively strong positive correlation since 1980, although winter and spring temperatures did not standout against other four years (Figure 37 D).

The size of the *Artemia* population preceding a peak appears to be the single most important variable to predict the magnitude of a post-meromictic population peak. Ammonium accumulation, too, was informative until the 2021 peak, which followed the declining population trend instead of responding as expected to ammonium accumulation. The only linear trend detected so far among environmental variables starting in the late 70s or early to mid-80s is ambient temperature and chlorophyll *a* concentration in the upper mixing layer[3] (r = 0.44 between monitoring years and mean annual chlorophyll *a* concentration, data not shown) (Figure 11, Figure 38). The positive trend of ambient temperature, however, was translated only into rising water temperature in the deepest part of the lake, below 30m in depth (Figure 39, Figure 40). This may not accurately reflect a temporal trend of water temperature because of the time period during which the data are available (1991-2021). Mono Lake was going through a monomictic period in the early 90s, during which the lake turned over annually introducing colder water of the upper mixing layer throughout the water column in the late fall to winter. Contrarily, Mono Lake was meromictic in the tail end of the available data period, during which relatively warmer water is trapped below chemocline, resulting in higher water temperature in the deeper part of the lake. These two factors may have confounded the apparent temporal trend. Continuous monitoring of water temperature should provide better insights into water temperature trends.

Chlorophyll *a* concentration in the upper mixing layer shows an increasing trend, especially in summer, starting in 2008 (r = 0.63). Much higher concentration levels were found after the 2013 *Artemia* population peak, coinciding with the five-year drought. The summer temporal trend of chlorophyll *a* is mirrored by the Secchi readings as the maximum readings in the summer started to decline in 2008 (one can argue the transition has started earlier than 2008 based on Figure 15), and remained below 1 m four times between 2015 and 2021. Chlorophyll *a* concentration follows the lake mixing regime as the concentration in the upper mixing layer decreases during meromixis due to ammonium locked up below the chemocline. The very similar pattern is exhibited by salinity, resulting in higher salinity leading to higher chlorophyll *a* and vice versa, except during the monomictic period (1987- 1994), during which chlorophyll *a* concentration was lower than subsequent years while salinity was much higher than subsequent years. Since 1995, salinity has remained relatively low due to four consecutive years with above normal runoff (1995-1999) and three wet years (2005, 2006, and 2011) compared to the period between the late 70s and early 90s during which the lake level hit its lowest on record and salinity was estimated to have approached 100 g/L.

[3] Chlorophyll *a* concentration data was incomplete in 2021; thus, the 2021 data were not included.

The pre-diversion salinity, estimated at less than 50 g/L, would likely have supported a more diverse phytoplankton community. Mono Lake most likely lost the diverse phytoplankton communities as salinity doubled, favoring species with high salinity tolerance, including *Artemia monica*. It has been reported that in terms of algal communities, higher salinity has resulted in dominance of one species (*Dunaliella* spp.), the loss of cyanobacteria, and more pronounced nitrogen deficiency in summer, leading to a reduction in primary production (Rushforth and Felix 1982). In Mono Lake, it has been reported that both the phototrophic *Picosystis* and diatom *Nitzschia* were equally abundant through the 90s (Jellison and Melack 1993, Roesler et al. 2002); however, *Picosystis* is now the dominant species, and *Nitzschia* has been absent in recent years (Phillips et al. 2021). When salinity decreased during and after the second meromixis (1995-2002), the algal community could have diversified as less salt tolerant species, including diatoms and cyanobacteria, were able to establish. Instead, *Picosystis* has become the dominant species, especially through the algal bloom induced by the severe drought (2012-2016), because of its ability to remain active under low light availability (Stamps et al. 2018). *Artemia* readily grazes *Picosystis*; yet, grazing intensity has declined in spite of increasing abundance of *Picosystis*, resulting the record low Secchi readings during summer months.

Between the late 1970s and early 1990s, salinity approached 100 g/L for the first in the recent history, and *Artemia monica* should have become the single most dominant invertebrate species as Dana and Lenz (1986) reported the salt tolerance between 159 g/L and 179 g/L for subadult *Artemia monica*. Notable increases in other invertebrate species have not been noted, even though salinity fell below 80 g/L and remained below 85 g/L since the second meromixis (1995-2002); thus, the diversification of the invertebrate community, which could have introduced predatorial species (i.e. *Corixidae* species), does not appear to have occurred. The minimum salinity of 75.6 g/L between 1 and 20m in depths reached in 1999 since the restoration efforts started was relatively low compare to the period between the late 1970s and early 1990s, but not low enough for other invertebrate species to establish. *Artemia* is the most important food source for migratory birds, especially California Gull and Eared Grebes; however, it appears that these species are responding to fluctuations of *Artemia* abundance, but not vice versa. A severe drought (dry and hot) affects *Artemia* population profoundly as seen in 2015, but it is not clear what are driving forces behind the overall declining trend of *Artemia* population.

Table 3.17. *Artemia* Population Summary during Meromixis and Monomixis

		Peak				
Meromixis	Duration	Year	Artemia abundance (m^{-3})	Average *Artemia* between peaks (m^{-3})	Reduction following a peak	NH_4 accumulation during meromixis (μM)*
1983-1987	5	1989	36,359		45%	180
				16,576		
1995-2002	8	2004	32,044		44%	144
				17,514		
2005-2007	3	2009	25,970		43%	41
				17,529		
2011	1	2013	26,033		48%	47
				12,108		
2017-2020	4	2021	23,177			116
Average			28,717	15,727	45%	

* Maximum monthly NH_4 reading recorded at depths of 28 and 35m at Station 6 prior to the population peak.

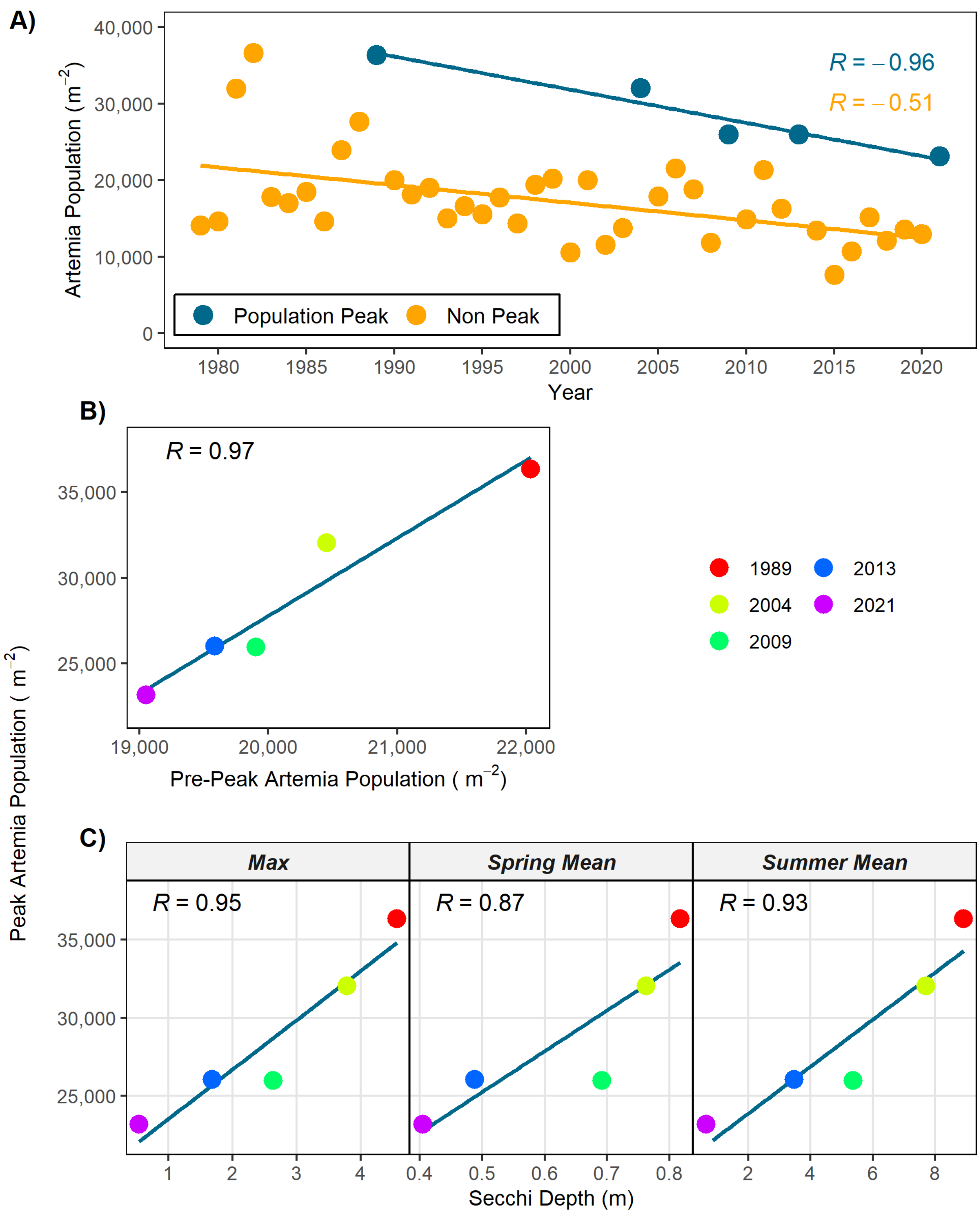


Figure 35. A) Mean Artemia Population since 1978, B) Relationship between Peak Artemia Population and 3 year Running Average of Artemia Population prior to the Peak, and C) Relationships between Peak Artemia Population and Secchi Readings

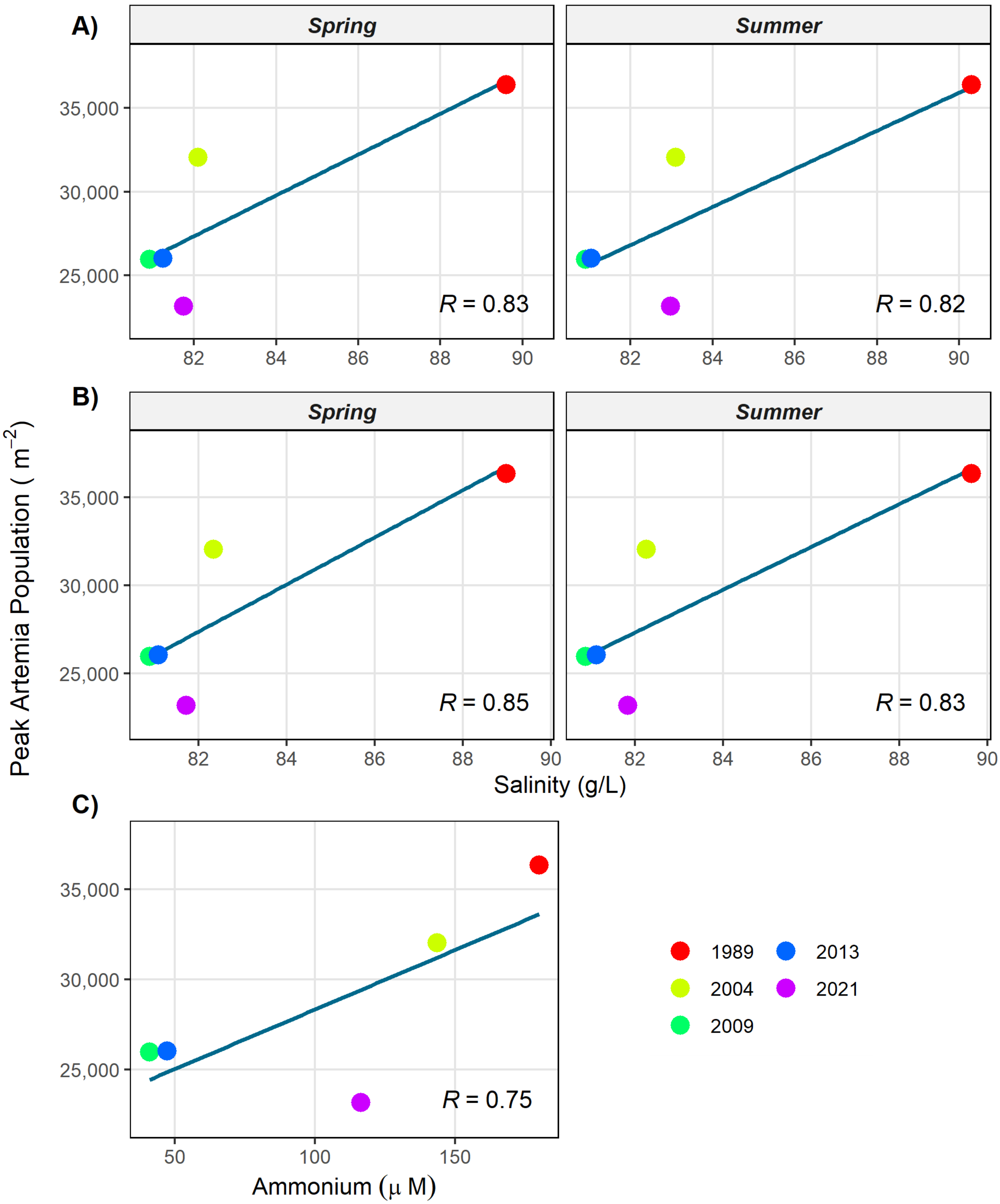


Figure 36. Relationship between Peak Artemia Population Mean and A) Salinity between 1 and 20 m, B) Salinity below 20 m, and C) Peak Ammonium Accumulation just before Artemia Population Peak

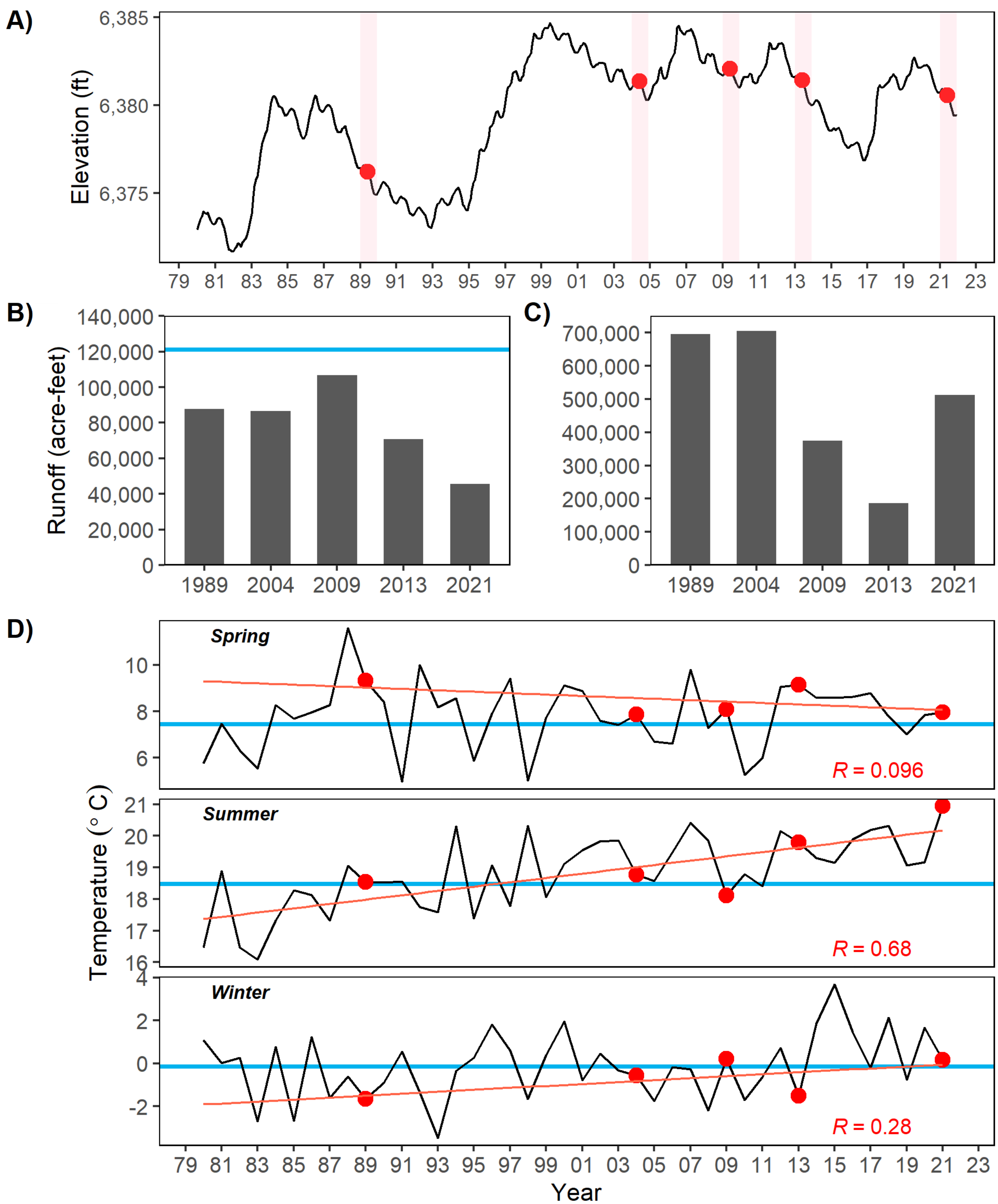


Figure 37. A) Mono Lake Monthly Elevation since 1980, B) Mono Basin Water Year Runoff, C) Total Mono Basin Water Year Runoff during the Formation of the Meromixis, D) Temperature for Lee Vining. Red dots indicate years with *Artemia* population peaks. Blue lines indicate the respective long-term averages.

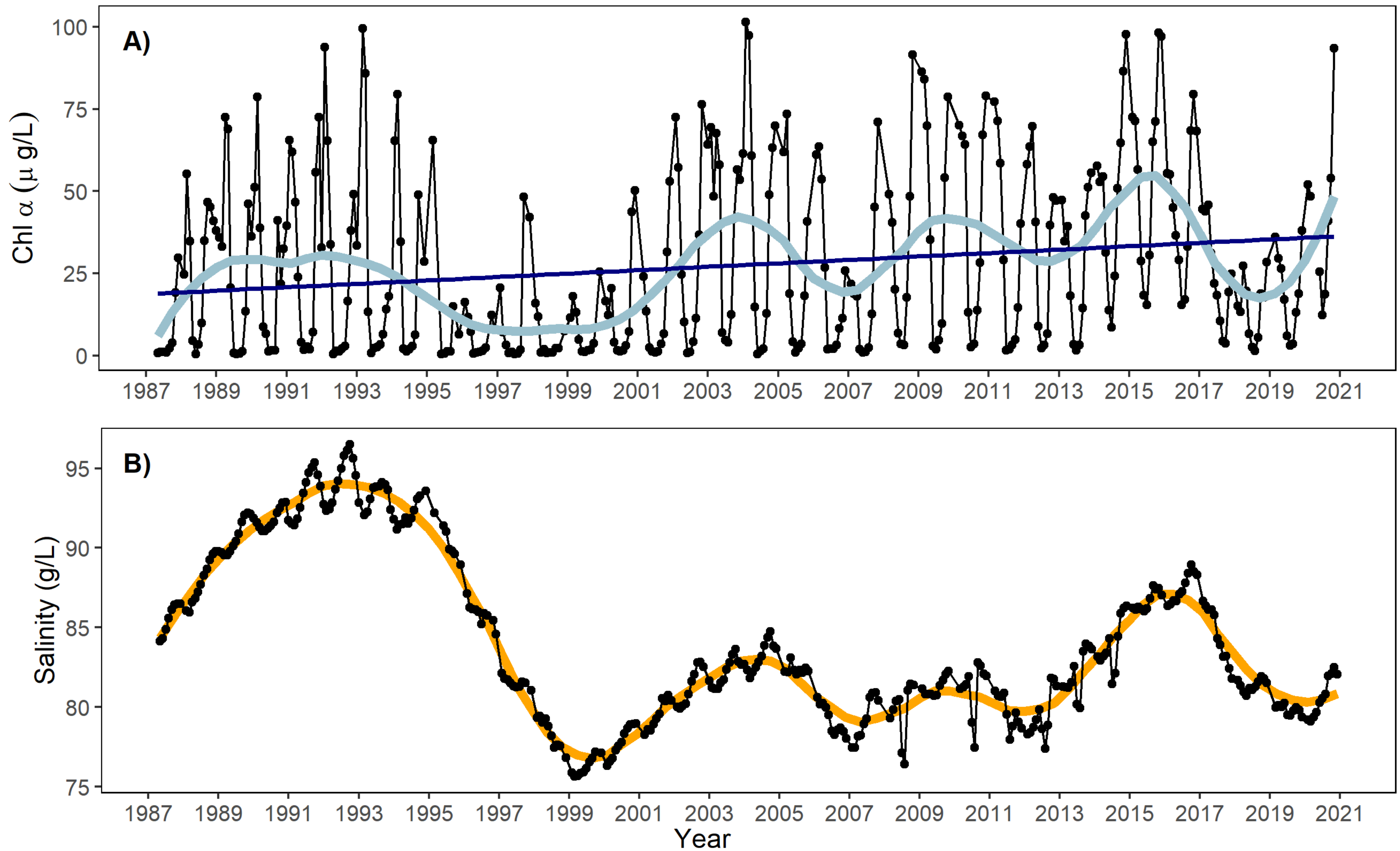


Figure 38. Monthly Readings with Loess (Locally Estimated Scatterplot Smoothing) using span of 0.2 since 1987 of A) 9m Integrated Chlorophyll a, B) Salinity between 1 and 20m

Salinity values prior to 1991 were estimated.

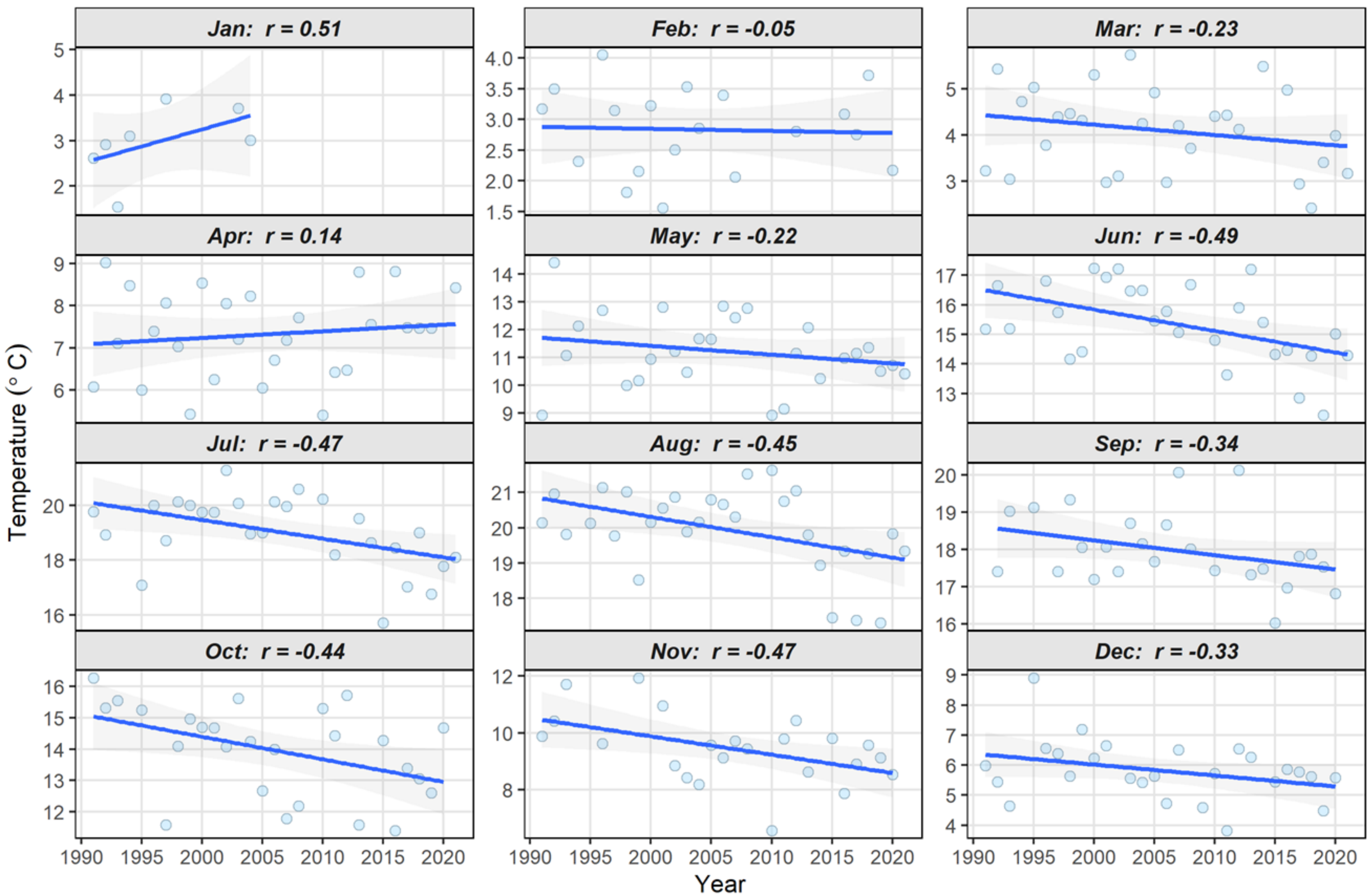


Figure 39. Average Water Temperature between 1 and 10 m of Depth for Each Month at Station 6

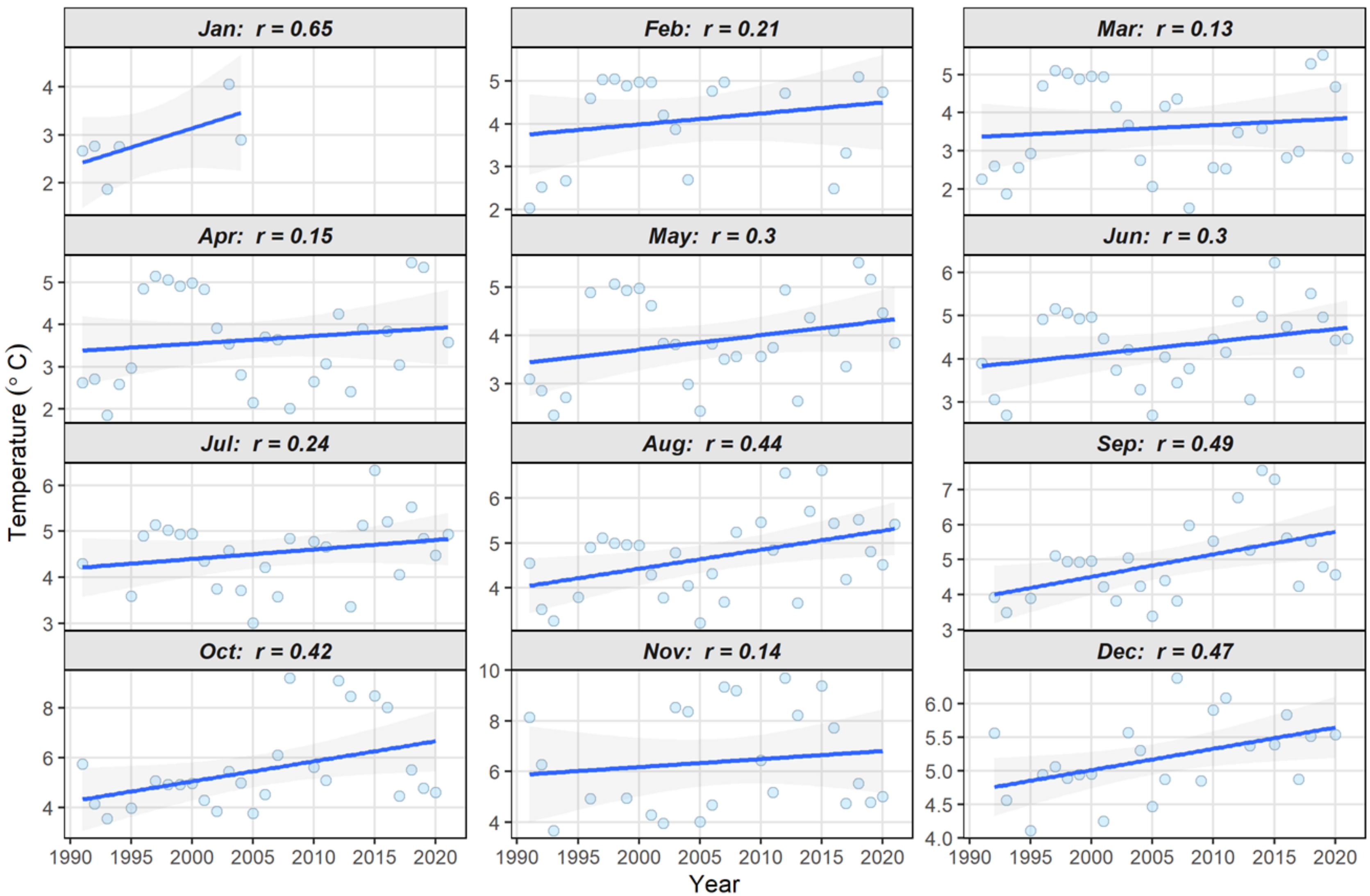


Figure 40. Average Water Temperature between 31 and 38 m of Depth for Each Month at Station 6

A Temporal Shift in Monthly Artemia Abundance

Figure 41, Figure 42, and Table 3-19 demonstrate a temporal shift in monthly *Artemia* abundance of adults and instars. For adults, a declining trend exists between August and November ($p < 0.002$), but the temporal trend is non-linear for months between May and July with a unimodal peak located somewhere between 2005 and 2010. Much stronger linear trends exist for instars, especially after monthly population means are log transformed[4]. Between February and April, trends were positive with April showing the strongest increasing trend ($r = 0.67$, $P < 0.0001$) while the rest of months showed negative trends. Negative trends were significant for summer months (June to August) and October. These trends suggest that instar population is peaking earlier especially in April and fewer second and third generations are present in later months. As reported previously, water temperature appears to be rising below 30m in depth, which could facilitate earlier hatching in spring, but water temperature in shallower depths appears falling, which could slow the growth. Overall chlorophyll *a* concentration is rising, but increasing concentration is not observed between February and April (Figure 43), when a more abundant food source in earlier months might support faster growth.

Females tend to produce cysts under low food availability and unfavorable environmental conditions as opposed to ovoviviparously-produced nauplii. It has been demonstrated previously that chlorophyll *a* concentration has been rising in summer months, indicating higher abundance of food sources, and monthly abundance of ovoviviparous females between July and September shows a rising trend even though trends are relatively weak with coefficient of coefficients ranging between 0.23 and 0.37 (Figure 44); females appear to be responding to higher availability of food sources. These trends, in turn, may be responsible for reversing a pattern of earlier occurrences of peak adult abundance in recent years. A rising trend of ovoviviparous females between July and September, however, is not sufficient to reverse the overall declining trend of *Artemia* adult population.

[4] There were very little differences in reported statistics between untransformed and transformed values for adults; thus, statistics based on untransformed values were reported.

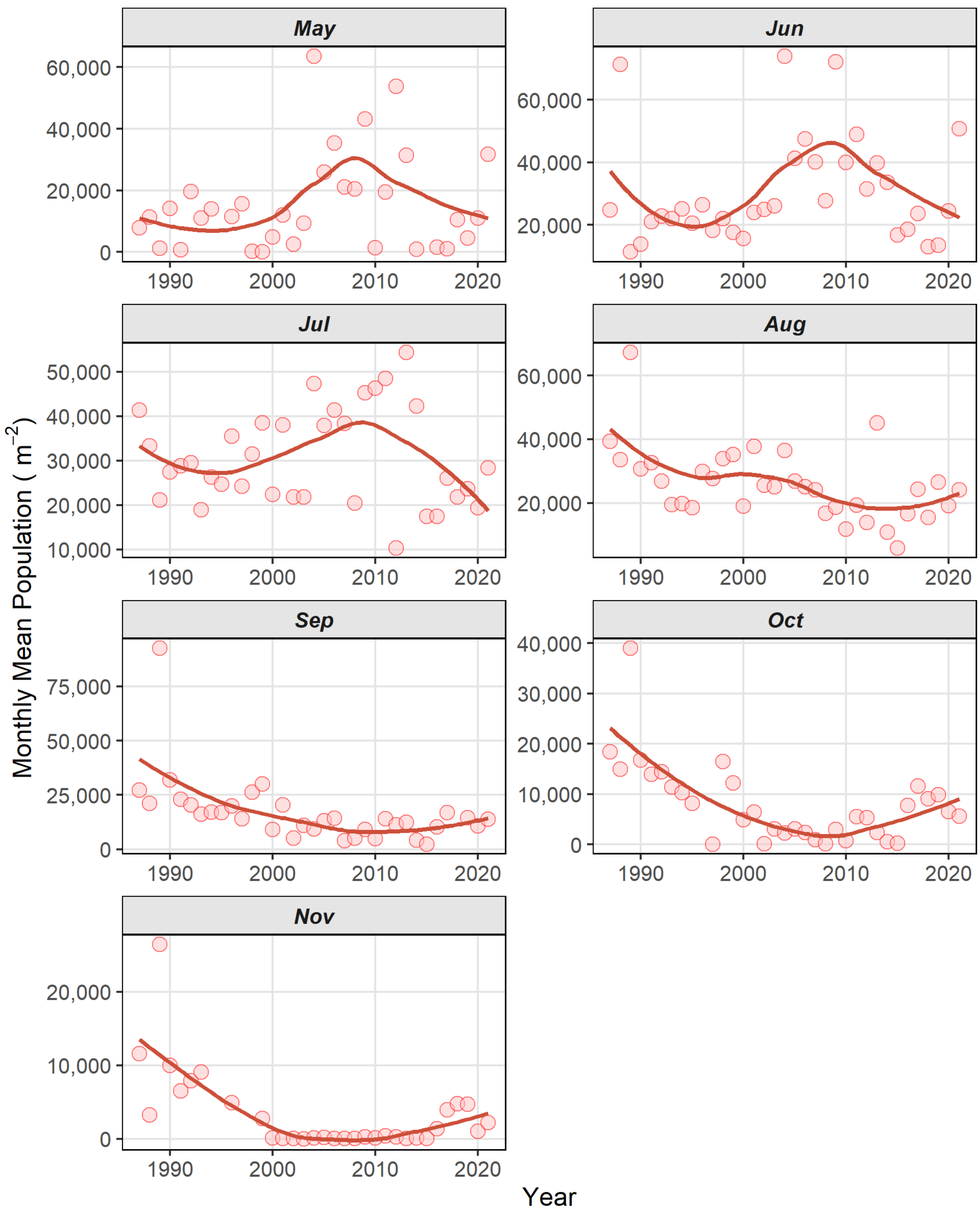


Figure 41. Monthly Trends between Mean Lake-wide Adult *Artemia* Population and Monitoring Years

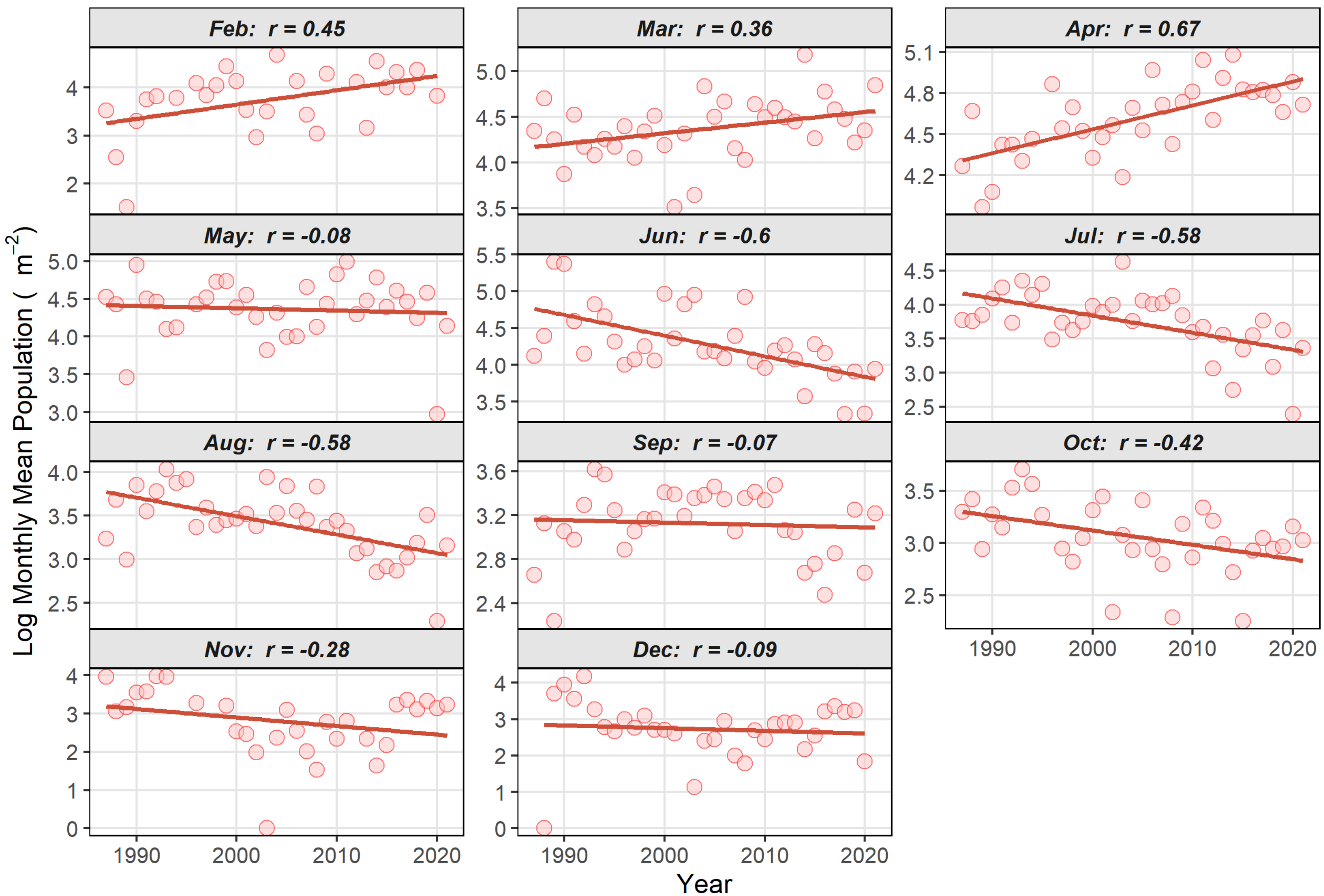


Figure 42. Monthly Trends between Mean Lake-wide Instar Artemia Population and Monitoring Years

Table 3.18. Monthly Trends between Mean Lake-wide Instar and Adult *Artemia* Population and Monitoring Years

Artemia	Month	r	p-value	r^2
Instar*	Feb	0.46	0.0150	0.18
	Mar	0.34	0.0429	0.09
	Apr	0.67	0.0000	0.43
	May	-0.05	0.7655	-0.03
	Jun	-0.60	0.0001	0.34
	Jul	-0.58	0.0002	0.32
	Aug	-0.58	0.0003	0.31
	Sep	-0.07	0.7051	-0.03
	Oct	-0.43	0.0112	0.16
	Nov	-0.28	0.1322	0.04
	Dec	-0.12	0.5008	-0.02
Adult	May	0.19	0.2781	0.01
	Jun	0.12	0.4976	-0.02
	Jul	-0.03	0.8624	-0.03
	Aug	-0.50	0.0020	0.23
	Sep	-0.52	0.0017	0.24
	Oct	-0.54	0.0010	0.27
	Nov	-0.56	0.0011	0.29

* Population means were transformed using base 10 logarithm for instars.

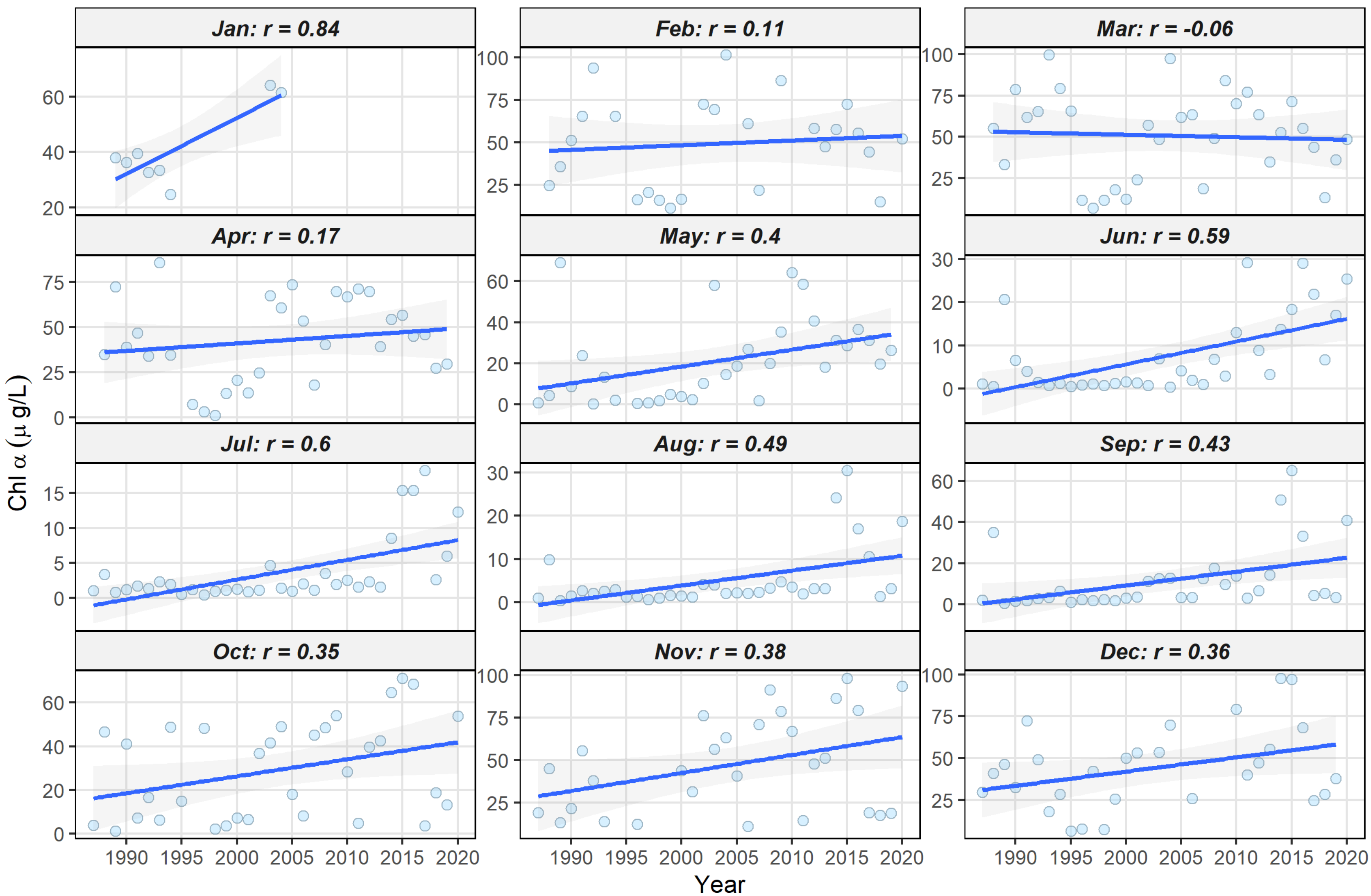


Figure 43. Relationships between Monthly Chlorophyll a Concentration and Monitoring Years

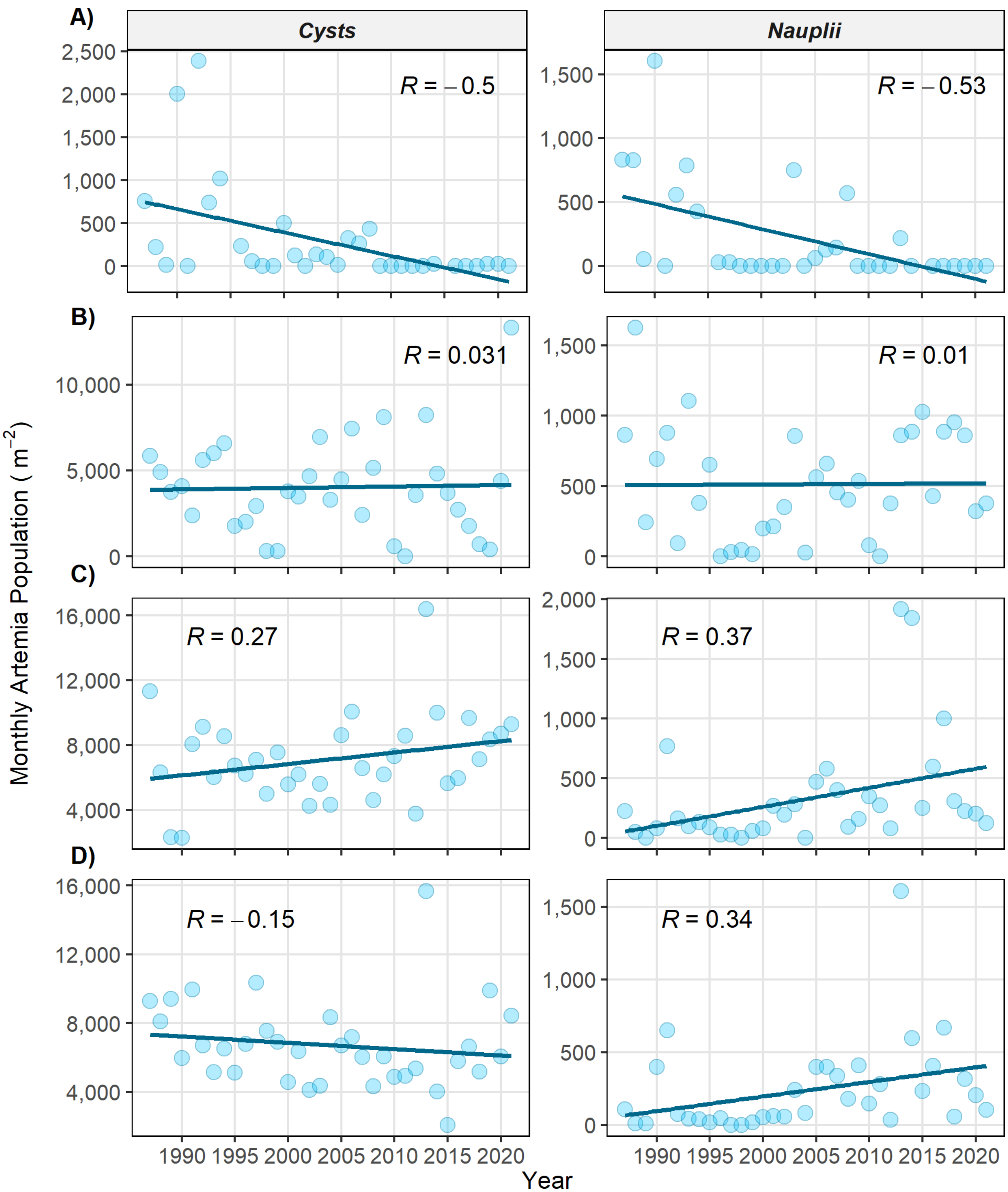
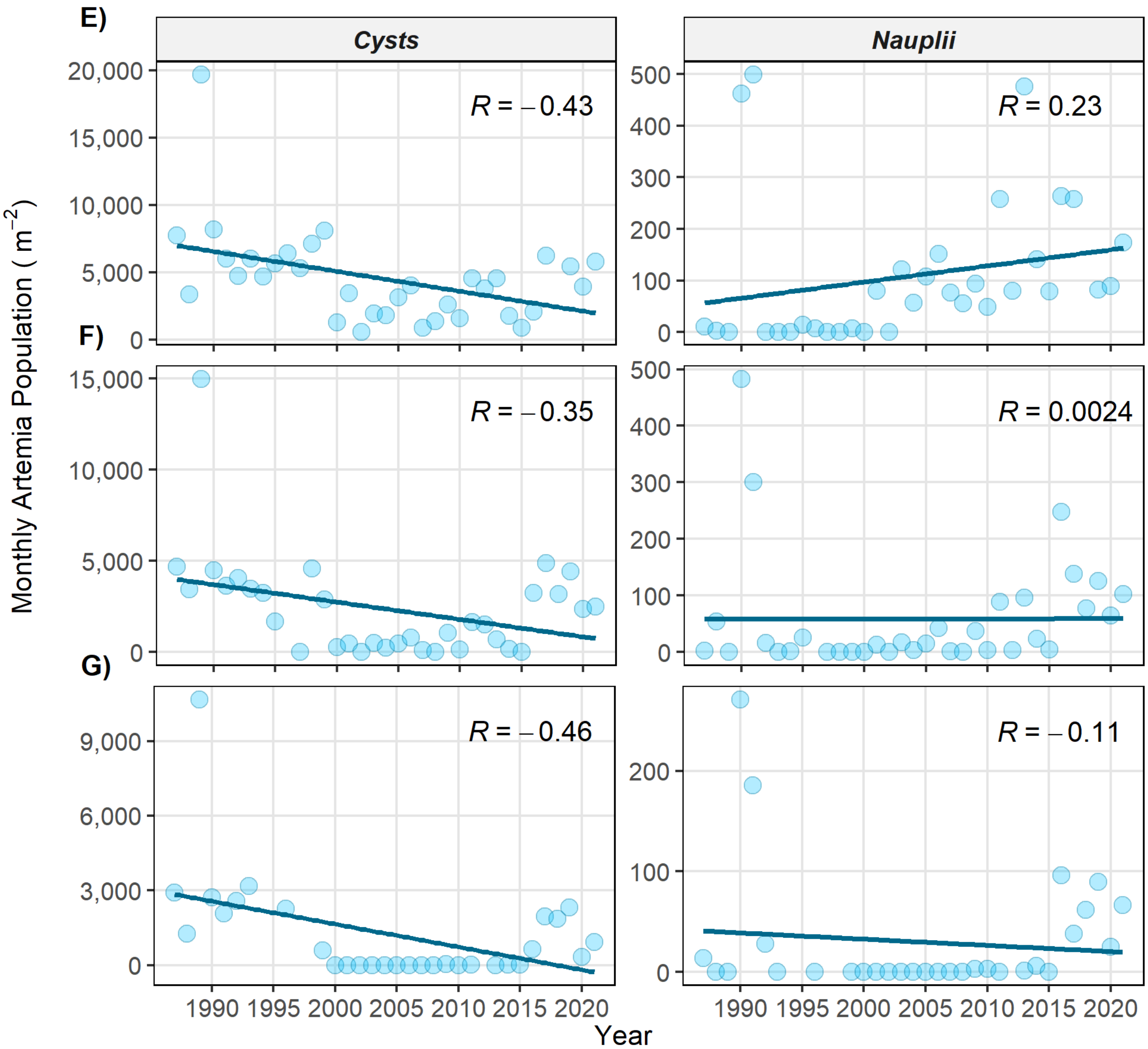


Figure 44. The relationship of Monthly Artemia Females to Cysts and Nauplii for A) May, B) June, C) July, D) August, E) September, F) October, and G) November

Figure 44 Continued



3.2.4 Limnology Discussion

2021 Condition

The 2021 monitoring year experienced an *Artemia* population peak following the breakdown of meromixis at the end of 2020. The mean *Artemia* population increased almost 100% from 12,991 m^{-2} in 2020 to 23,177 m^{-2} in 2021 as ammonium which had accumulated in the hypolimnion was released to the epilimnion as Mono Lake completely turned over at the end of 2020. There have been five recorded post meromictic *Artemia* population peaks: 1989. 2004, 2009, 2013, and 2021 and the 2021 population peak was the lowest among these five peaks. The *Artemia* population centroid was 195 days, falling below 200 days for the first time since 2015. The peak monthly instar abundance of 70,022 m^{-2} in March was the second highest recorded. The March instar peak follows the temporal shift of population peaks to earlier months, which has been noted before. The peak adult *Artemia* abundance of 50,731 m^{-2} was the fourth highest recorded for June. In 2021 higher instar abundance translated to higher adult abundance even though this relationship does not always hold.

Clarity of the lake remained below 1 m for the second year in row. The drop in hypolimnetic salinity level following the almost record-breaking 2017 runoff was reversed in 2021 due to very low runoff of 52% from two major tributaries. Salinity as of August 2021 was 85.5g/L and 82.2g/L in the epilimnion and hypolimnion, respectively, and most likely to rise in 2022 due to dismal snowpack so far in the winter of 2021-22. Epilimnetic water temperature in 2021 was found mostly below the LTA of respective month while the hypolimnetic water temperature was found mostly above the LTA of respective month mainly due to the holomixis at the end of 2020.

Long-Term Trend

The *Artemia* population is showing signs of decline, and this trend is much stronger when looking at post meromictic population peaks. The 2021 peak was the lowest among five such peaks in spite of much higher ammonium accumulation than the previous two peaks (2009 and 2013). Overall chlorophyll *a* concentration is rising, especially since the second meromictic period of 1995-2002, but also for summer months. The trend appears to have accelerated after the most recent five-year drought (2012-2016). Chlorophyll *a* concentration and maximum Secchi readings in summer months appear to have shifted starting sometime between 2005 and 2010 setting up a pattern of increasing chlorophyll *a* concentration combined with declining lake transparency. Higher algae abundance in summer months is not translating to higher *Artemia* populations, but is resulting in poor lake transparency. Salinity is lower relative to conditions found prior to the second meromixis (1995-2002), remaining mostly below 85 g/L, except toward the tail end of the five-year drought during when salinity approached 90 g/L.

Ambient temperature is rising, and water temperature below 30 m shows a warming trend while the upper mixing layer showing the opposite trend.

There has been a clear temporal shift in peak abundance of *Artemia* instars but a trend is not so clear for *Artemia* adults as the adult temporal pattern forms a unimodal, non-linear pattern when regressed against monitoring years, with a peak occurring somewhere between 2005 and 2010. Food sources are not particularly more abundant and spring water temperature has not been warmer; yet, instar nauplii are hatching earlier. A rising trend of ovoviviparous females in summer months may be enough to broaden a population peak during years with very low *Artemia* abundance, leading to a strong linear negative trend of centroid days with respect to monitoring years. The rising trend of ovoviviparous females, however, does not appear to have translated into higher instar abundance in summer months in spite of abundant food sources; consequently, the significant development of a second and third generation has not materialized in recent years.

Future Condition

Future limnological condition of Mono Lake will largely depend on future runoff conditions. Since the adaptation of Decision 1631, Mono Basin export by LADWP has been limited to 16,000 acre-feet for lake levels between 6,380 ft and 6391 ft, and 4,500 acre-feet for lake levels between 6,377 ft and 6,380 ft, with no export allowed when the lake level falls below 6,377 ft. The lake level is at 6,379.6ft as of December 2021 and expected to decline again in 2022 due to dismal snowpack as of mid-March 2022, resulting in very limited water diversion from the Mono Basin. The *Artemia* population has been declining, and the five-year drought (2012-2016) appeared to have had a profound impact on *Artemia* population and Mono Lake. In 2015, not only was the lowest adult population mean recorded, but for the first time recorded, lake transparency remained less than 1 m throughout the year. Algal food sources are available throughout the summer, but are remaining unconsumed by *Artemia*. In 2021, despite an increase in the *Artemia* population, the lake transparency remained below 1 m all year round. Mono Lake is currently less saline relative to the period between the late 1970s to early 1990s, during which the lake level hit its lowest on record, and salinity approached 100 g/L.

It should be noted that a much higher *Artemia* adult population was recorded in 1981, 1982, 1987, and 1988 under much higher salinity conditions, and the population mean in 1981 and 1982 was much higher than the last four post meromictic population peaks (2004, 2009, 2013, and 2021). *Artemia* population means averaged 21,700 m^{-2} between 1979 and 1988 (prior to the first meromictic population peak in 1989) compared to 15,286 m^{-2} for all other non-population peak years, and 12,234 m^{-2} between 2014 and 2020 (between two population peaks in 2013 and 2021). Because of its high salinity tolerance, the *Artemia* population in Mono Lake continued to thrive even when salinity exceeded 90 g/L and approached 100 g/L. Even though current lake salinities are lower prior to implementation of Decision 1631, *Artemia* populations are showing a long-term trend of decline, and the severe drought further reduced the already declining *Artemia* population. If no additional significant precipitation occurs in 2022, the Mono Basin may experience the driest three-year period on record. Summer and winter temperatures have been rising, and the rising trend appears strengthening in recent years. It is unclear the long-term impacts on the *Artemia* population of intensifying hydrological drought combined with warmer temperatures.

3.3 Vegetation Status in Lake-Fringing Wetlands

Permanent vegetation transects are established in lake-fringing wetlands to monitor vegetation cover and composition in response changing lake levels. Wetland monitoring transects were initially established in 1999 at accessible sites around the perimeter of Mono Lake that supported alkali meadow or wetland vegetation. The three wetland monitoring sites are: Warm Springs, Simons Springs and the DeChambeau Embayment (Figure 45).

Formerly absent from the Mono Basin, feral horses from the Montgomery Pass Herd were first noted at Mono Lake around 2012, when 40-50 were observed on the east side of the lake at Simons Spring (Wild Horse Meeting Management Notes 17 Aug 2020). Since that time, horse herds have increased in size, with 400-500 seen regularly at Warm Springs during waterfowl surveys (D. House, pers. obs.). Up through 2021, horses or signs of horse activity have been observed from the east side of Mono Lake, and all along the south shore to South Tufa.

In 2021, the Waterfowl Director recommended to LADWP management that the vegetation monitoring be conducted in order to document conditions, and potential impacts from the feral horse grazing. Monitoring was conducted at the lake-fringing wetland sites in 2021.

3.3.1 Lake-fringing Wetland Monitoring Sites

Warm Springs

The Warm Springs shoreline area is on the east side of Mono Lake. There are several springs in the Warm Springs area, most of which are brackish. Alkaline meadow dominates the area, however emergent wetland vegetation occurs in the immediate vicinity of existing springs.

Simons Springs

The Simons Springs area is along the southeastern shore of Mono Lake. Numerous freshwater springs occur in this area, supporting vast stands of freshwater wetlands. The vegetation transects are located in the vicinity of the Simons Spring Fault line, a feature contributing to the existence of springs in the area.

DeChambeau Embayment

The DeChambeau Embayment is along the north shoreline of Mono Lake, just east of Black Point. The Dechambeau Embayment area supports springs that vary from fresh to saline.

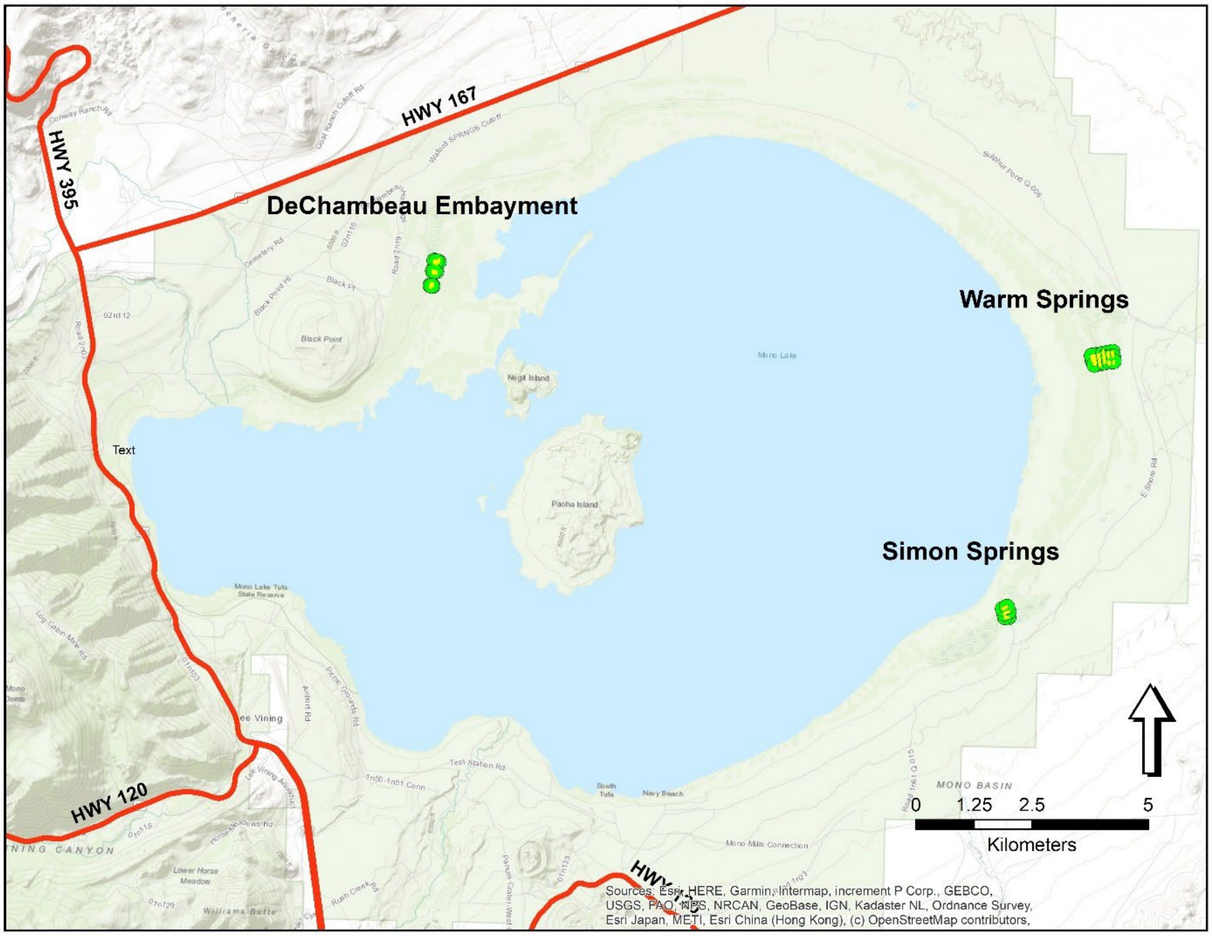


Figure 45. Wetland Vegetation Monitoring Sites at Mono Lake

3.3.2 Lake-fringing Wetland Monitoring Methodologies

Warm Springs

At Warm Springs, a total of 18 permanent transects were established in 1999 (Figure 46). The starting points for the transects were arranged into three east-to-west and six north-to-south parallel lines. The six north-to-south lines run parallel to the shoreline in this area to allow for the evaluation of change in vegetation as a function of distance to the shoreline. When established in 1999, the starting points for each of the three east-west oriented lines extended from the 1999 lake elevation of 6384 feet, to approximately the 6392-foot level. In 2021, the shoreline was over 300 meters downgradient of these start points due to a lower lake elevation.

At each of the 18 starting locations, 50-meter long sampling transects were established parallel to the lake shore. Sampling transects ran either north or south from the permanent transect, the direction was chosen randomly in 1999 and has remained the same since. Transect names, positions relative to the shoreline, and GPS start points can found in Table 3-20.

Line-point sampling was conducted along each of the 18 sampling transects using the point intercept method (Mueller-Dombois and Ellenberg 1974). A visual assessment of the first hit of a plant species or cover class (in the absence of live cover) was recorded at one-meter intervals along a sampling tape. Plant species were identified to the lowest taxa possible. Cover classes used were standing mulch (dead emergent plants or dead shrubs), litter, bare ground, rock, and water. Caution was taken to minimize disturbance to existing vegetation along the permanent transects.

Photos were taken at each end of the sampling transect, facing back towards the opposite end of the 50-meter transect. In 2021, sampling at Warm Springs was conducted August 24-25.

Table 3.19. Warm Springs Vegetation Transects

Shoreline Area	Transect Name	Position Relative to Shoreline (A = closest)	Transect Length	Start Location (NAD83)		Transect Direction
				Easting	Northing	
Warm Springs	WS_N_250S	A	50 m	332305	4210772	South
	WS_M_250N		50 m	332320	4210672	North
	WS_S_225N		50 m	332358	4210577	North
	WS_N_200S	B	50 m	332355	4210781	South
	WS_M_200N		50 m	332369	4210681	North
	WS_S_200N		50 m	332383	4210582	North
	WS_N_100S	C	50 m	332453	4210800	South
	WS_M_100S		50 m	332467	4210701	South
	WS_S_100S		50 m	332481	4210602	South
	WS_N_ON	D	50 m	332551	4210819	North
	WS_M_ON		50 m	332565	4210721	North
	WS_S_ON		50 m	332578	4210622	North
	WS_N_100SP	E	50 m	332649	4210839	South
	WS_M_100NP		50 m	332663	4210740	North
	WS_S_100SP		50 m	332676	4210642	South
	WS_N_200SP	F	50 m	332747	4210857	South
	WS_M_200NP		50 m	332760	4210760	North
	WS_S_200SP		50 m	332776	4210649	South

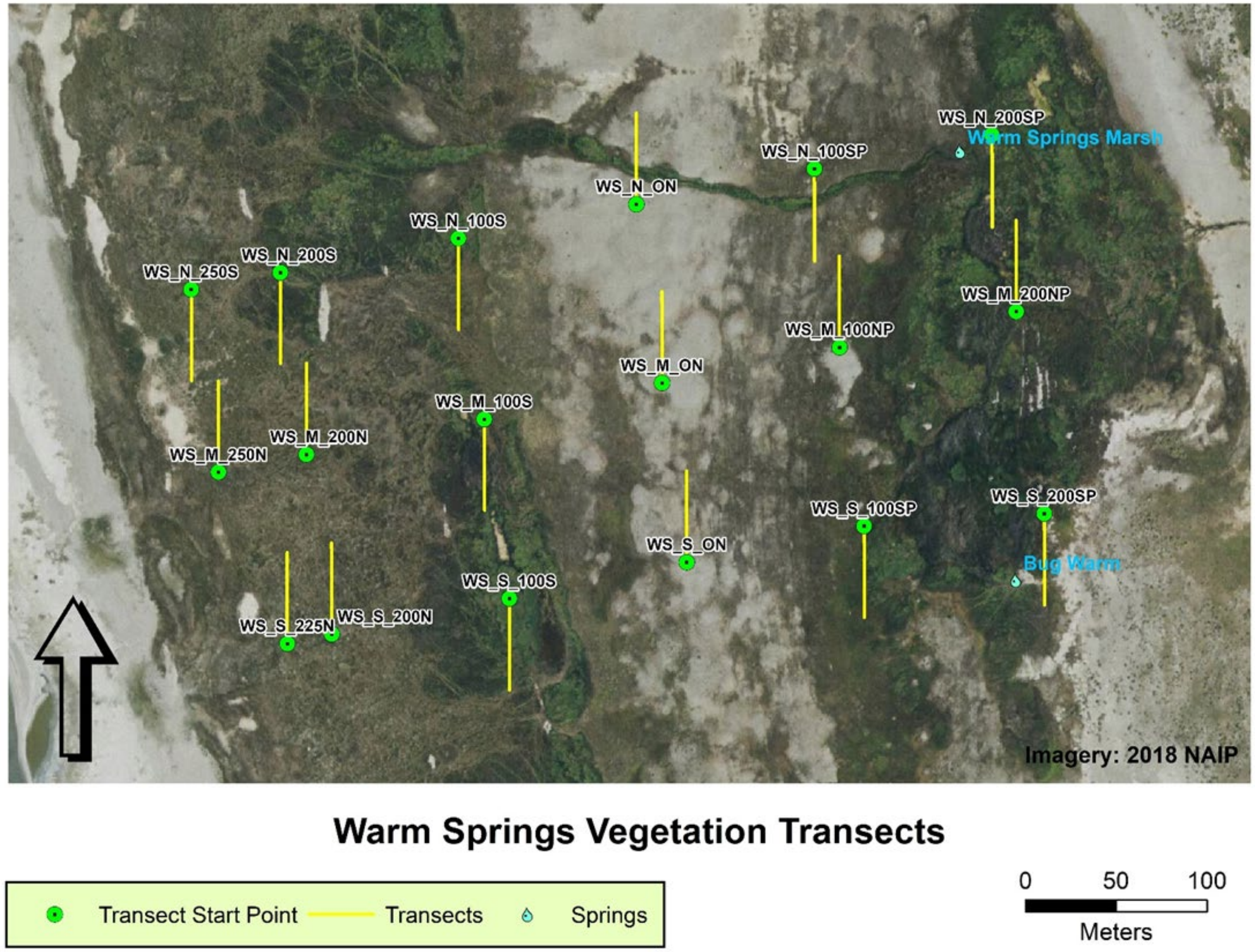


Figure 46. Warm Springs Vegetation Sampling Transects

Simons Springs

At Simons Springs, three permanent transects were established in 1999 (Figure 47). At this site, t-posts were not installed in order to minimize the number of permanent markers visible at this popular tufa viewing site. The three transects are oriented roughly perpendicular to the Simons Springs fault line. Transects vary in length as SAM_1 and SAM_2 are 100 meters long and SAM_3 is 75 meters long. Transect names, positions relative to the shoreline, and GPS start and end points can be found in Table 3-21.

Line-point sampling was conducted along each of the 18 sampling transects using the point intercept method (Mueller-Dombois and Ellenberg 1974). A visual assessment of the first hit of a plant species or cover class (in the absence of live cover) was recorded at one-meter intervals along a sampling tape. Plant species were identified to the lowest taxa possible. Cover classes used were standing mulch (dead emergent plants or dead shrubs), litter, bare ground, rock, and water. Caution was taken to minimize disturbance to existing vegetation along the permanent transects. Photos were taken at each end of the sampling transect, facing back towards the opposite end of each transect. In 2021, sampling at Simon Springs was conducted August 25.

Table 3.20. Simons Springs Vegetation Transects

Shoreline Area	Transect Name	Position Relative to Shoreline (A = closest)	Transect Length	Starting (NAD83)		End (NAD83)	
				Easting	Northing	Easting	Northing
Simons Springs	SAM_1	A	100	330388	4205320	330479	4205361
	SAM_2	B	100	330420	4205217	330510	4205262
	SAM_3	C	75	330507	4205133	330433	4205138

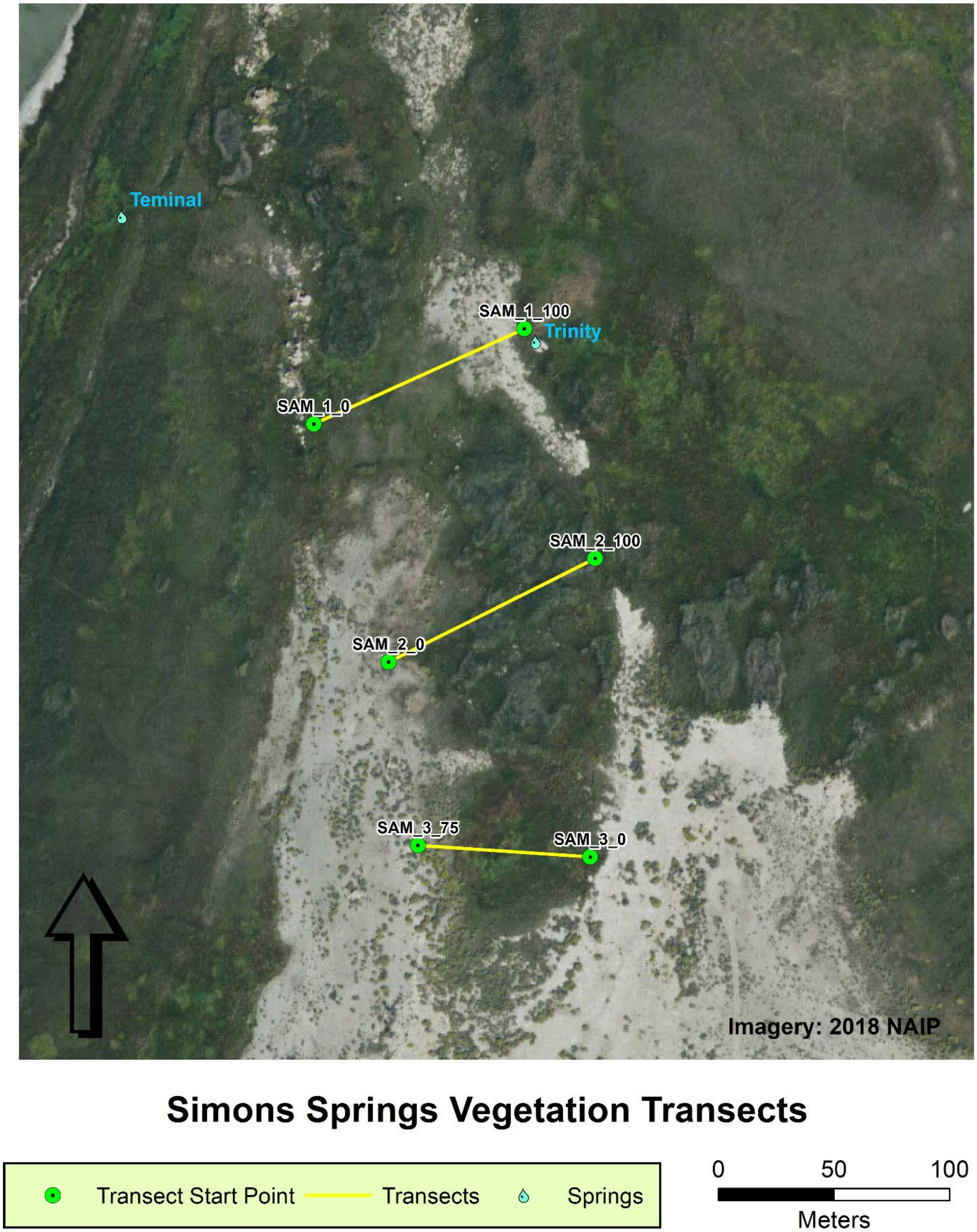


Figure 47. Simons Springs Vegetation Sampling Transects

DeChambeau Embayment

At DeChambeau Embayment, a total of nine permanent transects were established in 1999 (Figure 48). The starting points for the transects were arranged into three east-to-west and three north-to-south roughly parallel lines. The three north-to-south lines run parallel to the shoreline in this area to allow for the evaluation of change in vegetation as a function of distance to the shoreline. When established in 1999, the starting points for each of the three east-west oriented lines were approximately 100 meters from the shoreline. In 2021, the shoreline was over 700 meters downgradient of these start points due to a lower lake elevation.

At each of the 9 starting locations, 50-meter long sampling transects were established parallel to the lake shore. Sampling transects ran either north or south from the permanent transect, the direction chosen randomly in 1999 and has remained the same since. Transect names, positions relative to the shoreline, and GPS start points can found in Table 3-22.

Line-point sampling was conducted along each of the 18 sampling transects using the point intercept method (Mueller-Dombois and Ellenberg 1974). A visual assessment of the first hit of a plant species or cover class (in the absence of live cover) was recorded at one-meter intervals along a sampling tape. Plant species were identified to the lowest taxa possible. Cover classes used were standing mulch (dead emergent plants or dead shrubs), litter, bare ground, rock, and water. Caution was taken to minimize disturbance to existing vegetation along the permanent transects.

Photos were taken at each end of the sampling transect, facing back towards the opposite end of the 50-meter transect. In 2021, sampling at DeChambeau Embayment was conducted September 9-10.

Table 3.21. DeChambeau Embayment Vegetation Transects

Shoreline Area	Transect Name	Position Relative to Shoreline (A = closest)	Transect Length	Starting Location (NAD83)		Transect Direction
				Easting	Northing	
Dechambeau Embayment	DE_N_0S	A	50 m	318152	4212819	South
	DE_M_0S		50 m	318118	4212602	South
	DE_S_0S		50 m	318064	4212288	South
	DE_N_50S	B	50 m	318197	4212803	South
	DE_M_37.5S		50 m	318155	4212592	South
	DE_S_25N		50 m	318088	4212280	North
	DE_N_100N	C	50 m	318248	4212793	North
	DE_M_75S		50 m	318191	4212585	South
	DE_S_50N		50 m	318112	4212275	North

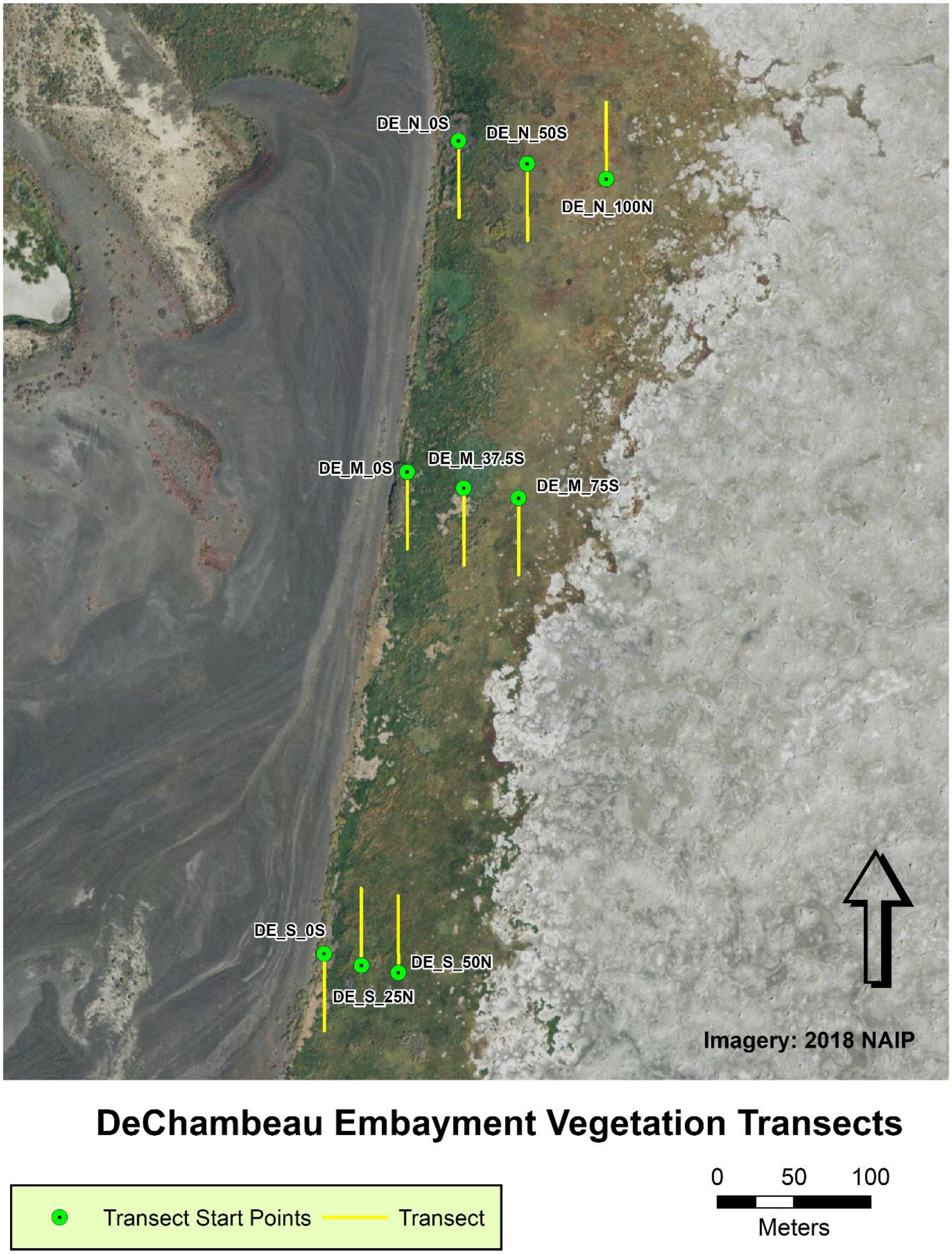


Figure 48. DeChambeau Embayment Vegetation Sampling Transects

3.3.3 Lake-fringing Wetland Monitoring Data Summary and Analysis

Each sample point along the transect, or “hit” was categorized as supporting live cover or not (e.g. standing mulch, litter, bare ground, rock, and water). Plant species were further categorized into forb, graminoid or shrub based on their growth habit. Forbs are annual or perennial herbaceous flowing plants with nonwoody stems. Graminoids are annual or perennial grasses, sedges and rushes. Shrubs are perennial plants with woody stems.

The total hits were summed by species, cover class and transect, and then divided by the total sample points per transect to determine the percent cover of species/cover class. Total forb, graminoid, shrub and live cover were summed by transect.

Photos of the transects were compiled. Only the photo taken from the start location of each transect is included in the report.

Trends in live cover were evaluated for each monitoring site by summarizing the percent cover by cover class for each monitoring year (1999, 2004, 2005, 2009, 2014, and 2021). The cover of litter and standing mulch, and bare ground and rock were combined for this analysis. The 2021 values were compared to the average +/- standard error for five previous monitoring years for which there are data. Species composition was evaluated by summing the live cover of the dominant species by site and year. Species were selected for evaluation if they comprised >5% of the total cover from 1999-2014, or in 2021.

3.3.4 Lake-fringing Wetland Monitoring Results

Warm Springs

Total live cover at Warm Springs averaged 40.3% across the 18 transects (Table 3-23). The plant community at Warm Springs was composed primarily of graminoids. Of the eight species found at Warm Springs, Nevada bulrush (*Scirpus nevadensis*) and Chairmakers bulrush (*Schoenoplectus americanus*) were dominant, comprising 90% of the live cover. Bare ground averaged 27% and litter 26.9%.

At the Warm Springs site, several of the t-posts were on the ground, likely a result of feral horse activity. The photos (Figures 58-75) show denuded areas due to heavy grazing from feral horses, and numerous piles of horse dung present.

The live cover at Warm Springs in 2021 (40.3%) was at its lowest since monitoring was initiated in 1999. The percent of live cover in 2021 was significantly lower than the average of all other monitoring years combined (Figure 67). The percent of litter/standing dead was slightly above the long-term mean, but a five-fold increase in bare ground was observed. The decrease in live cover at Warm Springs can largely be attributed to a decline in *Scirpus nevadensis* (Figure 68).

Table 3.22. Warm Springs 2021 Results

Species or Cover Class	Growth Habit	Transect Position Relative to Shore																		Average Cover
		A			B			C			D			E			F			
		WS_N_250S	WS_M_250N	WS_S_225N	WS_N_200S	WS_M_200N	WS_S_200N	WS_N_100S	WS_M_100S	WS_S_100S	WS_N_ON	WS_M_ON	WS_S_ON	WS_N_100SP	WS_M_100NP	WS_S_100SP	WS_N_200SP	WS_M_200NP	WS_S_200SP	
Nitrophila occidentalis	Forb														6%					0.3%
Distichlis spicata	Graminoid										8%	4%	8%	6%	2%					1.6%
Juncus arcticus	Graminoid								4%										20%	1.3%
Schoenoplectus acutus	Graminoid								14%								4%			1.0%
Schoenoplectus americanus	Graminoid	2%			34%			24%	26%	36%	2%		6%	6%		22%	40%	74%	28%	16.7%
Scirpus nevadensis	Graminoid	22%	34%	42%	12%	36%	40%	42%			2%	12%	16%	12%	40%	40%		4%		19.7%
Triglochin concinna	Graminoid																		2%	0.1%
Ericameria nauseosa	Shrub										8%									0.4%
Forb Cover															**6%**					**0.3%**
Graminoid Cover		**24%**	**34%**	**42%**	**46%**	**36%**	**40%**	**66%**	**44%**	**36%**	**12%**	**16%**	**30%**	**24%**	**42%**	**62%**	**44%**	**78%**	**50%**	**40.3%**
Shrub Cover											**8%**									**0.4%**
Total Live Cover		**24%**	**34%**	**42%**	**46%**	**36%**	**40%**	**66%**	**44%**	**36%**	**12%**	**16%**	**30%**	**24%**	**42%**	**62%**	**44%**	**78%**	**50%**	**40.3%**
Standing mulch									44%								24%		2%	3.9%
Litter		20%	42%	40%	18%	58%	46%	34%	10%	14%	10%	10%	28%	64%	44%	12%	8%	12%	14%	26.9%
Bare Ground		56%	24%	18%	36%	6%	14%		2%	46%	68%	74%	42%	12%	8%	22%	20%	6%	32%	27.0%
Rock																			2%	0.1%
Water										4%	2%					4%	4%	4%		1.0%

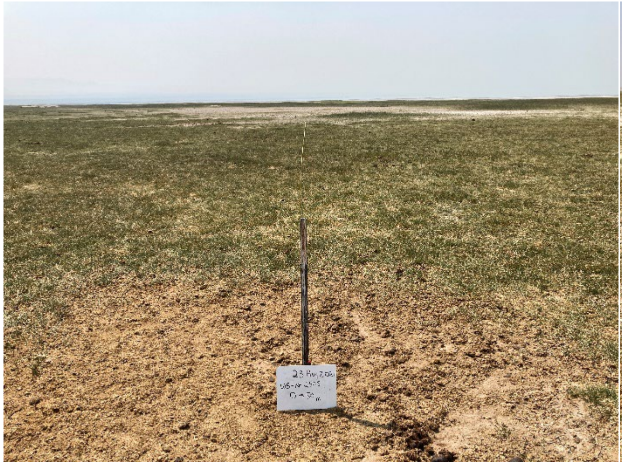


Figure 50. WS_N_250S

Figure 49. WS_M_250N

Figure 52. WS_S_225N

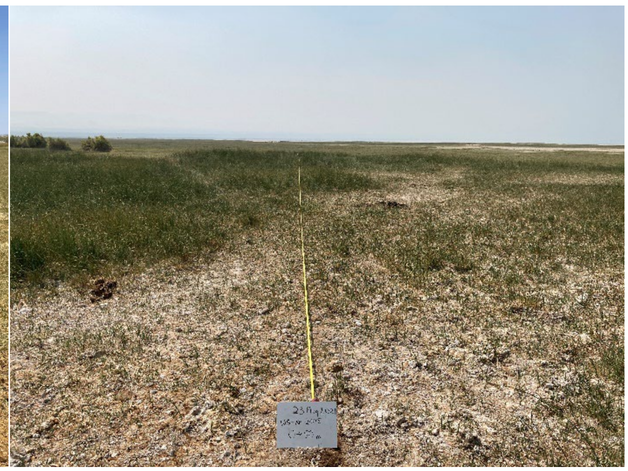


Figure 51. WS_N_200S

Figure 54. WS_M_200N

Figure 53. WS_S_200N

Figure 56. WS_N_100S

Figure 55. WS_M_100S

Figure 58. WS_S_100S

Figure 57. WS_N_ON

Figure 60. WS_M_ON

Figure 59. WS_S_ON

Figure 62. WS_N_100SP

Figure 61. WS_M_100NP

Figure 64. WS_S_100SP

Figure 63. WS_N_200SP

Figure 66. WS_N_200NP

Figure 65. WS_S_200SP

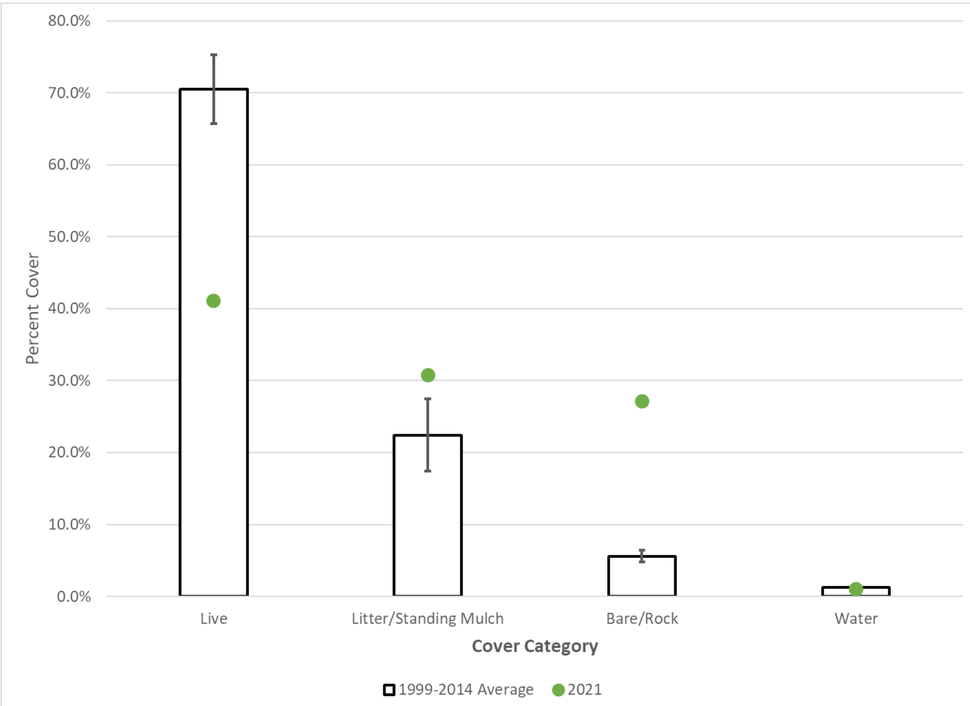


Figure 67. 2021 Warms Springs cover vs. 1999-2014 monitoring year averages

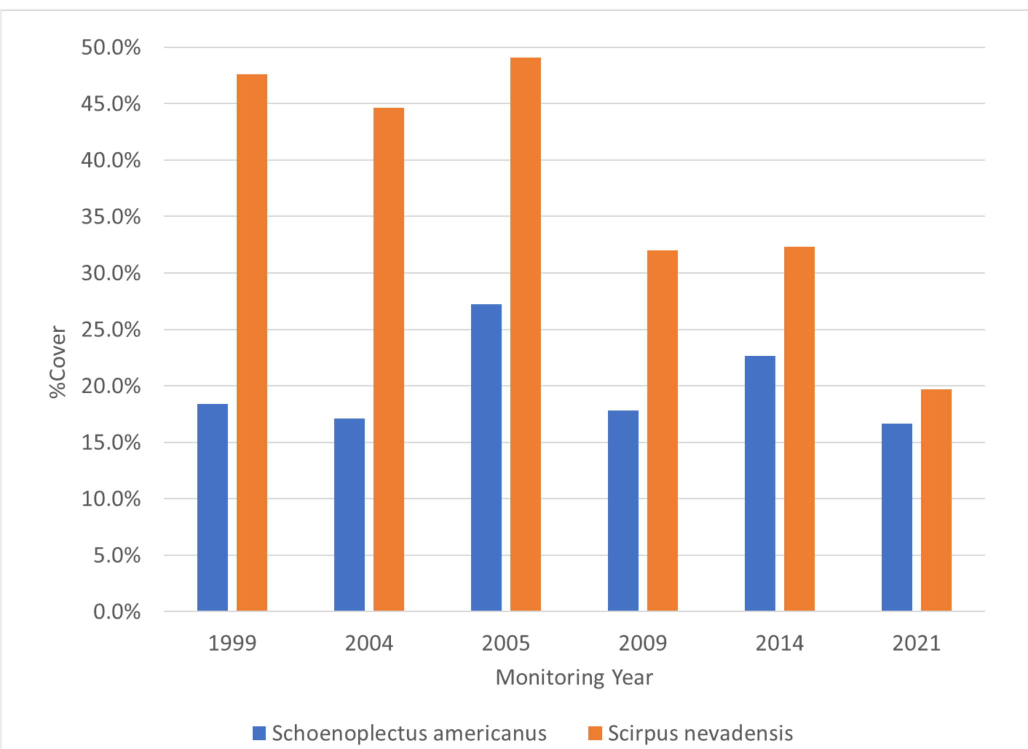


Figure 68. Cover of dominant species at Warms Springs by monitoring year

Simons Springs

Total live cover at Simons Springs averaged 83.3% across the 3 transects (Table 3-23). The plant community at Simons Springs was more diverse than Warm Springs, but similarly, composed primarily of graminoids. Of the fifteen species found at Simons Springs, an unidentified sedge (*Carex* sp.) and Arctic rush (*Juncus arcticus*) were most abundant, comprising 55% of the live cover. Bare ground averaged 5% and litter 8%.

The Simons Springs transects continue to support a lush wet meadow community (Figures 78-80). Only light to moderate horse activity was noted in the vicinity, but the transects did not appear grazed or trampled.

Live cover at Simons Springs in 2021 was slightly above the 1999-2014 mean, and litter/standing mulch slightly below (Figure 72). The amount of bare ground and water did not differ from the average of the previous years. With the exception of a decline in Juncus arcticus cover from a high observed in 2005, there is no clear-cut pattern in the cover of dominant species observed at Simons Springs (Figure 73). Because this site is dominated by diverse and dense stands of graminoids, the inherent nature of line point sampling could lead to minor fluctuations in species cover values recorded at the Simons Spring site.

Table 3.23. Simons Springs 2021 Results

Species or Cover Class	Growth Habit	Position Relative to Shore A SAM_1	B SAM_2	C SAM_3	Average Cover
Epilobium ciliatum	Forb			1%	0.4%
Senecio hydrophilus	Forb			1%	0.4%
Solidago canadensis	Forb	4%		1%	1.8%
Agrostis stolonifera	Graminoid	11%			3.7%
Carex sp.	Graminoid	7%	19%	51%	25.6%
Distichlis spicata	Graminoid	2%	1%	4%	2.3%
Eleocharis sp.	Graminoid	2%			0.7%
Hordeum jubatum	Graminoid		1%		0.3%
Juncus arcticus	Graminoid	21%	27%	12%	20.0%
Muhlenbergia asperifolia	Graminoid	8%	4%	5%	5.8%
Schoenoplectus acutus	Graminoid		21%		7.0%
Schoenoplectus americanus	Graminoid	1%	1%	9%	3.8%
Scirpus nevadensis	Graminoid	14%			4.7%
Typha latifolia	Graminoid		12%	1%	4.4%
Ericameria nauseosa	Shrub	2%		5%	2.4%
Forb Cover		4%	0%	4%	2.7%
Graminoid Cover		66%	86%	83%	78.2%
Shrub Cover		2%	0%	5%	2.4%
Total Live Cover		72%	86%	92%	83.3%
Standing mulch		3%	6%	1%	3%
Litter		10%	7%	7%	8%
Bare Ground		15%			5%
Water			1%		0%

Figure 69. SAM_1

Figure 70. SAM_2

Figure 71. SAM_3

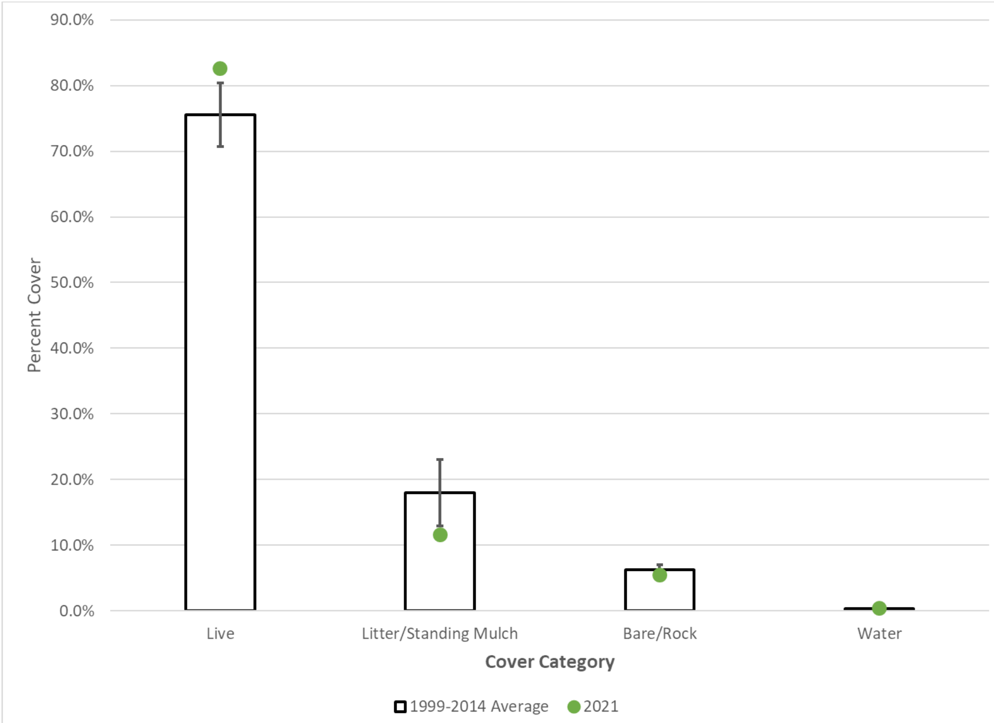


Figure 72. 2021 Simons Springs cover vs. 1999-2014 monitoring year averages

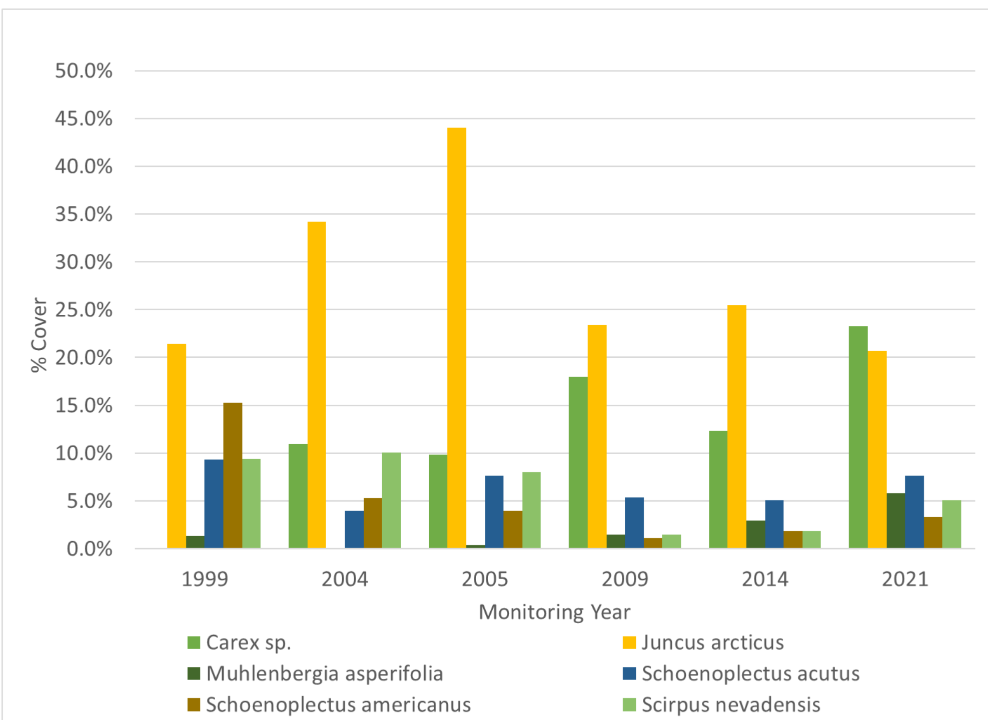


Figure 73. Cover of dominant species at Simons Springs by monitoring year

DeChambeau Embayment

Total live cover at DeChambeau Embayment averaged 82.9% across the 9 transects (Table 3-25). The plant community at DeChambeau Embayment was more diverse than the other two sites, however, it was dominated by weedy forbs, and secondarily graminoids. Of the eighteen species found at DeChambeau Embayment, four nonnative forbs: *Atriplex micrantha*, *Atriplex prostrata*, *Bassia hyssopifolia* and *Salsola tragus* comprised 38.4% of the cover. No bare ground was encountered, and litter averaged 15%.

The DeChambeau Embayment site was very densely vegetated in 2021 (Figures 83-91). The site seemed quite dry and previous stands of emergent vegetation were dry and decadent (Figure 74, Figure 76). No horse activity was or has been noted in the vicinity.

Live cover was slightly below previous years, and litter/standing mulch slightly higher (Figure 83). A shift in species composition was observed at DeChambeau Embayment. The cover of Chairmakers bulrush (*Schoenoplectus americanus*) was substantially lower than all previous years, likely do the drying that had occurred (Figure 84). The nonnative weedy species *Atriplex micrantha* and *Bassia hyssopifolia* were the most abundant species. This was the first year *Atriplex micrantha* had been recorded.

Table 3.24. DeChambeau Embayment 2021 Results

		Position Relative to Shore									
		A			B			C			
Species or Cover Class	Growth Habit	DE_N_0S	DE_M_0S	DE_S_0S	DE_N_50S	DE_M_37.5S	DE_S_25N	DE_N_100N	DE_M_75S	DE_S_50N	Average Cover
Atriplex micrantha	Forb	56%	2%	26%	4%	4%	36%		12%	30%	18.9%
Atriplex prostrata	Forb		22%								2.4%
Bassia hyssopifolia	Forb			32%	4%	50%	48%		8%	10%	16.9%
Chenopodium album	Forb							2%			0.2%
Salsola tragus	Forb		2%								0.2%
Symphyotrichum frondosum	Forb				2%						0.2%
Agrostris stolonifera	Graminoid								8%	2%	1.1%
Distichlis spicata	Graminoid		2%		20%	8%		36%	12%		8.7%
Hordeum jubatum	Graminoid	2%			24%	10%			24%	2%	6.9%
Juncus arcticus	Graminoid				18%	2%					2.2%
Leymus triticoides	Graminoid					4%					0.4%
Schedonorus pratensis	Graminoid				4%			18%			2.4%
Schoenoplectus acutus	Graminoid	14%									1.6%
Schoenoplectus americanus	Graminoid				4%	14%		6%	22%	56%	11.3%
Scirpus nevadensis	Graminoid				4%						0.4%
Typha latifolia	Graminoid	8%	2%	4%							1.6%
Ericameria nauseosa	Shrub			2%							0.2%
Salix exigua	Shrub		64%								7.1%
Forb Cover		56%	26%	58%	10%	54%	84%	2%	20%	40%	38.9%
Graminoid Cover		24%	4%	4%	74%	38%		60%	66%	60%	36.7%
Shrub Cover			64%	2%							7.3%
Total Live Cover		80%	94%	64%	84%	92%	84%	62%	86%	100%	82.9%
Standing mulch		18%									2%
Litter		2%	6%	36%	16%	8%	16%	38%	14%		15%
Bare Ground											0%
Water											0%

Figure 75. DE_N_0S

Figure 74. DE_M_OS

Figure 77. DE_S_OS

Figure 76. DE_N_50S

Figure 79. DE_M_37.5S

Figure 78. DE_S_25N

Figure 81. DE_N_100N

Figure 80. DE_M_75S

Figure 82. DE_S_50N

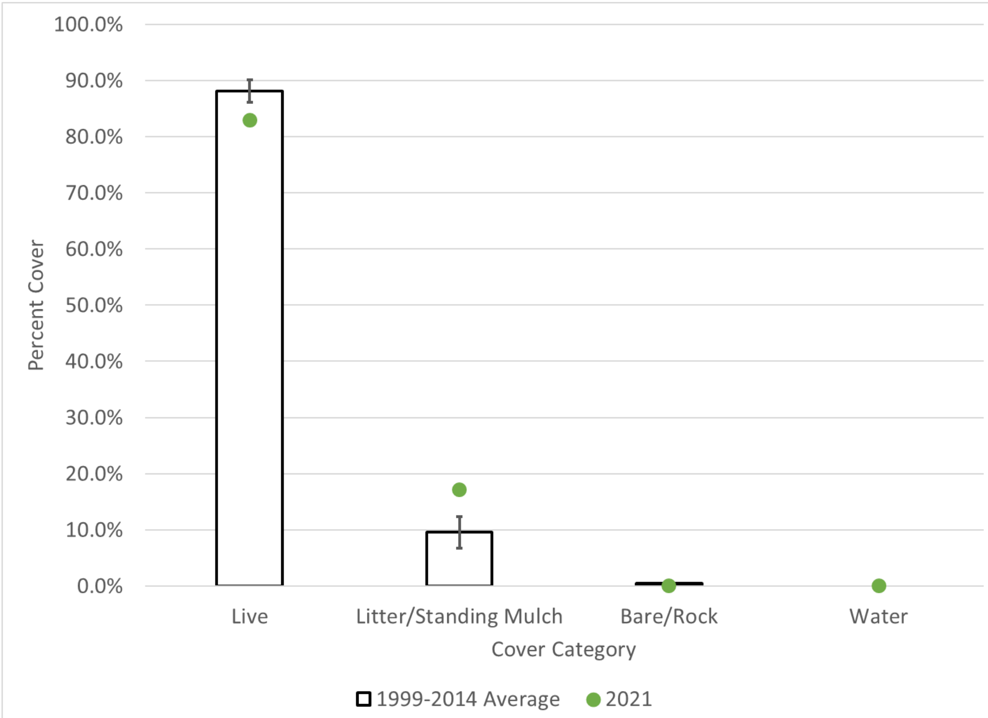


Figure 83. 2021 DeChambeau Embayment cover vs. 1999-2014 monitoring year averages

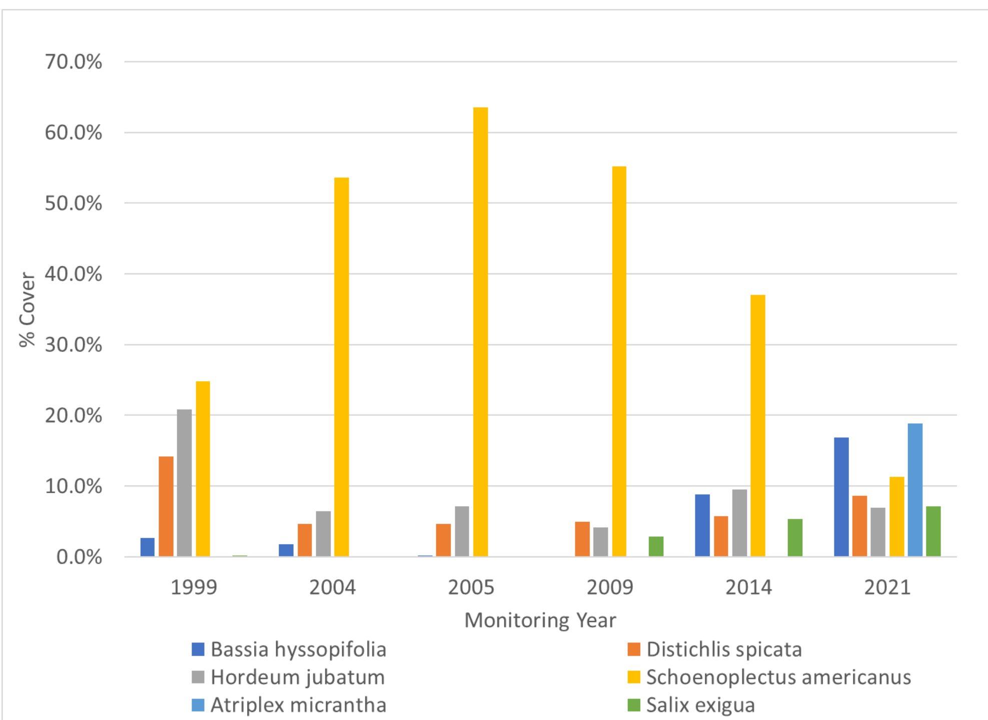


Figure 84. Cover of dominant species at DeChambeau Embayment by monitoring year

3.3.5 Lake-fringing Wetland Monitoring Discussion

The condition of the lake-fringing wetland monitoring sites varied by site in 2021, showing impacts from feral horse grazing at Warm Springs, and drought, and/or lowered a lake level at DeChambeau Embayment.

A significant decrease in live cover was observed at the Warm Springs site. The decrease in live cover at Warm Springs was accompanied by a five-fold increase in bare ground, and only slight increase in litter. Thus, live cover is not being replaced by litter, but by bare ground. To date, however, there has not been a change in plant species composition. The decrease in live cover at Warm Springs is believed to be largely attributable to grazing by feral horses as there has been no other notable disturbance at the site (e.g. fire, off-road activity). More often than not, large herds totaling several hundred animals have been present at Warm Springs during site visits summer through fall over the last several years. Outside of the transects, the feral horse herd has denuded the spring channels, compacted soils, and removed emergent vegetation around lake-fringing ponds at Warm Springs.

The Simons Springs site was stable, with little change in live cover or species composition. The transects at Simons Springs did not show any sign of impacts from horses, although horse impacts are evident elsewhere in the Simon Springs shoreline area. The specific area where the transects are located continues to support diverse and dense wetland vegetation.

At the DeChambeau Embayment site, changes in species composition were observed as nonnative and weedy annuals showed an increase in dominance. Stands of dry, decadent emergent vegetation (e.g *Schoenoplectus americanus* and *Typha latifolia*) suggests a drying of the area. Weedy nonnative species largely replaced the emergent vegetation as bare ground was absent. The cause of the drying is uncertain as lake level in July 2021 (6,380.4 feet) was higher than the previous sampling year (2014; 6,379.6 feet). This area of the shoreline is more influenced by changes in lake level due to the shallow sloping shoreline. The 2012-2016 drought plus lowered lake level may have contributed to the drying of the area and change in species composition observed in 2021.

3.4 Saltcedar Eradication

3.4.1 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 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.

3.4.2 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.

3.4.3 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 six sites were treated in 2021. Of these six sites, five were new and one site was retreatment of a plant previously treated (Joe Woods, pers. comm.).

Table 3.25. Total Tamarisk treatments sites by year and shoreline segment area

	Year						Total Treated per
Shoreline Area	2016	2017	2018	2019	2020*	2021	Shoreline Area 2016-2021
Bridgeport Creek	2		1	1			4
Lee Vining Creek	8	2	2	1			13
Mill Creek	62	7	8	6		2	85
Ranch Cove	30	9	6	5			50
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	23
Wilson Creek	10					1	11
Yearly Total Treated	151	35	35	36	1	6	264

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

3.4.4 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 five new plants were found in 2021, this number is small compared to previous years.

3.5 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 once supported a much larger waterfowl population. The SWRCB determined that diversion-induced impacts to waterfowl were more significant than for other waterbird species.

Waterfowl population monitoring in 2021 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.

3.5.1 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 85). Mono Lake is a highly productive, deep-water saline lake. Although the highly saline water, overall depth, and low diversity of food items limit habitat quality for waterfowl, waterfowl habitat is present in the form of freshwater streams, springs, and shoreline ponds. These resources, combined with high invertebrate production, provide opportunities for those waterfowl species able to exploit them.

<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 86). 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.

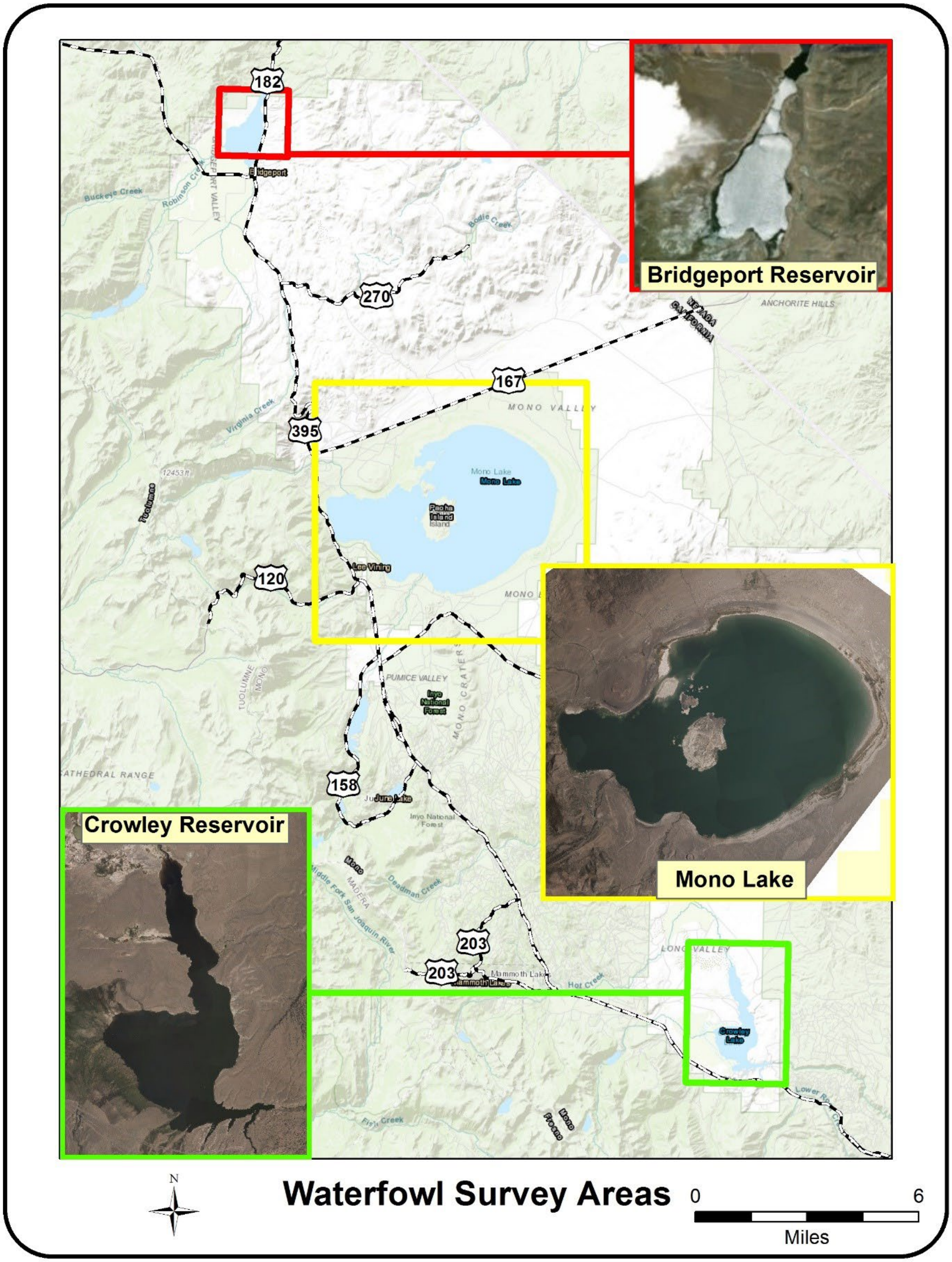


Figure 85. Waterfowl Survey Areas

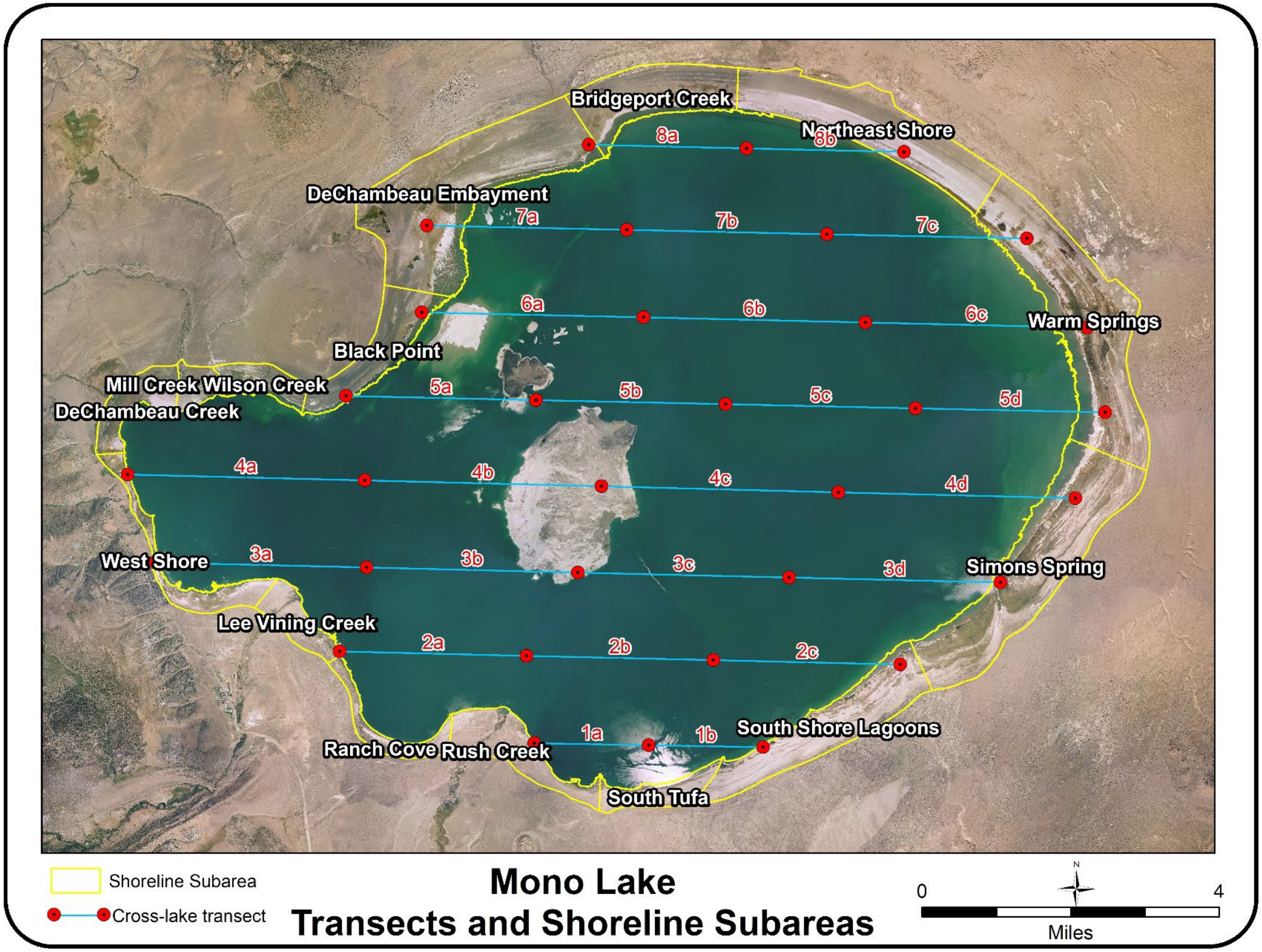


Figure 86. Mono Lake Shoreline Subareas and Cross-lake Transects

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 87). 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 1996) 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 1996, 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. 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 two years.

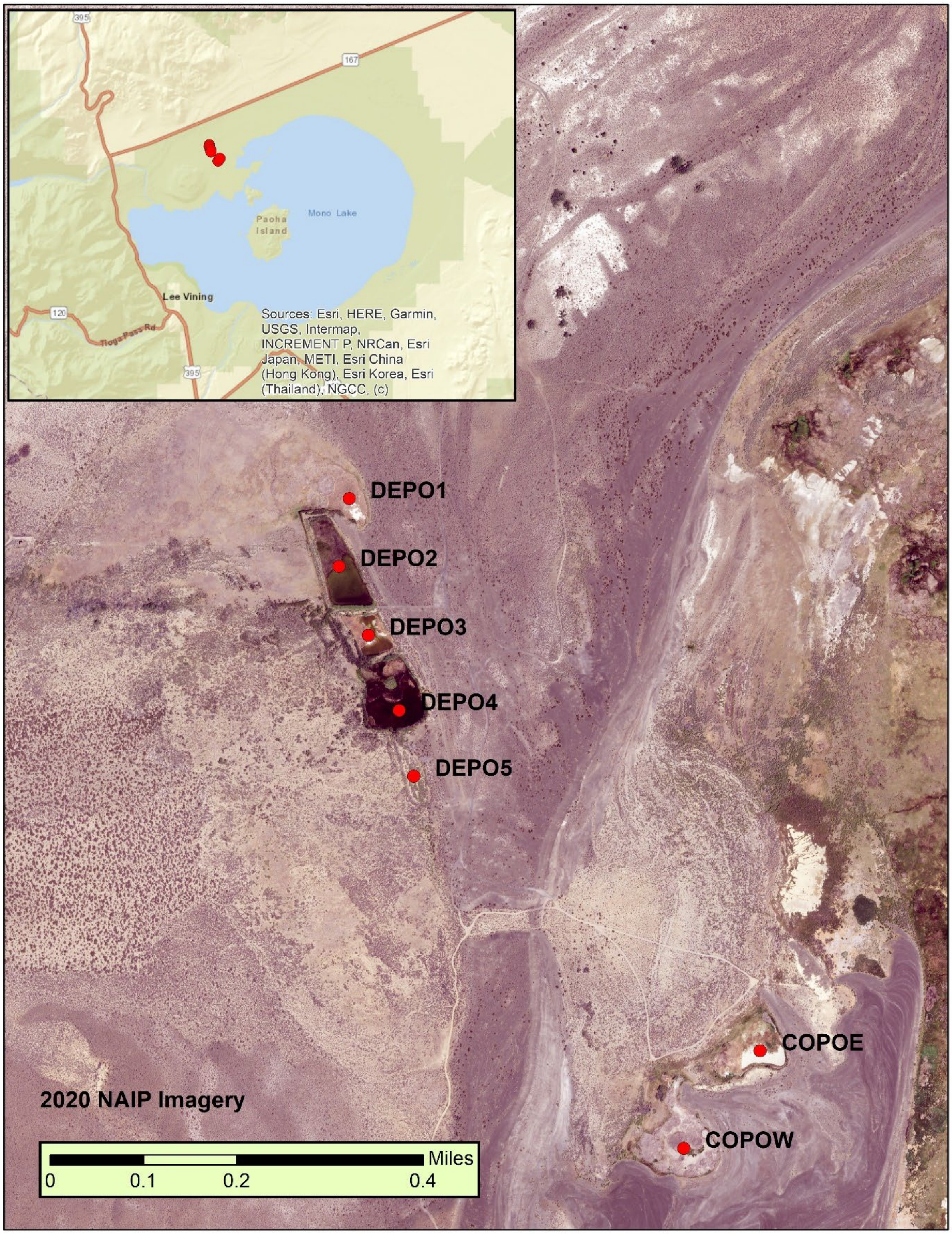


Figure 87. Mono Basin Restoration Ponds Locator Map

Bridgeport Reservoir

Bridgeport Reservoir is approximately 22 miles northwest of Mono Lake near the town of Bridgeport (Figure 85). 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 90's. 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, most 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 88). 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 2021, Bridgeport Reservoir held 4,129 acre-feet (https://cdec.water.ca.gov/dynamicapp/QueryMonthly?s=BDP). The September 2021 storage level was approximately 50% lower than September of 2020.
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).

The shoreline of Bridgeport Reservoir was subdivided three shoreline survey areas (Figure 88).

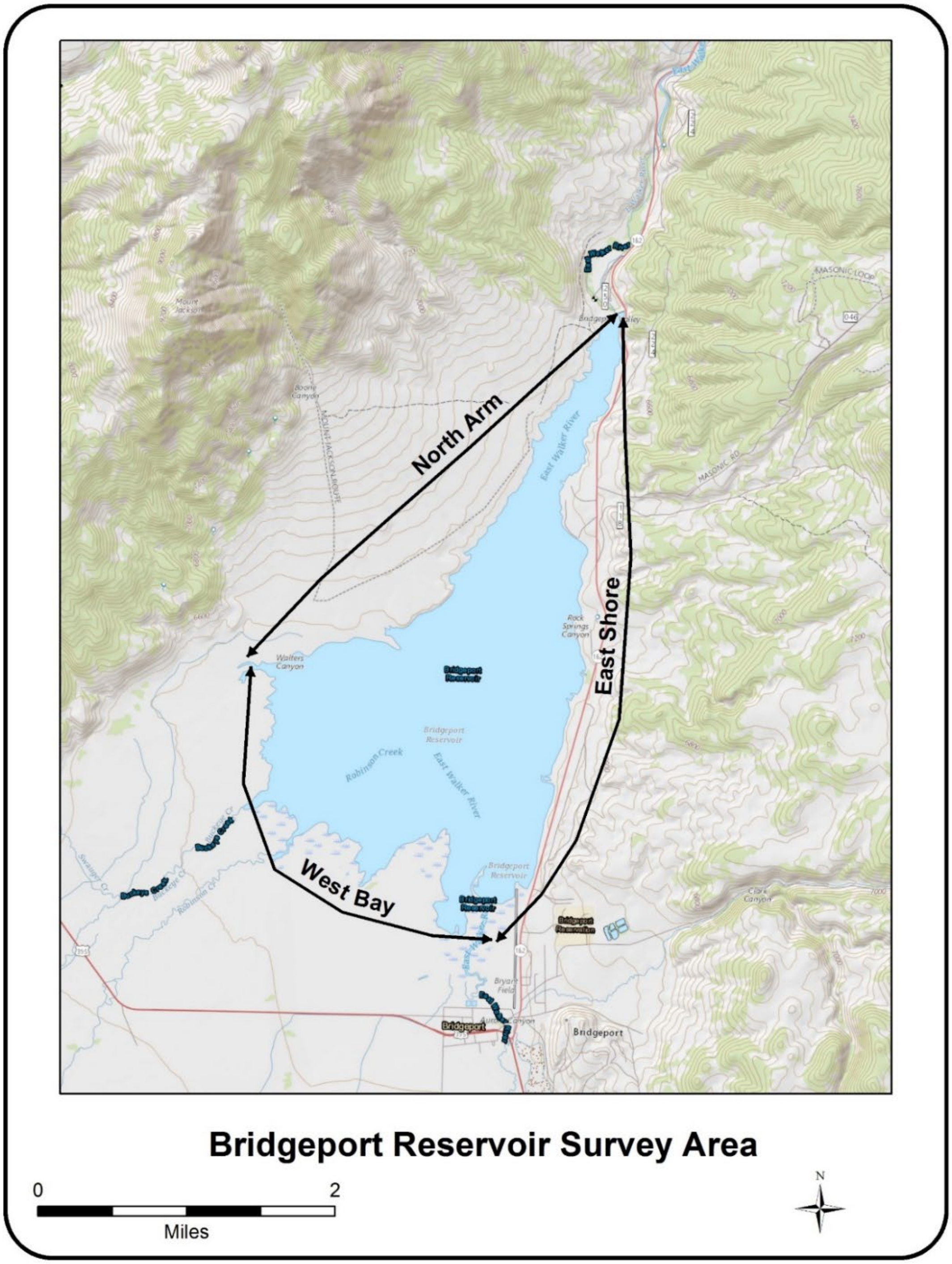


Figure 88. Bridgeport Reservoir Shoreline Subareas

Crowley Reservoir

Crowley Reservoir is approximately 31 miles southeast of Mono Lake, and 12 miles southeast of the town of Mammoth Lakes (Figure 85). 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 2021, Crowley Reservoir held 86,550 acre-feet (https://cdec.water.ca.gov/dynamicapp/QueryMonthly?s=CRW. The September 2021 storage level was within 5% of the September of 2020 level of 91,182.

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). 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 89).

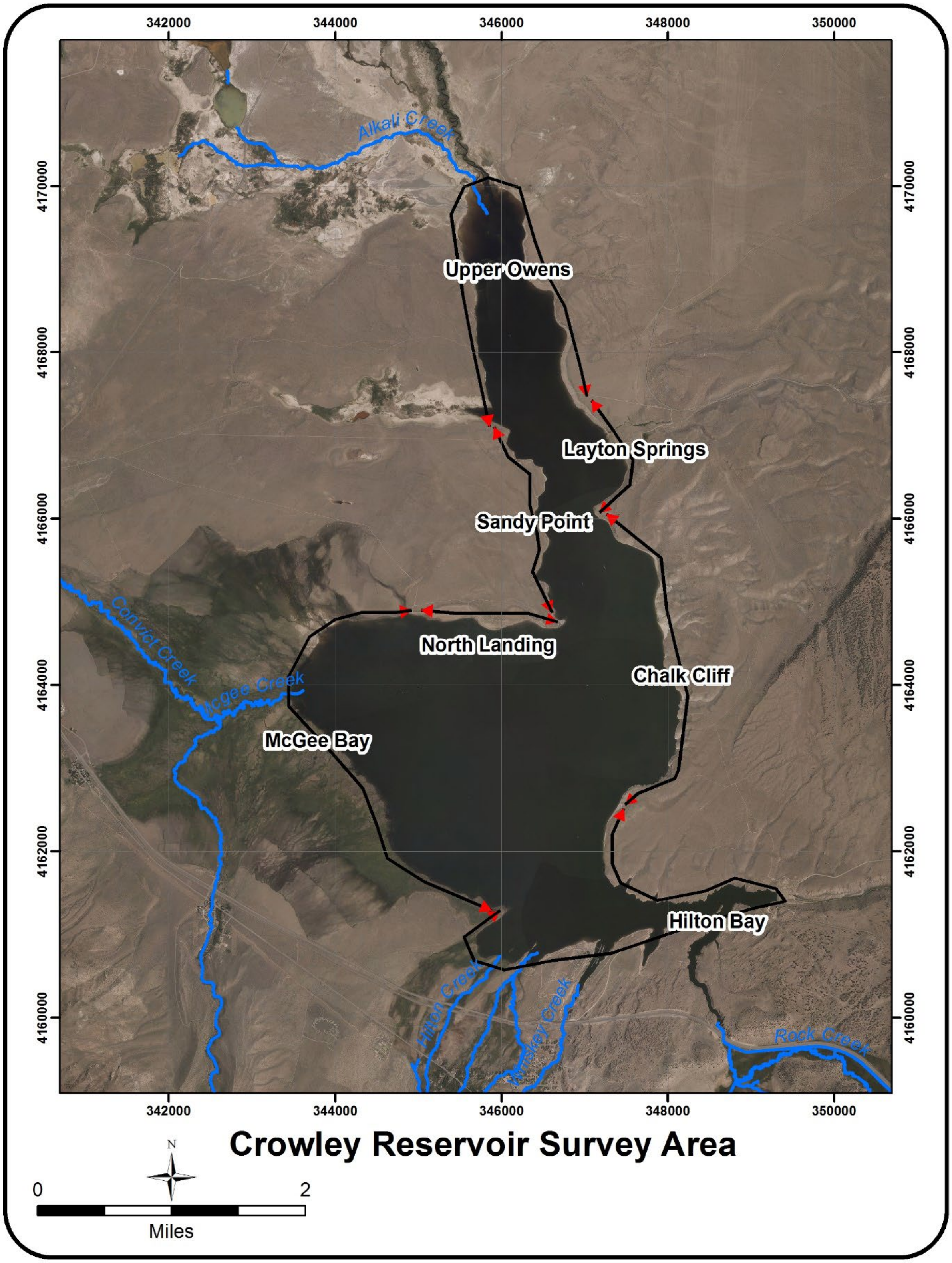


Figure 89. Crowley Reservoir Shoreline Subareas

3.5.2 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 2021. Surveys were conducted at three-week intervals beginning in early June (Table 3-27). 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 3.26. Summer ground count survey dates, 2021

Subarea	2021 Survey Number and Date		
	Survey 1	**Survey 2**	**Survey 3**
DECR	8-Jun	30-Jun	20-Jul
LVCR	8-Jun	30-Jun	19-Jul
MICR	8-Jun	30-Jun	20-Jul
RUCR	9-Jun	2-Jul	19-Jul
SASP	9-Jun	29-Jun	22-Jul
SOTU	7-Jun	2-Jul	20-Jul
SSLA	7-Jun	28-Jun	23-Jul
WASP	10-Jun	1-Jul	21-Jul
WICR	8-Jun	30-Jun	20-Jul
COPO	8-Jun	1-Jul	21-Jul
DEPO	8-Jun	1-Jul	21-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.

<u>Fall Surveys</u>

The fall 2021 surveys included the entire shoreline of Mono Lake, a subset of the cross-lake transects, and all seven restoration ponds. Six fall surveys were completed at two-week intervals between August 31 and November 10 (Table 3-28).

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 2021 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 2021 fall waterfowl surveys were conducted by the Mono Basin Waterfowl Program Director Deborah House and LADWP Watershed Resources Specialists Bill Deane.

Table 3.27. Fall 2021 Mono Lake Survey Dates

Survey Period	Shoreline	Cross-lake	Restoration Ponds
Survey 1	31 Aug-1 Sept	1-Sep	31-Aug
Survey 2	14-Sep	13-Sep	13-Sep
Survey 3	28-Sep	29-Sep	27-Sep
Survey 4	13-Oct	14-Oct	13-Oct
Survey 5	27-Oct	28-Oct	29-Oct
Survey 6	8-Nov	10-Nov	8-Nov

Bridgeport Reservoir Fall Waterfowl Surveys

The fall 2021 surveys included the entire shoreline and open water areas of Bridgeport Reservoir. Six fall surveys were completed at two-week intervals between August 31 and November 8 (Table 3-29). With the exception of Survey 5, which was a helicopter-based survey, all surveys were ground-based. The West Bay shoreline area was surveyed by walking out on the exposed reservoir bottom from Highway 182 and surveying the southern portion of the reservoir with a spotting scope. The remainder of the reservoir was surveyed at stationary viewing locations accessed from Highway 182.

Table 3.28. Fall 2021 Bridgeport Reservoir Survey Dates

Survey Period	Shoreline
Survey 1	31-Aug
Survey 2	14-Sep
Survey 3	27-Sep
Survey 4	13-Oct
Survey 5	27-Oct
Survey 6	8-Nov

Crowley Reservoir Fall Waterfowl Surveys

The fall 2021 surveys included the entire shoreline and open water areas of Crowley Reservoir. Six fall surveys were completed at two-week intervals between August 30 and November 9 (Table 3-30). All seven shoreline areas were surveyed during ground surveys. Ground access is good at most locations of Crowley, but limited in the area of highest waterfowl use in the McGee Bay area. Boat surveys work well at Crowley Lake as water depth is not an issue. This method is preferred because it is a time efficient way to completely survey the lake, however the boat was not always available. Because of good visibility and access from the shoreline, boat and ground survey are comparable in terms of providing good coverage of the lake.

Ground surveys were completed using spotting scopes and binoculars along shoreline transects or at stationary viewing locations along the shoreline. The McGee Bay shoreline area was thus surveyed by walking the shoreline. Surveys 1 and 5 were conducted from a boat by paralleling the shoreline at low speeds, stopping to survey shoreline areas when lighting and viewing conditions were most favorable.

Table 3.29. Fall 2021 Crowley Reservoir Survey Dates

Survey Period	Shoreline
Survey 1	30-Aug
Survey 2	15-Sep
Survey 3	28-Sep
Survey 4	16-Oct
Survey 5	29-Oct
Survey 6	8-9 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 2021, digital photographs were taken from a helicopter to document shoreline conditions. Photos of all three waterfowl survey areas were taken 20 October 2021. 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 2021 was 6,379.4 feet. This work was conducted by Deborah House, Mono Basin Waterfowl Program Director.

3.5.3 Waterfowl Data Summary and Analysis

Mono Lake

<u>Summer Waterfowl Community</u>

The summer waterfowl community data summary includes all breeding, migrant, and non-breeding/oversummering species observed in 2021. 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 2021 summer waterfowl survey data were summarized by survey number.

<u>Breeding Population Size and Composition</u>

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 2021 breeding waterfowl population total was compared to the long-term 2002-2019 mean. The breeding waterfowl community composition was evaluated by comparing 2021 values to the 2002-2019 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 2002-2019 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 2021. The total broods observed per shoreline subarea was compared with the long-term averages by shoreline subarea.

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

<u>Factors Influencing Waterfowl Breeding Populations</u>
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.

<u>Fall Surveys</u>

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 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 assumptions of simple linear regression.

Comparison with Reference Data

Waterfowl use of Mono Lake was compared to the reference sites by first calculating annual means +/- SE. The 2021 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 2021.

3.5.4 Waterfowl Population Survey Results

Mono Lake Summer Surveys

Summer waterfowl community

In 2021, 1,108 waterfowl and 10 waterfowl species were observed over the three summer shoreline surveys (Table 3-31) including eight breeding and two non-breeding species. Breeding waterfowl comprised the overwhelming majority of waterfowl present in summer (1,093 of 1,108). Waterfowl numbers highest on Survey 2 and lowest on Survey 3. Of the breeding species, Gadwall was most abundant, comprising 62% of breeding waterfowl at Mono Lake in 2021.

Table 3.30. Summer Ground Count Waterfowl Detections in 2021.

Mono Lake breeding waterfowl species are in bold type.

Species	Survey 1 June 7-10	Survey 2 June 30-July 2	Survey 3 July 19-23	Total Detections
Canada Goose	64	95	72	231
Cinnamon Teal	5	5	15	25
Gadwall	266	313	95	674
Mallard	45	66	17	128
Northern Pintail	1	6	1	8
Green-winged Teal	10	8	6	24
Redhead		12		12
Bufflehead	1	1	1	3
Common Merganser			1	1
Ruddy Duck			2	2
Total waterfowl by survey	392	506	210	1108

Breeding population size and composition

The breeding waterfowl population at Mono Lake in 2021 is estimated to be 364, or approximately 182 pairs. The 2021 breeding population was significantly higher as compared to the long-term mean of 304.8 +/- 20.7 SE or 152 pairs. Breeding was confirmed for Canada Goose, Cinnamon Teal, Common Merganser, Gadwall, Green-winged Teal and Mallard. Canada Goose and Gadwall numbers were well above average in 2021 (Figure 90). With the exception of Mallard, numbers of all other breeding species were slightly below their respective long-term means.

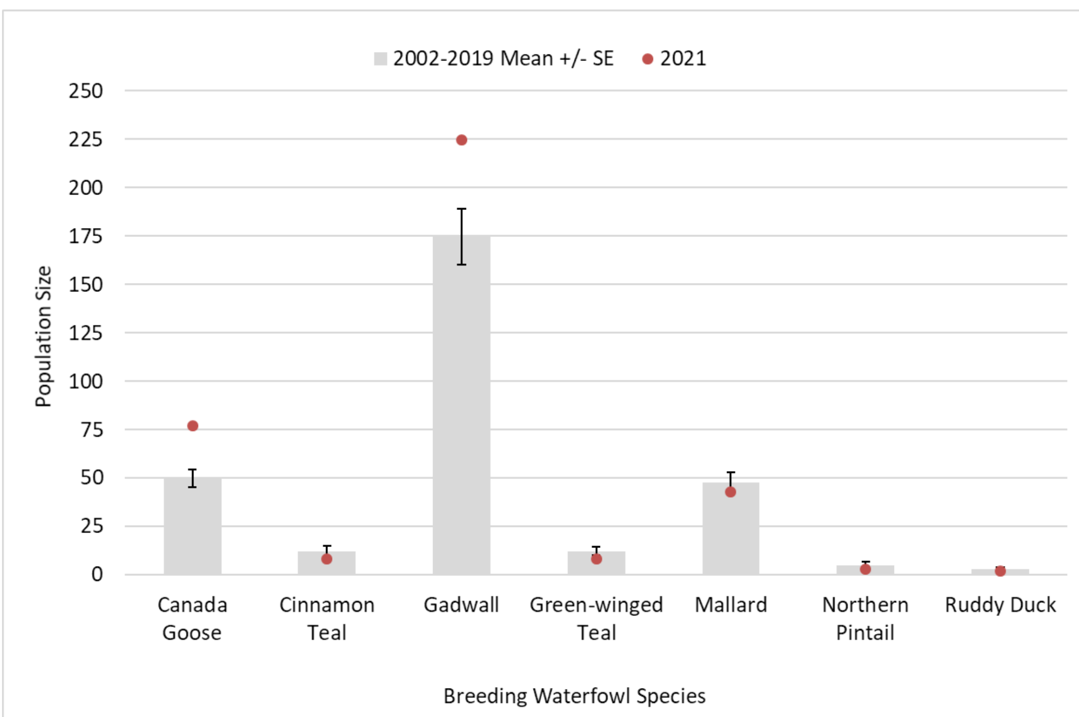


Figure 90. 2021 Breeding Waterfowl Population vs. Long-term Mean

A total of 83 waterfowl broods were found on the three surveys conducted in 2021, including nine Canada Goose and 73 dabbling duck, and one diving duck brood. Breeding was confirmed for six species, with brood numbers highest for Gadwall (Table 3-32). In 2021, broods were found at all shoreline areas, except South Tufa. The majority of broods (46; 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. This was the first year a Common Merganser brood was seen within the survey area.

The total number of waterfowl broods found (exclusive of Canada Goose) in 2021 (74) was significantly higher than long-term average of 46.9 +/- 3.9 of all three surveys combined. While conducting Survey 3, three additional hens were acting broody, but no brood was seen, and no other pairs or females without broods were observed.

Table 3.31. Waterfowl Broods by Shoreline Area, 2021

Breeding Waterfowl Species	DECR	LVCR	MICR	RUCR	SASP	SOTU	SSLA	WASP	WICR	Total 2021 Broods
Canada Goose	4		2		3					**9**
Cinnamon Teal	1		1							**2**
Common Merganser				1						**1**
Gadwall	8	11	11	6	8		5	1	13	**63**
Green-winged Teal	1		1							**2**
Mallard	2		1				2		1	**6**
Total broods per shoreline area	**16**	**11**	**16**	**7**	**11**	**0**	**7**	**1**	**14**	**83**

Habitat Use

Most dabbling duck activity was concentrated in and around nearshore water features including freshwater ponds, brackish ponds, and ria (Table 3-33). Secondarily, mudflats were used frequently by Mallard. The habitat use patterns of Canada Goose differed from the dabbling duck species in their greater use of meadow/marsh landtypes and barren shoreline areas of Mono Lake. Waterfowl with broods were seen most frequently in ria and freshwater ponds. Dabbling ducks fed most often in ria. Canada Goose foraged most frequently in ria, on mudflats, and on barren areas. Although classified as barren due to the very low cover, the areas where Canada Goose forage often support fresh young growth of wetland plants.

Table 3.32. Proportional Habitat use by Breeding Waterfowl Species, 2021

Landtypes		Breeding Waterfowl Species				
Modeled	Mapped	Canada Goose	Cinnamon Teal	Gadwall	Green-winged Teal	Mallard
Meadow Marsh		12%	0%	0%	0%	2%
	Marsh	*0%*	*0%*	*0%*	*0%*	*2%*
	Wet Meadow	*2%*	*0%*	*0%*	*0%*	*0%*
	Alkaline Wet Meadow	*10%*	*0%*	*0%*	*0%*	*0%*
	Dry Meadow/Forb	*0%*	*0%*	*0%*	*0%*	*0%*
Water		44%	56%	25%	33%	60%
	Freshwater Stream	*0%*	*0%*	*1%*	*0%*	*2%*
	Streambar	*0%*	*0%*	*8%*	*0%*	*1%*
	Freshwater Pond	*1%*	*56%*	*10%*	*33%*	*16%*
	Brackish Pond	*0%*	*0%*	*4%*	*0%*	*22%*
	Hypersaline Pond	*0%*	*0%*	*0%*	*0%*	*0%*
	Mudflat	*43%*	*0%*	*1%*	*0%*	*20%*
Upland		0%	0%	0%	4%	0%
Ria		24%	36%	68%	54%	38%
Riparian		0%	0%	0%	0%	0%
Barren Lake Bed		16%	8%	0%	4%	0%
Open Water		5%	0%	6%	4%	0%

<u>Restoration Ponds</u>

Summer Surveys

In 2021, a total of 35 waterfowl and five species were seen at the Restoration Ponds (Table 3-34). DEPO4 and DEPO3 were the most heavily used ponds. DEPO1 had water in it, but no waterfowl were observed using it. The water in DEPO1 was warm and had a heavy growth of algae all summer. Eleven waterfowl broods were observed at the Restoration Ponds including five at DEPO3 and six at DEPO4 (Table 3-35). As was the case for the shoreline areas, Gadwall broods were most abundant.

The number of waterfowl averaged over the three surveys in 2021 was below the long-term 2002-2019 mean (Figure 91). Waterfowl broods at the ponds were slightly above the long-term mean.

Table 3.33. Total Summer Waterfowl by Pond and Species, 2021

Species	DEPO1	DEPO2	DEPO3	DEPO4	DEPO5	COPOW	COPOE	Species Total
Cinnamon Teal			1	3				4
Gadwall		2	6	12	D	D	D	20
Northern Pintail		1			r	r	r	1
Redhead			2		y	y	y	2
Ruddy Duck			1	7				8
Pond Totals	0	3	10	22	0	0	0	35

Table 3.34. Waterfowl Broods at the Restoration Ponds, 2021

Species	DEPO3	DEPO4	Species Total
Cinnamon Teal	1		1
Gadwall	3	5	8
Ruddy Duck	1	1	2
Total Broods	5	6	11

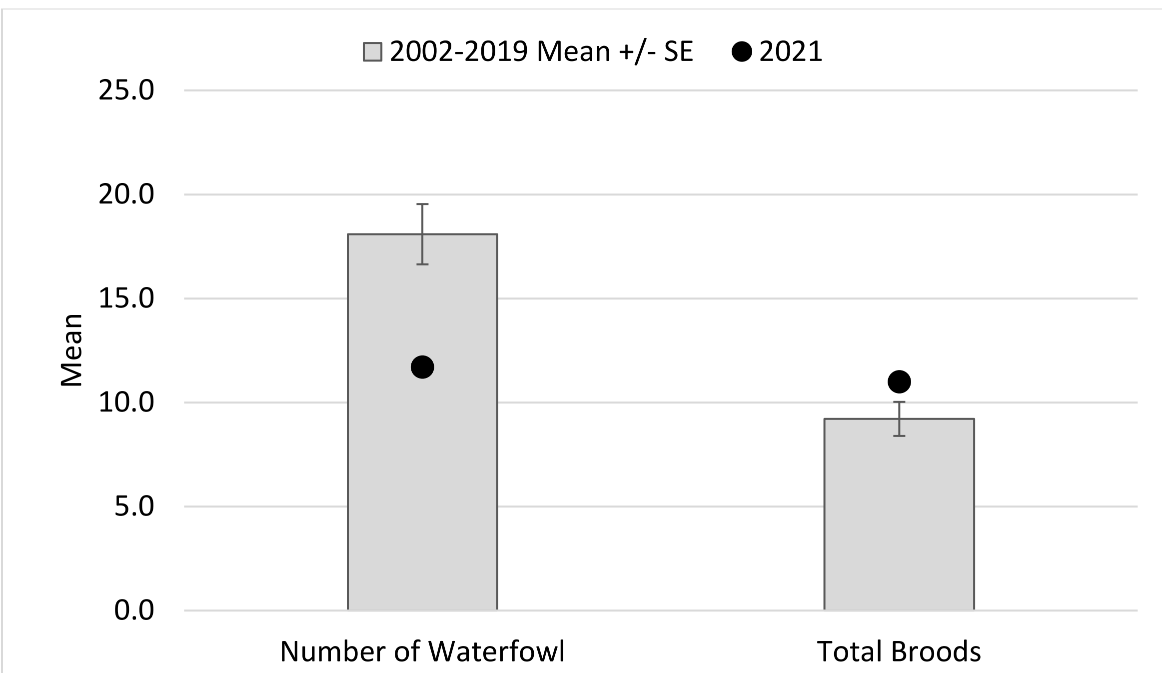


Figure 91. Mean Number of Waterfowl and Total Broods-Restoration Ponds, 2021

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 92) (r^2_{adj} = 0.513, p=0.0002). 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 93) (Student's t-statistic = -5.61, df=9, p<0.001).

No correlations were found between monthly values of adult *Artemia* or monthly *Artemia* biomass.

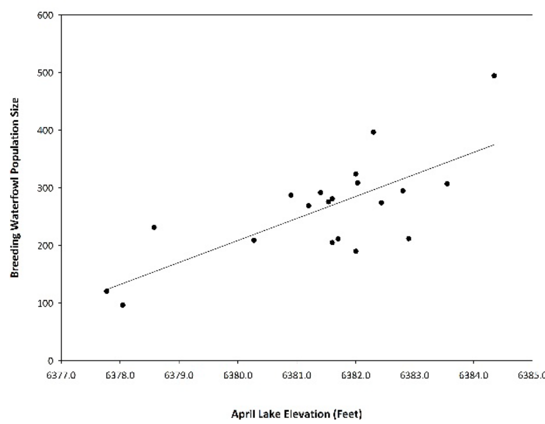


Figure 92. Annual breeding population size vs. April lake elevation

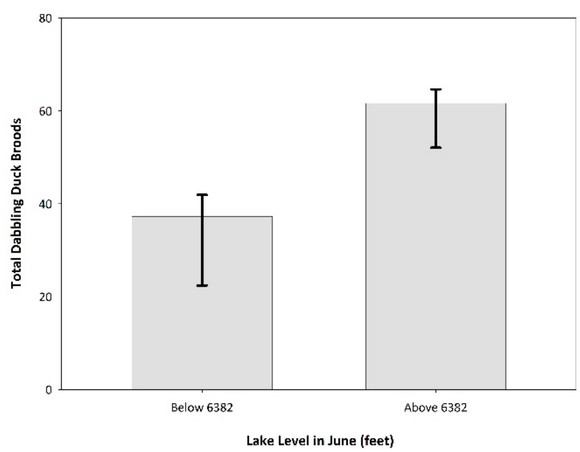


Figure 93. Total broods vs. lake level

Mono Lake Fall Surveys

Fall Waterfowl Population Size and Species Composition

A total of 12 waterfowl species and 39,336 individuals were detected during the six 2021 Mono Lake fall surveys (Table 3-36). Northern Shoveler and Ruddy Duck were the most abundant species, and combined, comprised 96% 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. In 2021, numbers were highest in early September, however a significant second pulse of birds arrived at Mono Lake in mid-October, such that total numbers were comparable to that seen in Early September. Ruddy Duck numbers typically show a seasonal peak the end of September through the end of October, and in 2021, 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.

Table 3.35. Species Totals, 2021 Mono Lake Fall Waterfowl Surveys

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Greater White-fronted Goose				17			17
Canada Goose	6				20		26
Cinnamon Teal	6	27					33
Northern Shoveler	7511	4097	4184	7037	2031	1354	26214
Gadwall	57	8	2		32		99
American Wigeon				30		16	46
Mallard	23	36	52	157	36	104	408
Northern Pintail		26	20			30	76
Green-winged Teal	38	66	80	400	88	225	897
Unidentified Teal			42	60			102
Canvasback						10	10
Ring-necked Duck					13		13
Ruddy Duck	34	462	5850	3093	1251	705	11395
Totals	7675	4722	10230	10794	3471	2444	39336

The 2021 fall waterfowl counts showed above average numbers during most of the fall period. 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 94). This early season peak has been largely due to the abundance of Northern Shovelers, an early season migrant in the region. In 2021, waterfowl totals remained high through mid-October due to a second large pulse of Northern Shoveler arriving at Mono Lake (see Table 3-36). After mid-October of 2021, waterfowl numbers at Mono Lake declined substantially, driven largely by departure of shovelers from the Mono Basin, and a significant reduction in the number of Ruddy Duck.

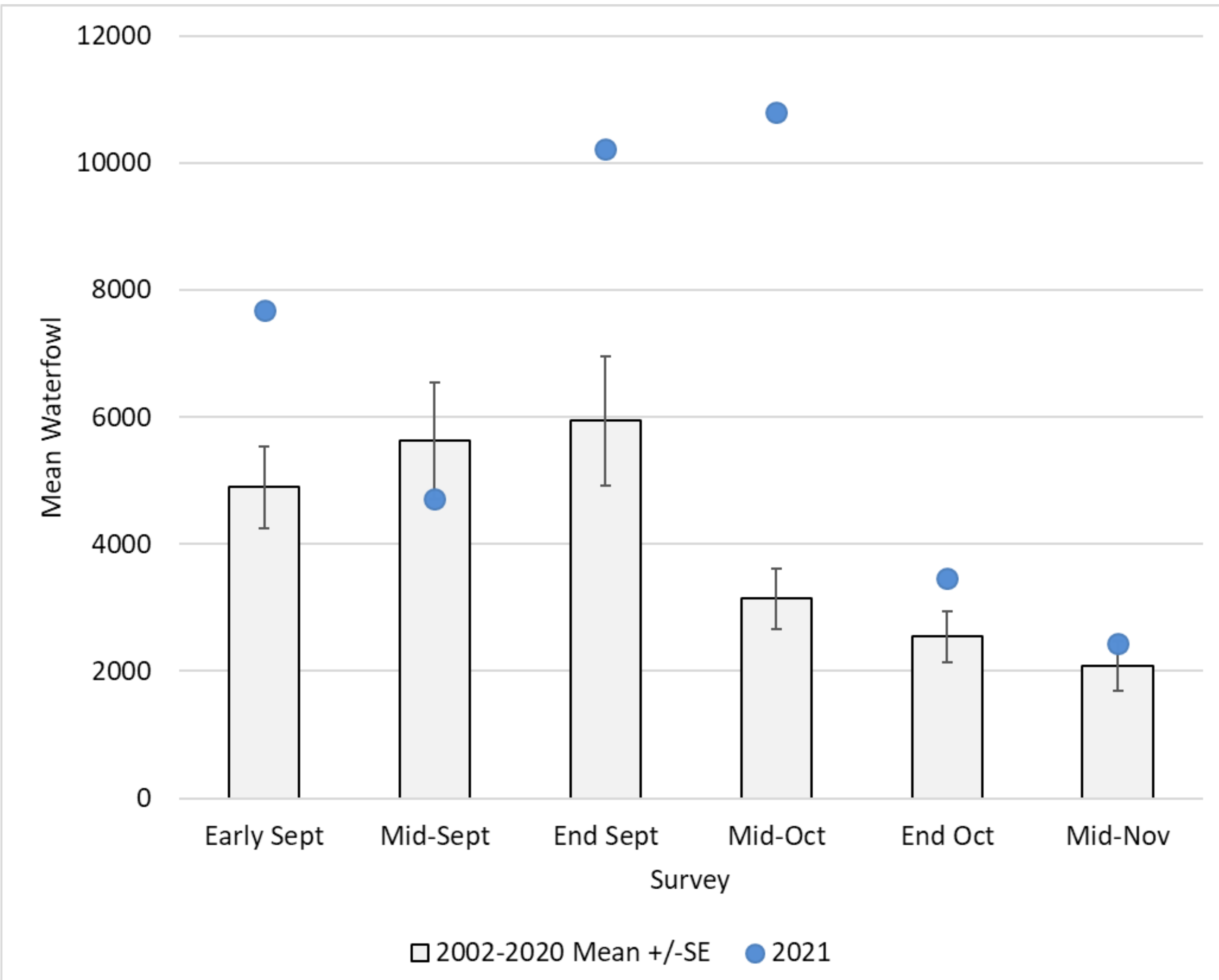


Figure 94. 2021 Mono Fall Waterfowl Survey Totals and 2002-2020 Means

Spatial Distribution

At Mono Lake, 75% of all fall waterfowl were detected during shoreline surveys, and 25% on cross-lake transects. During shoreline surveys, the majority of waterfowl were seen in the DeChambeau Embayment and Wilson Creek area (Figure 95). Wilson Creek is typically the main staging area in fall for waterfowl, and in particular, Northern Shoveler, and often the only location where large numbers are seen (1,000's). In 2021, large numbers did use Wilson Creek (over 11,000 seen over the six surveys), however there was also very high use of the DeChambeau Embayment area. Use of most other areas was below the long-term average in terms of overall proportion of observations.

Offshore use was almost entirely by Ruddy Ducks. Most Ruddy Ducks were encountered along the cross-lake transects closest to shore, between Black Point and Northeast Shore (Table 3-37).

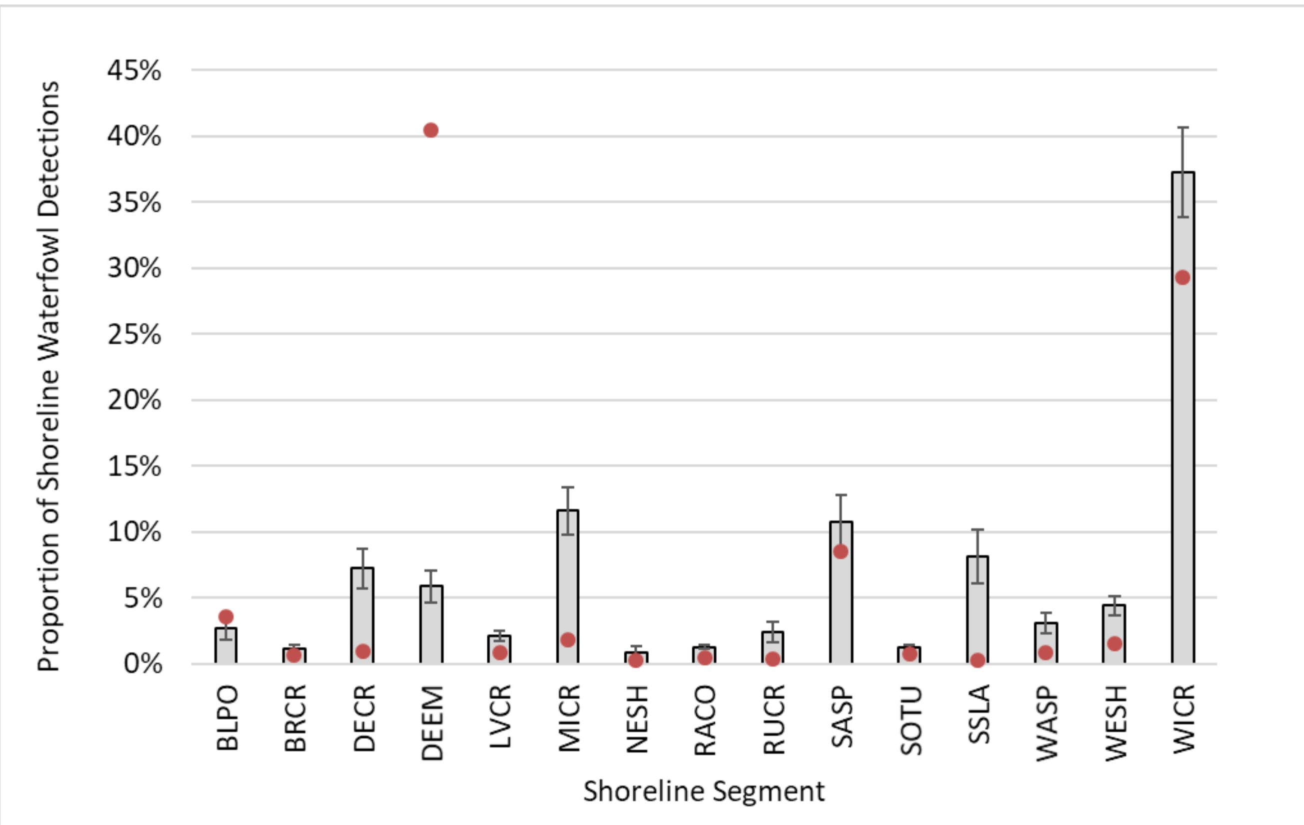


Figure 95. Shoreline Spatial Distribution of Waterfowl at Mono Lake, Fall 2021

Table 3.36. Distribution of Ruddy Ducks on Cross-lake Transects

Cross-lake subsection	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Total
4B			1				**1**
5A			25	200	38	62	**325**
5B				31	12	13	**56**
6A				55	40	14	**109**
7A		20	347	84	21	54	**526**
7B			8	1	6		**15**
7C		8	14	30	176	28	**256**
8A		258	363	152	128	103	**1004**
8B		136	519	345	137	10	**1147**
Total Offshore	**0**	**422**	**1277**	**898**	**558**	**284**	**3439**

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 105 waterfowl of six species were seen over the six fall surveys (Table 3-38). Unlike shoreline surveys, Gadwall were most abundant in fall at the Restoration Ponds. Similar to summer, DEPO4 attracted the most waterfowl. The 2021 total of 105 waterfowl over the six surveys was significantly below the 2002-2019 mean of 314.2 +/-130.4.

Table 3.37. Fall Waterfowl Totals by Pond, 2021

Species	DEPO1	DEPO2	DEPO3	DEPO4	DEPO5	COPOW	COPOE	Species Total
American Wigeon			4		Dry	Dry	Dry	4
Cinnamon Teal			1					1
Gadwall		8	8	42				58
Green-winged Teal		1		16				17
Mallard		11		1				12
Ruddy Duck		10	2	1				13
Pond Totals	0	30	15	60	0	0	0	105

Factors Influencing Mono Lake Fall Waterfowl Populations

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=0.625) (Figure 96).

The fall migratory population has been positively correlated with mean *Artemia* population size (r=0.463) (Figure 97). Variation in annual mean *Artemia* population size has explained just 21%

of the variation in total waterfowl number, however this relationship has been significant (p=0.40).

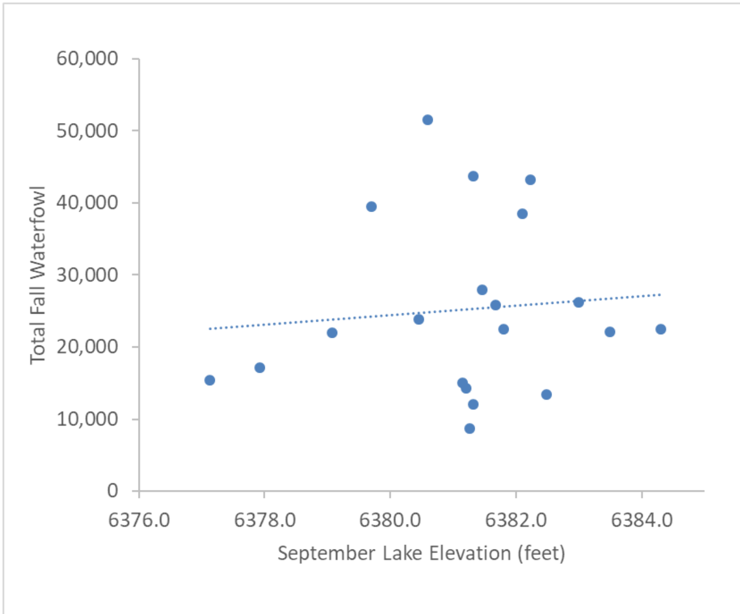


Figure 96. Total fall waterfowl population vs. September lake elevation

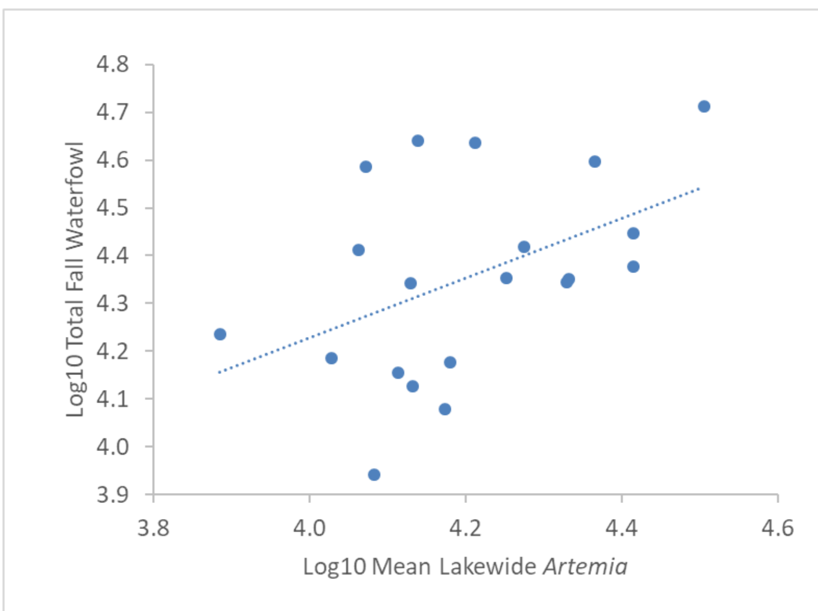


Figure 97. Total fall waterfowl vs. mean lakewide *Artemia*

Bridgeport Reservoir

<u>Fall Waterfowl Totals and Species Composition</u>

A total of 17 waterfowl species and 16,958 individuals were recorded at Bridgeport Reservoir over the six fall surveys in 2021 (Table 3-39). Geese and swans comprised approximately 21% of all waterfowl, and of this group, only Canada Goose was abundant and present on all surveys. Dabbling ducks totaled 72% of all waterfowl, and of the six dabbling duck species identified, Northern Shoveler and Green-winged Teal were most abundant. The most species-rich group was diving ducks, with eight species detected and divers as a whole comprised approximately 7% of all waterfowl.

Table 3.38. Species Totals, 2021 Bridgeport Reservoir Fall Waterfowl Surveys

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Snow Goose					1		1
Cackling Goose						39	39
Canada Goose	756	756	557	558	525	371	3,523
Northern Shoveler	1,900	2,049	323	4		20	4,296
Gadwall	662	349	108	20	160	50	1,349
American Wigeon		6	20		20	208	254
Mallard	100	247	2	16	486	252	1,103
Northern Pintail	12	21	2	4	60	10	109
Green-winged Teal	2,300	484	486	347	680	665	4,962
Unidentified Teal					200		200
Canvasback						4	4
Redhead	2		19	13			34
Ring-necked Duck	2		1	2			5
Bufflehead				8	30	21	59
Common Goldeneye						5	5
Hooded Merganser						1	1
Common Merganser	52	57	20	2		3	134
Ruddy Duck	12	70	127	184	250	237	880
Totals	5,798	4,039	1,665	1,158	2,412	1,886	16,958

Spatial distribution

Of the three subareas at Bridgeport Reservoir, waterfowl numbers were highest in the West Bay throughout the season (Table 3-40), although use of the East Shore (particularly the East Walker River bay) was similar in Early September. 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 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 3.39. Bridgeport Reservoir, Spatial Distribution by Survey, 2021

Survey	EASH	NOAR	WEBA
Early September	2,761	56	2,981
Mid-September	195	98	3,746
End of September	203	45	1,417
Mid-October	142	12	1,004
End of October	246		2,166
Mid-November	466	38	1,382
Total waterfowl by shoreline segment	**4,013**	**249**	**12,696**

Crowley Reservoir

Fall Waterfowl Totals and Species Composition

A total of 21 waterfowl species and 79,651 individuals were recorded at Crowley Reservoir over the six fall surveys in 2021 (Table 3-41). Geese and swans comprised only 0.8% of all waterfowl. Dabbling ducks totaled 70% of all waterfowl, and of the seven dabbling duck species identified, Northern Shoveler, Gadwall, Mallard, and Green-winged Teal were most abundant. Seven species of diving ducks were observed and divers as a whole comprised approximately 30% of all waterfowl. Ruddy Duck was overwhelmingly the most abundant of the divers.

Table 3.40. Species Totals, 2021 Crowley Reservoir Fall Waterfowl Survey

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Snow Goose				5			5
Greater White-fronted Goose			6	1			7
Cackling Goose				33		3	36
Canada Goose	151	51	76	35	160	77	550
Tundra Swan						41	41
Cinnamon Teal	134	22					156
Northern Shoveler	2,425	2,922	6,443	388	21	56	12,255
Gadwall	2,612	5,296	2,483	24	116	81	10,612
American Wigeon	2	100	190	44	192	62	590
Mallard	1,310	1,253	5,369	859	1,739	890	11,420
Northern Pintail	152	200	800	556	223	73	2,004
Green-winged Teal	550	1,753	3,308	1,778	3,087	1,299	11,775
Unidentified Teal	4,110	1,270	934	320			6,634
Canvasback	1	3	3	44	84	107	242
Redhead	12	14	187	31	2	9	255
Ring-necked Duck	4		34	139	60	181	418
Lesser Scaup				65	54	105	224
Surf Scoter					4		4
Bufflehead	1	6	8	21	268	266	570
Common Merganser			2		1	1	4
Red-breasted Merganser						1	1
Ruddy Duck	390	520	2,127	4,275	10,963	3,573	21,848
Totals	11,854	13,410	21,970	8,618	16,974	6,825	79,651

Spatial Distribution

During the 2021 surveys, the largest waterfowl concentrations at Crowley Reservoir were in McGee Bay and the delta of the Owens River (Table 3-42), with more than four times as many in McGee Bay than were observed in the Upper Owens. 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. Due to a low reservoir level in late fall at Crowley, the Upper Owens shoreline segment area appeared to be reduced in extent as compared to most years. 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. Waterfowl continued to show a pattern, however, of late-season use of the Chalk Cliffs area when increased numbers of dabbling ducks are then seen offshore or loafing along the narrow, dry beach. Yearly, increased use of Chalk Cliffs area has coincided with the opening of waterfowl hunting season, and waterfowl may be seeking refuge in this area of more difficult access. Hilton Bay has good waterfowl habitat with adjacent meadows, some fresh water inflow, and shallow waters, but the area is small in size, and supports fewer numbers of waterfowl than areas of comparable quality, likely because of the size difference. 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 3.41. Crowley Reservoir, Spatial Distribution by Survey, 2021

Survey	CHCL	HIBA	LASP	MCBA	NOLA	SAPO	UPOW
Early September	-	216	10	8,480	42	1	3,105
Mid-September	-	135	20	11,630		25	1,600
End of September	54	343	644	16,635	172	132	3,990
Mid-October	1	386	189	6,695	40	6	1,301
End of October	612	167	38	12,156	56	14	3,931
Mid-November	109	335	705	3,984	440	458	794
Total waterfowl by shoreline segment	776	1,582	1,606	59,580	750	636	14,721

Comparison to Reference Sites

Annual waterfowl totals from 2003-2020, have differed among sites (Figure 98). Although despite its much larger size, 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 2021, waterfowl use of Bridgeport Reservoir was well below the long-term mean. Totals at both Crowley Reservoir and Mono Lake were well above their long-term means.

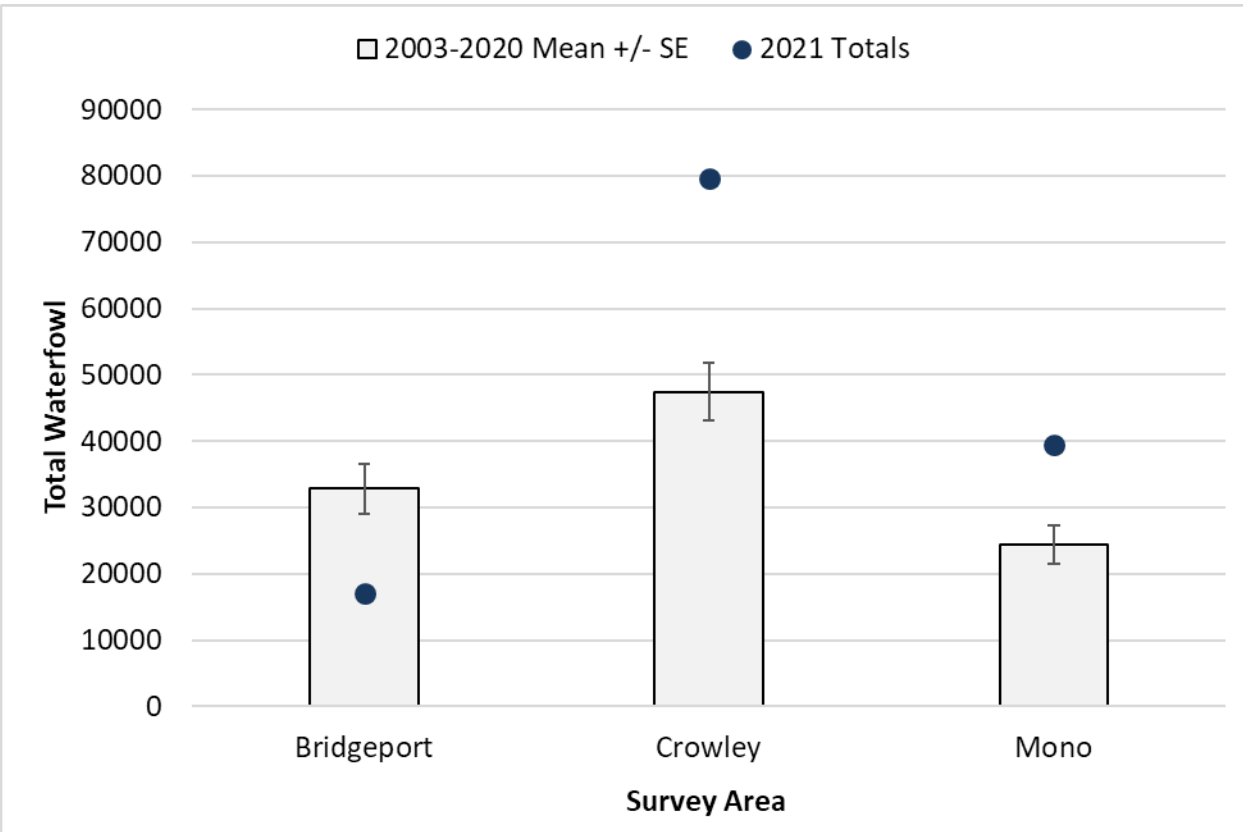


Figure 98. Comparison of Mean Fall Waterfowl at each of the Three Surveys Areas

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 Northern Shoveler and Ruddy Duck fall migratory populations. Mono Lake has on average attracted the largest proportion of the Mono County Northern Shoveler population, particularly in 2021 (Figure 99). 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 100).

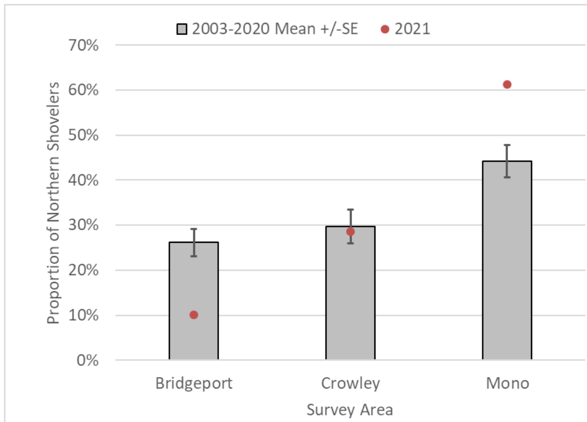


Figure 99. Proportional abundance of Northern Shovelers by survey area

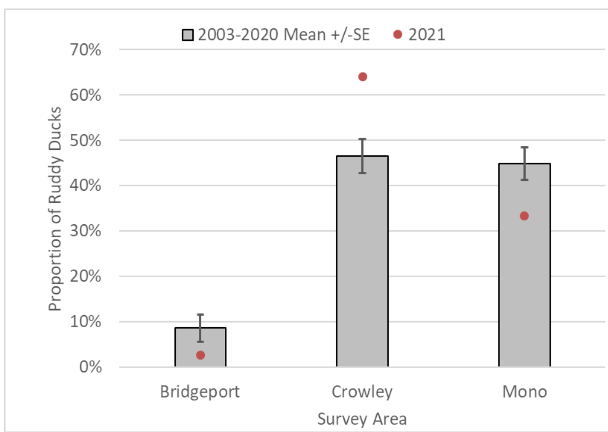


Figure 100. Proportional abundance of Ruddy Ducks by survey area

Aerial Photography of Waterfowl Habitats

<u>Mono Lake Shoreline Subareas</u>
<u>Black Point (BLPO)</u>

The Black Point (BLPO) shoreline area lies at the base of a volcanic hill on the northwest shore of Mono Lake (see Figure 85). 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 2021, the Black Point shoreline area was barren and dry (Figure 101, Figure 102), lacking any apparent brackish ponds. The decrease in lake level in 2021 resulted in fewer shoreline ponds as compared to 2020.

Figure 101. Black Point Shoreline Area, Western Half

Figure 102. Black Point Shoreline Area, Eastern Half

Bridgeport Creek (BRCR)

This shoreline area is at the terminus of the Bridgeport Creek (BRCR) 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 lake bed increases substantially without a subsequent expansion of vegetation, and brackish ponds disappear. In 2021, the eastern portion supported primarily meadow vegetation and extensive barren playa (Figure 103). 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 104), attracting shorebirds and small numbers of waterfowl.

Figure 103. Bridgeport Creek Shoreline Area, Eastern Portion

Figure 104. 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 (DECR) shoreline area is along the northwest shore of Mono Lake (see Figure 86). Flow in DeChambeau Creek is intermittent, and does not consistently reach the lakeshore. The DECR shoreline area has abundant freshwater resources, however, 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 2021, the exposed playa was mudflats, but by fall, the beach had dried (Figure 105, Figure 106). Bird activity was very high in this part of Mono Lake in the summer, but reduced in fall. A small beaver dam near shore was first noted in this area in 2014 was still active. Spring flow continued to reach the lake shore in numerous places (Figure 105, Figure 106).

Figure 105. The DeChambeau Creek Area, Looking Northeast

Figure 106. The DeChambeau Creek Area, Looking North

DeChambeau Embayment (DEEM)

The DeChambeau Embayment (DEEM) area lies just east of the DeChambeau Ranch, and the DeChambeau and County restoration ponds (see Figure 86). 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 - 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 an offshore island, as was last seen in 2015. 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 fall of 2021, waterfowl activity was well above the long-term mean, and centered around Tower Hot Spring (Figure 107). In the eastern extent, small, isolated brackish ponds were present, and areas of spring flow to the lake shore (Figure 108).

Figure 107. DeChambeau Embayment, Tower Hot Spring

Most of the waterfowl activity was around this off-shore spring.

Figure 108. DeChambeau Embayment, Eastern Extent

Spring flow to the lake and a complex shoreline existed in this part of DEEM in 2021.

Lee Vining Creek (LVCR)

Lee Vining Creek (LVCR), 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 2021, the decline in lake level as compared to 2020 resulted in an increase in exposed playa and drying of some delta soils. There was increased channelization at the mouth of the creek, creating a complex delta (Figure 109). In the northern portion of the delta, the fresh shoreline pond that dried last year due to channelization of flow and draining, remained dry. As has been the case for the last few years, waterfowl broods in the Lee Vining Creek area were generally seen near shore in a small bay just west of the delta outflow (Figure 110).

Figure 109. Lee Vining Creek Delta

Figure 110. Lee Vining Creek Delta, western portion.

Most waterfowl broods were observed in this small bay to the west of the Lee Vining Creek outflow

Mill Creek (MICR)

Mill Creek (MICR) 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 111). 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 2021, the beaver activity and amount of ponding appeared increased over 2020. Due to sediment buildup, the outflow channel to the lake had narrowed and more ponding was occurring near shore, downstream of the beaver dams (Figure 111).

Figure 111. Mill Creek Delta

Northeast Shore (NESH)

In the Northeast Shore (NESH) 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 112). 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 2021, the Northeast Shore area consisted primarily of dry playa, as is typical (Figure 113).

Figure 112. An Unnamed Spring Along Northeast Shore

In 2021, this small spring did not discharge directly to the lake.

Figure 113 Northeast Shore, Looking North

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

Ranch Cove (RACO)

The Ranch Cove (RACO) 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 2021 Ranch Cove showed a dry beach lacking onshore ponds or direct spring input (Figure 114).

Figure 114. 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 2020. There was an increase in exposed playa and the flow of Rush Creek was confined to the mainstem (Figure 115). Fresh water ponds in the delta were receiving less inflow from the mainstem, were drying and becoming algae-covered (Figure 116). Beaver activity in the form of cut willows was seen in the delta for the first time.

Figure 115. Rush Creek Delta

In 2021, flows were confined to the mainstem in the delta. Fresh ponds left of the mainstem in this picture were drying and becoming algae-covered.

Figure 116. Fresh ponds in delta were algae-covered in summer (photo taken July 19, 2021)

Simons Spring (SASP)

The Simons Spring subarea (SASP) includes the southeastern portion of the lakeshore (Figure 86). 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). Being subject to the action of longshore currents, shoreline features of Simons Spring are dynamic, particularly west of Simons Spring fault line. Due to the shoreline gradient, small changes in lake elevation result in large changes in the degree of shoreline flooding.

Open fresh water ponds are a prominent feature of the Simons Spring area, however their presence 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 have developed, supported by inflow from up gradient springs. Many of the freshwater springs in this area reach the lakeshore through breaks in littoral bars, creating extensive mudflats on exposed playa. Although there may be a physical connection between the mudflats and lake water, the very shallow ponds formed on shore are fresh due to the high spring flow, and are colonized within 1-2 years by wet meadow 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 likely occurs due to the multiple areas of spring flow that reach 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 fairly good at Simons Springs in 2021, 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 117). Although emergent vegetation continued to encroach, small open, fresh water ponds upgradient of a large berm persisted through the year. 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 Simon's Spring shoreline area, further enhancing the resources for waterbirds (Figure 118).

East of the Fault line, several open water fresh and brackish ponds persisted (Figure 119). The shoreline east of the Fault line was steeply eroded, lacking shallow areas and mudflats. The area east of the Fault line has seen more feral horse activity than to the west to date and springs and ponds have shown impacts including soil compaction and the removal of wetland vegetation through grazing. Feral horse herds were seen frequently summer through fall, primarily east of the Fault line.

Figure 117. Simon's Spring, West of the Fault line

Figure 118. Outflow of Goose Springs, at West End of Simon’s Spring Shoreline Area

Figure 119. Shoreline Ponds East of the Fault line at Simon’s Springs

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 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 scattered wetlands filled with groundwater, creating a series of scattered fresh water ponds in the South Shore Lagoons subarea.

Waterfowl habitat conditions in the South Shore Lagoons area were poor in 2021. Very few ponds were present and the shoreline was dry and steeply eroded along much of its length (Figure 120). The semi-permanent pond at the western extent of the subarea was almost dry in early June, and dry by fall (Figure 121). At Sand Flat Spring, there was very little open water habitat, and no direct connection between spring flow and the open water (Figure 122).

The Goose Springs area, which was excellent waterfowl habitat for many years, continued to degrade. The outflow of Goose Springs is now fairly channelized and exiting to the Simon's Spring area to the east (Figure 123). 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 120. Overview of the South Shore Lagoons Area

This area was dry and the shoreline eroded limiting shallow feeding areas.

Figure 121. South Shore Lagoons, West

The semi-permanent pond at the western extent of the subarea was dry by October when this photo was taken.

Figure 122. Sand Flat Spring

In 2021, there was no direct connection between spring flow from Sand Flat Spring and lake waters.

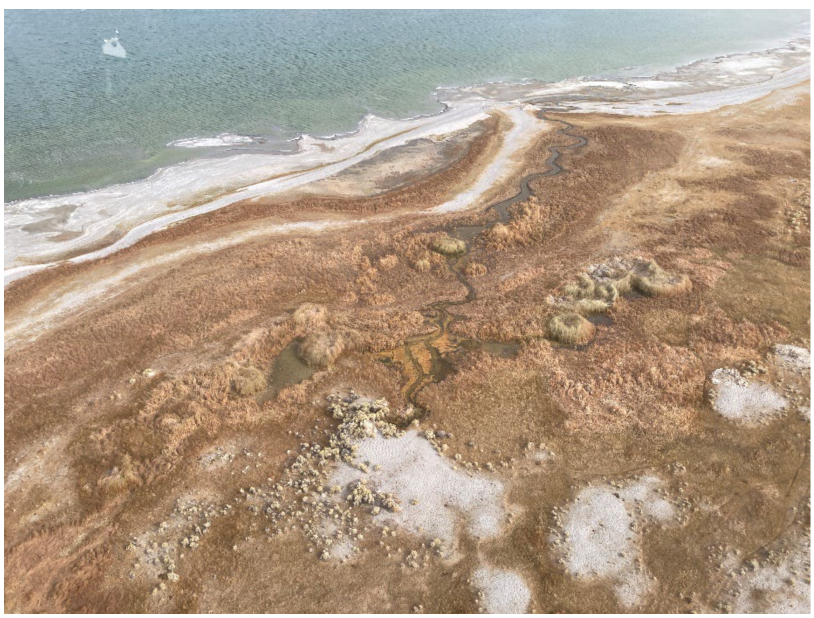


Figure 123. Goose Springs

The outflow of Goose Springs is channelized and fresh and brackish ponds in the area have degraded, reducing habitat quality for waterfowl.

South Tufa (SOTU)

The South Tufa area (SOTU) 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.

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 124). The eastern portion was dry and sandy in summer as well as fall (Figure 125).

Figure 124. South Tufa, near Navy Beach

Figure 125. 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 the only place in the Warm Springs subarea where waterfowl are consistently encountered.

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 this year, as all of emergent vegetation along the spring channels and around the ponds had been consumed, and the meadows were grazed down to almost zero stubble, and bare patches of soil were appearing.

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 (at least this was the case in 2021). In 2021, the area continued to be very wet, creating multiple shallow, open water ponds (Figure 126). Whether the 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 were 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 127).

Figure 126. Overview of Warm Springs

The Warm Springs area was very wet in 2021.

Figure 127. Warm Springs, North Pond, Looking East.

Feral horses had removed all of the emergent vegetation surrounding this pond in 2021.

West Shore (WESH)

The majority of the West Shore subarea (WESH) 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 128).

Figure 128. Overview of the West Shore, Looking North/Northwest

Wilson Creek (WICR)

Wilson Creek is along the northwest shore (see Figure 86) and 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. Due to the shelter, 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 2021, the Wilson Creek area continued to support a fresh water pond along the west side of the bay. Unlike 2020 however when multiple waterfowl broods were seen in the pond, there appeared to be little fresh inflow, the pond was stagnant, and little used by waterfowl in summer or fall (Figure 129). 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 130).

Figure 129. Wilson Creek Bay, as Viewed From the Southwest

The freshwater pond on the west side of the bay was receiving little inflow in 2021, and stagnant conditions resulted in little use in summer and fall.

Figure 130. Wilson Creek Bay, as Viewed From the East

Bridgeport Reservoir Shoreline

All three shoreline segments at Bridgeport Reservoir: North Arm, West Bay, and East Shore are shown in Figure 131. 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. In 2021, some reduction in the amount of flooding and the complexity of the shoreline as compared to fall of 2020, when the reservoir was higher.

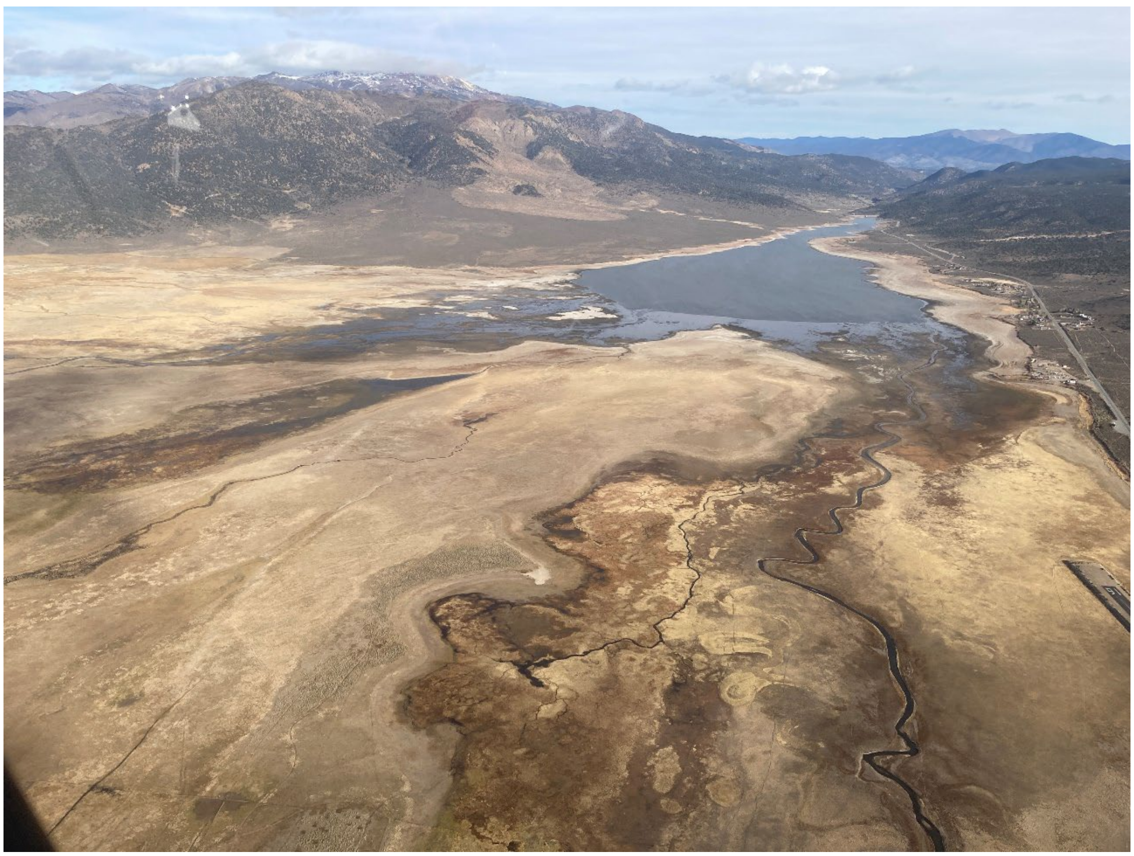


Figure 131. Bridgeport Reservoir, Looking Northwest

Crowley Reservoir Shoreline Subareas

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 132).

Figure 132. Chalk Cliffs

Hilton Bay (HIBA)

Hilton Bay includes Big Hilton Bay to the north and Little Hilton Bay to the south (Figure 133). 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 133. 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. Reservoir level was very low in late fall of 2021, exposing a large amount of barren reservoir bottom in the Layton Springs area (Figure 134)

Figure 134. Layton Springs

McGee Bay (MCBA)

The McGee Bay shoreline area supports mudflat areas immediately adjacent to wet meadow habitats. 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. In late fall of 2021, a low reservoir level resulted in a broad expanse of mudflats in much of the area (Figure 135) and exposed reservoir bottom in other parts with no direct inflow (Figure 136).

Figure 135. McGee Bay Shoreline South of McGee and Convict Creek Outflow

Figure 136. 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 (Figure 137). The low reservoir level in late fall 2021 resulted in the development of extensive mudflats in the North Landing area.

Figure 137. North Landing

Sandy Point (SAPO)

Most of the length of Sandy Point area is bordered by cliffs or upland vegetation. Small areas of meadow habitat occur in this area, and limited freshwater input occurs at Green Banks Bay. A low reservoir level in late fall of 2021 created a large sandy beach in this area (Figure 138).

Figure 138. 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. In 2021, this subarea included a large area of exposed reservoir bottom due to the low reservoir level (Figure 139).

Figure 139. Upper Owens Delta

3.5.5 Waterfowl Survey Discussion

3.5.5.1 Summer Ground Surveys – Mono Lake Shoreline

Breeding Population Size and Composition

Waterfowl breeding activity at Mono Lake in 2021 was high. The breeding waterfowl population at Mono Lake in 2021 was the third largest since 2002. Waterfowl breeding success was also good, as the number of broods observed at Mono Lake was the highest over the entire 2002-2020 study period. Breeding activity was high in 2020 as well, and this trend continued through 2021, despite a decline in lake level. Many breeding waterfowl species exhibit a homing tendency, returning to either their natal grounds to nest, or returning to where they nested successfully the prior year (Johnson and Grier 1988). In 2020, the lake was still above 6382 feet in spring and early summer, and breeding activity was high. Thus, the high breeding population and numbers of broods in 2021, despite a decline in lake elevation could be a carryover from last year. Despite the decline in lake level, breeding conditions remained good in some key areas. Another potential explanation to be explored is that Mono Lake attracted birds that were displaced from other areas due to drought and poor conditions elsewhere.

Spatial distribution

Breeding waterfowl are concentrated into highly localized areas around the shoreline of Mono Lake, where fresh water resources occur for young ducklings. In 2021, 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. Limnology data for June 2021, indicates that the western sector of Mono Lake supported higher numbers of shrimp than the east side. Throughout the month of June, waterbird activity was quite high along the northwest shore, especially in comparison to other areas along the lakeshore, suggesting this area had good biological productivity in early summer.

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 2021, 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, and this was one of the important breeding areas in 2021.

Waterfowl conditions at Goose Springs continue to degrade, and although for many years, the Goose Springs area supported the highest number of broods, little waterfowl activity, and few broods were observed in 2021.

The breeding waterfowl at Mono Lake is responsive to annual changes in habitat conditions as these annual changes in spatial distribution indicate.

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, Pietz et al. 2003, Kaminski and Prince 1981, Krapu et al. 1983). 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 2021, breeding dabbling duck activity was concentrated in and around freshwater sources including ponds, spring and creek outflow areas of ria, and brackish ponds. As is typically seen, Canada Goose used meadows areas more than dabbling ducks, however in 2021, Canada Goose were also seen frequently in freshwater outflow areas of ria. Freshwater ponds are an important component of the breeding waterfowl habitat at Mono Lake that was used by all dabbling ducks species, but not Canada Goose. Freshwater outflow areas of creeks and springs (=”ria”) were used primarily by Gadwall for feeding, suggesting use of *Artemia*. Brackish ponds at Mono Lake were used heavily by all waterfowl for feeding, including hens with broods. Although not studied, the use of brackish ponds by waterbirds at Mono Lake (D. House, pers. obs.) suggests they can be highly productive systems. The presence of brackish ponds, particularly when associated with or near freshwater ponds enhances habitat productivity and available feeding opportunities for breeding waterfowl at Mono Lake. The only species that regularly used meadow habitats was Canada Goose, which was often seen feeding with broods in alkaline wet meadow habitats near or on shore. Canada Goose is almost exclusively herbivorous feeding on 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.

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 the June lake level above a threshold of 6,382 feet. Below this lake level, there has been no significant effect of lake level. As lake level decreases, the number and size of ponds- particularly along the south shore from South Shoreline to Simons Springs- decrease. 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. Other factors such as access to food, which are influenced by lake level and bathymetry, could potentially influence waterfowl breeding.

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 2021. Waterfowl use of the restoration pond complex as a whole continued to be below the long-term average, since the County Ponds remain inactive, however the improved brood numbers are perhaps promising. Repair work to the infrastructure of the DeChambeau ponds continues, with a goal of further improvements in habitat conditions.

3.5.5.2 Fall Aerial Counts

Mono Lake - Population Size and Species Composition

Fall waterfowl numbers at Mono Lake in 2021 were the fourth highest of the entire 2002-2021 study period. A slight seasonal shift in use was observed in 2021. Past monitoring has shown that waterfowl totals at Mono Lake have been highest during the month of September, with significantly reduced numbers by mid-October through mid-November. In 2021, 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 possible that early season Northern Shoveler flocks at Mono Lake may be 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.

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. Although the Wilson Creek area makes up <2% of the entire shoreline area, it supported 29% of all dabbling ducks in 2021. 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. The DeChambeau Embayment area attracted more waterfowl in fall than any other shoreline area. The area around Tower Hot Spring is relatively shallow and would provide dabbling ducks good access to food resources present. Use of the South Shore Lagoons shoreline area continued to be low into fall due to small scale habitat changes noted earlier that affect the quality and quantity of fresh and brackish ponds onshore.

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 portion 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.

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. 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. 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. Waterfowl totals at Mono Lake are driven largely by Northern Shoveler and Ruddy Duck numbers and these species are found primarily in deltas or nearshore waters – thus are not tied to onshore ponds or the presence of suitable nesting sites or cover as are breeding birds. The abundance of *Artemia* however, one of the two primary food resources, has shown some influence on annual total fall waterfowl. 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.

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.

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.

3.5.6 Aerial Photography of Waterfowl Habitats

In fall of 2021 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 2020. Slight increases in the amount of exposed playa were evident in all shoreline areas. Grazing by feral horses was particularly heavy in the Warm Springs and Simons Spring areas.

Conditions were fairly similar to 2020, with some notable exceptions. Conditions in the Rush Creek delta deteriorated as flow into the fresh water ponds was reduced, and the ponds became algae-covered. Waterfowl habitat conditions continued to be fairly good at Simons Springs in 2021, at least in the area west of the fault line. Waterfowl habitat conditions in the South Shore Lagoons area were poor in 2021 as very few ponds were present and the shoreline was dry and steeply eroded along much of its length. At Mill Creek, beaver activity continues creating open water ponds used by waterfowl.

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.

A decrease in reservoir level at Bridgeport Reservoir resulted in a reduction in the aerial extent of flooding of feeding areas near the deltas of the East Walker River, Robinson and Buckeye Creeks. The low level of Crowley Reservoir, resulted in increased mudflats in some areas such as McGee Creek. The Upper Owens area was reduced in extent due to the low reservoir level. Heavy algal growth was not seen in early fall of 2021, as has occurred in previous years.

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) **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.

2) **Restoration Pond Monitoring** – Various partners including Inyo National Forest, Mono Lake Committee, other local groups and volunteers are implementing infrastructure repairs at the DeChambeau Ponds to restore the habitat for waterfowl and other birds. If these partners are conducting waterfowl monitoring, the monitoring being conducted under the Waterfowl Restoration Plan may be redundant. It is recommended the Waterfowl Director work with these groups to determine the need for continued monitoring under the Plan.
3) **Vegetation Status in Lake-fringing Wetlands** - The vegetation monitoring conducted at the lake-fringing wetland sites in 2021 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 transects in Rush Creek and Lee Vining Creek be completed in 2022, and thereafter resume vegetation transect monitoring every five years as required under the plan, or more frequently, if data needs demand.

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