Los Angeles Department of Water & Power

CUSTOMERS FIRST

May 10, 2019

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 (SWRCB) 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 2019", which contains the following four reports required by the Orders. The reports are as follows:

- Section 2: Mono Basin Operations: Runoff Year (RY) 2018-19 and Planned Operations for RY 2019-20. Please note that the planned operations through September 30, 2019, will follow the Temporary Urgency Change Petitions as approved by your agency on April 16, 2019.

- Section 3: Mono Basin Fisheries Monitoring Report: Rush, Lee Vining, and Walker Creeks 2018

- Section 4: RY 2018 Mono Basin Stream Monitoring Report

- Section 5: Mono Basin Waterfowl Habitat Restoration Program 2018 Monitoring Report with Recommendations by Ms. Debbie House, Interim Mono Basin Waterfowl Monitoring Program Director

In addition to these reports, the submittal also includes Section 1: the RY 2018-19 Status of Restoration Compliance Report, which summarizes the status of LADWP's compliance activities in the Mono Basin to date and planned activities for the upcoming runoff year.

111 N. Hope Street, Los Angeles, California 90012-2607 Mailing Address: PO Box 51111, Los Angeles, CA 90051-5700
Telephone (213) 367-4211 ladwp.com

The filing of these reports, along with the restoration and monitoring performed by LADWP in the Mono Basin, fulfills LADWP's requirements for RY 2018-2019, as set forth in Decision No. 1631 and the Orders.

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 Dr. Paul C. Pau, Eastern Sierra Issues Supervisor, at (213) 367-1187.

Sincerely,

Anselmo G. Collins
Director of Water Operations

PCP:jm
Enclosures
c/enc: Distribution List
Dr. Paul C. Pau

Mono Basin Distribution List
Runoff Year 2018-19

Mr. Erik Ekdahl Division of Water Rights State Water Resources Control Board 1001 I Street, 14th Floor Sacramento, CA 95814	Ms. Lisa Cutting Mono Lake Committee P.O. Box 29 Lee Vining, CA 93541
Ms. Amanda Montgomery Division of Water Rights State Water Resources Control Board 1001 I Street, 14th Floor Sacramento, CA 95814	Mr. Bartshe Miller Mono Lake Committee P.O. Box 29 Lee Vining, CA 93541
Mr. Scott McFarland Division of Water Rights State Water Resources Control Board 1001 I Street, 14th Floor Sacramento, CA 95814	Dr. Eric Huber California Trout Inc. P.O. Box 3442 Mammoth Lakes, CA 93546
Dr. William Trush Humboldt State University River Institute c/o Department of Environmental Science and Management 1 Harpst Street Arcata, CA 95521-8299	Mr. Richard Roos-Collins Water and Power Law Group 2140 Shattuck Avenue, Suite 801 Berkeley, CA 94704-1229
Mr. Ross Taylor 1254 Quail Run Court McKinleyville, CA 95519	Mr. Marshall S. Rudolph Mono County Counsel P.O. Box 2415 Mammoth Lakes, CA 93546
Mr. Jon C. Regelbrugge USDA Forest Service P.O. Box 148 Mammoth Lakes, CA 93546	Mr. Steve Parmenter Department of Fish and Wildlife 787 North Main Street, Suite 220 Bishop, CA 93514
Ms. Tamara Sasaki California Department of Parks and Recreation P.O. Box 266 Tahoma, CA 96142	Mr. Doug Smith Grant Lake Reservoir Marina P.O. Box 21 June Lake, CA 93529
Mr. Matthew Green California Department of Parks and Recreation P.O. Box 266 Tahoma, CA 96142	Board of Supervisors Mono County P.O. Box 715 Bridgeport, CA 93517

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 2019
Los Angeles Department of Water and Power

CONTENTS

Section 1

Status of Restoration Compliance Report

Status of Restoration Compliance Report (SORC)

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

May 2019

Los Angeles Department of Water and Power

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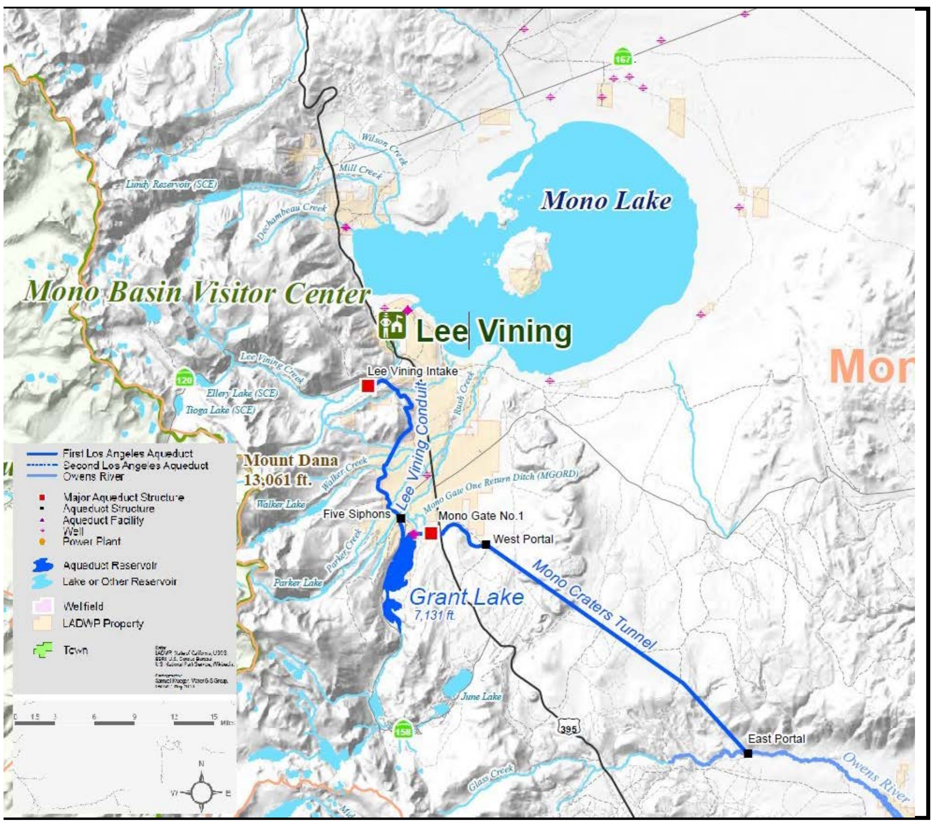
Introduction

Pursuant to State Water Resources Control Board (SWRCB) Decision 1631 and Order Nos. 98-05 and 98-07 (Orders), the Los Angeles Department of Water and Power (LADWP) is to undertake certain activities in the Mono Basin to be in compliance with the terms and conditions of its water right licenses 10191 and 10192. In particular, the Orders state that LADWP is to undertake activities to monitor stream flows, and to restore and monitor the fisheries, stream channels, and waterfowl habitat. This chapter includes the Status of Restoration Compliance Report, which summarizes the status of LADWP compliance activities in the Mono Basin to date. It is expected that the Water Board will amend LADWP's water rights license. Following SWRCB adoption of the amended license, the new requirements will be reflected in future SORC Reports.

Figure 1: Map of Mono Basin showing major Streams and LADWP facilities.



Status of Restoration Compliance Report

This document was first submitted as draft to the interested parties on April 1, 2019. It was developed to include a 21 day review period during which LADWP will review and address comments submitted by the interested parties. Following the 21 day review period, LADWP will finalize it as part of the May 2019 Status of Restoration Compliance Report as below.

Status of Restoration Compliance Report
State Water Resource Control Board Decision 1631 and Order Nos. 98-05 & 98-07

The Status of Restoration Compliance Report ("SORC Report") is organized into the following sections:

1. **Introduction** – Description of the SORC Report
2. **Definitions** – Explanations of what each category represents
3. **Updates from Previous SORC Report** – Changes over the past year
4. **Plans for the Upcoming Runoff Year** – Planned activities for the upcoming year
5. **Requirements** – Categories of the entire list of LADWP's requirements in the Mono Basin
6. **Completion Plans** – Long term plans for completing all requirements
7. **Ongoing Items Definitions** – Ongoing activities necessary for LADWP operations in the Mono Basin.

1. Introduction:

The SORC Report details the status of the Los Angeles Department of Water and Power's (LADWP) restoration requirements in the Mono Basin as outlined by the State Water Resources Control Board (SWRCB) Decision 1631 and Order Numbers 98-05 and 98-07, and any subsequent decision letters distributed by the SWRCB. This initial structure and content of the SORC report was cooperatively prepared by LADWP and the Mono Lake Committee (MLC) through an extensive series of staff discussions and a workshop held in the Mono Basin in August 2005. LADWP and MLC believe this report represents the most thorough and complete listing of Mono Basin restoration requirements and their current status available in a unified document. These requirements are categorized as ongoing, complete, in progress, incomplete or deferred as defined below in Section 2. The final section of the SORC Report details how LADWP plans to proceed with those items not listed as ongoing or completed (i.e. items in progress, incomplete, and/or deferred).

The SORC Report will be submitted by LADWP to SWRCB as part of the annual Compliance Reporting. By April 1 each year, LADWP will update and submit a draft SORC Report to the interested parties. Within 21 days of the draft submission, LADWP will accept comments on the draft SORC Report from the interested parties. Then, LADWP will finalize the SORC Report, incorporating and/or responding to comments. The final SORC Report will then be included into the final Compliance Reporting to SWRCB by May 15 of each year.

It is expected that the Water Board will amend LADWP's current water rights license following a CEQA analysis of proposed actions related to the Mono Basin settlement agreement. The new requirements are expected to take effect immediately after the Water Board issues an order, and those new requirements will be reflected in future SORC Reports. Any items no longer relevant under the new order will be moved to a new category "Eliminated" in the SORC. The new SORC will show both a new numbering system for all active items as well as the old numbering system for cross reference. Once agreement is reached on the items in the "eliminated" category, those items as well as the old numbering will no longer be shown in future SORC Reports.

2. Definitions:

Below are the definitions of the categories where each requirement has been grouped.

A. Ongoing — Items that are current and require continuous action (e.g. Maintain road closures in floodplains of Rush and Lee Vining Creeks)

B. Complete — Items that have been finalized (e.g. Rehabilitation of the Rush Creek Return Ditch)

C. In-Progress — Items started and not yet finalized because of time or the timeline extends into the future (e.g. Waterfowl monitoring and reporting)

D. Incomplete — Items not yet started or not complete because plans for completion not finalized.

E. Deferred — Items placed on hold which need input from the Stream Scientists and/or SWRCB before plans commence (e.g. Prescribed burn program)

3. Updates from Previous SORC Report:

Since the last SORC Report of May 15, 2018, there has been no change to the report and Section 4, the Plans for Runoff Year RY2018-19, will apply to RY2019-20.

4. Plans for the Upcoming Runoff Year:

During the upcoming runoff year, RY2019-20, LADWP plans to:

1. Continue with all requirements listed under Category A – Ongoing Items, as needed based on the runoff year.
2. Continue Category C – In-Progress Items C17 "Sediment Bypass for Parker Creek". Sediment bypass will continue in the non-Dry RY.
3. Continue Category C – In-Progress Items C18 "Sediment Bypass for Walker Creek". Sediment bypass will continue in the non-Dry RY.

5. Requirements:

This section lists and categorizes the individual requirements based on the status of each item. The requirements are derived from SWRCB Decision 1631, and/or Order Nos. 98-05 and 98-07, and/or any subsequent decision letters distributed by SWRCB. The requirements are either described in the cited section of the order and/or are described in the cited page of the specified plan and/or document (Stream Plan, Waterfowl Plan, GLOMP, etc.) that the Order references, and/or detailed in the SWRCB letter. Plans for completing in-progress, incomplete, and deferred items are further explained in Section 6, Completion Plans. Finally, plans for those items described as ongoing are detailed in Section 7, Ongoing Items Description.

<u>Category A – Ongoing Items</u>

1. Maintain road closures in floodplains of Rush and Lee Vining Creeks – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 71-75

2. Base flow releases – *Stream Management*
   Order 98-05 order 2.a.; GLOMP p. 2, table A

3. Low winter flow releases – *Stream Management*
   Order 98-05 order 2.b.

4. Annual operations plan – *Stream Management*
   Order 98-05 order 3; GLOMP p. 103, 104

5. Notification of failure to meet required flows – *Stream Management*
   Order 98-05 order 3

6. Grant operations and storage targets – *Stream Management*
   Order 98-05 order 1.a.; Decision 1631 order 1; GLOMP p. 84

7. Amount and pattern of export releases to the Upper Owens River – *Stream Management*
   Order 98-05 order 2; Decision 1631 order 7; GLOMP p. 84, 85

8. Diversion targets from streams – *Stream Management*
   Order 98-05 order 2; GLOMP p. 85

9. Export amounts dependent on Mono Lake level – *Stream Management*
   Decision 1631 order 6

10. Year type designation and guidelines – *Stream Management*
    Order 98-05 order 2; Decision 1631 order 3; GLOMP p. 87-96

11. Dry and wet cycle contingencies for stream restoration flows and base flows – *Stream Management*
    Order 98-05 order 2; GLOMP p. 97

12. Deviations from Grant Lake Operation Management Plan (GLOMP) – *Stream Management*
    Order 98-05 order 2; GLOMP p. 98, 99

13. Ramping rates – *Stream Management*
    Order 98-05 order 2; Decision 1631 order 2; GLOMP p. 90-96

14. Stream restoration flows and channel maintenance flows – *Stream Management*
    Order 98-05 order 1.a.

15. Salt Cedar eradication – *Waterfowl*
    Order 98-05 order 4.e.; Waterfowl Plan p. 27

16. Aerial photography every five years or following an extreme wet year event – *Monitoring*
    Order 98-05 order 1.b; Stream Plan p. 103

17. Make basic data available to public – *Monitoring*
    Order 98-05 order 1.b as revised by Order 98-07; Order 98-07 order 1.b(2); Stream Plan p. 110

18. Operation of Lee Vining sediment bypass – Stream Facility Modifications
    Order 98-05 order 2

19. Operation of the Rush Creek augmentation from the Lee Vining Conduit when necessary – *Stream Management*
    Order 98-05 order 2

20. Make data from all existing Mono Basin data collection facilities available on an internet web site on a same-day basis – *Stream Management*
    Order 98-05 order 2.c

Category B – Completed Items

1. Placement by helicopters of large woody debris into Rush Creek, completed fall 1999 – *Stream Work*
   Order 98-05 order 1; order 1.d.; Stream Plan p. 67, 68

2. Placement by helicopters of large woody debris into Lee Vining Creek, completed fall 1999 – *Stream Work*
   Order 98-05 order 1; order 1.d.; Stream Plan p. 67, 68

3. Rewater Rush Creek side channels in reach 3A, completed fall 1999 – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 68-71

4. Rewater Rush Creek side channel in reach 3B, completed fall 1999 with changes (see LADWP annual Compliance Reporting, May 2000) – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 68-71

5. Rewater Rush Creek side channel in reach 3D, completed fall 2002 with changes (see LADWP annual Compliance Reporting, May 2003) – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 68-71

6. Revegetate approximately 250 Jeffrey Pine trees on Lee Vining Creek, completed in 2000 – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 71-75

7. Revegetate willows on Walker Creek. No planting necessary in judgment of LADWP and MLC as area revegetated rapidly without intervention – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 71-75

8. Revegetate willows on Parker Creek. No planting necessary in judgment of LADWP and MLC as area revegetated rapidly without intervention – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 71-75

9. Limitations on vehicular access in Rush and Lee Vining Creek floodplains, completed fall 2003 – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 78-80

10. Removal of bags of spawning gravel, completed fall 2003 – *Stream Work*
    Order 98-05 order 1; Stream Plan p. 85, 86

11. Removal of limiter logs, completed 1996 – *Stream Work*
    Order 98-05 order 1; Stream Plan p. 86

12. Removal of Parker Plug, completed by California Department of Transportation 2000 – *Stream Work*
    Order 98-05 order 1; Stream Plan p. 87

13. Sediment bypass facility for Lee Vining Creek, completed winter 2005 – *Stream Facility Modifications*
    Order 98-05 order 1.f.

14. Flood flow contingency measures, completed by California Department of Transportation's Highway 395 improvements in 2002 – *Stream Management*
    Order 98-05 order 1; Stream Plan p. 76

15. Stream monitoring site selection, completed 1997 – *Monitoring*
    Order 98-05 order 2; Stream Plan p. 109

16. Waterfowl and limnology consultants, completed 2004 – *Monitoring*
    Order 98-05 order 4; Waterfowl Plan p. 27-29

17. Status report on interim restoration in Mono Basin, completed 2006 – *Other*
    Decision 1631 order 8.d (3)

18. Cultural resources investigation and treatment plan report to SWRCB, completed 1996 – *Other*
    Decision 1631 order 9, 10

19. Revegetate or assess the need to revegetate Rush Creek side channels in reach 3A five years after rewatering, assessed annually and reported in May 2006 Monitoring Report – *Stream Work*
Order 98-05 order 1; Stream Plan p. 71-75

20. Revegetate or assess the need to revegetate Rush Creek side channels in reach 3B five years after rewatering, assessed annually and reported in May 2006 Monitoring Report – *Stream Work*
Order 98-05 order 1; Stream Plan p. 71-75

21. Revegetate or assess the need to revegetate Rush Creek side channel in reach 3D and reported in May 2008 Monitoring Report – *Stream Work*
Order 98-05 order 1; Stream Plan p. 71-75

22. Rewater Rush Creek side channel 11 in reach 4C. Final review was conducted by the Stream Scientists. After presentation of the final review, LADWP followed the recommendations of the Stream Scientists not to do any action on the channel. This item is now approved by SWRCB and is therefore considered completed in 2008. – *Waterfowl*
Order 98-05 order 4.a., order 4.d.; Waterfowl Plan p. 22

23. Rewater Rush Creek side channel 14 in reach 4C. Final review was conducted by the Stream Scientists. After presentation of the final review, LADWP followed the recommendations of the Stream Scientists not to do any action on the channel. This item is now approved by SWRCB and is therefore considered complete in 2008. – *Stream Work*
Order 98-05 order 1; Stream Plan p. 68-71

24. Revegetate or assess the need to revegetate Rush Creek side channel 11 in reach 4C for five years following rewatering. LADWP followed the recommendations of the Stream Scientists not to do any action on the channel. This item is now approved by SWRCB and is therefore considered completed in 2008. – *Waterfowl*
Order 98-05 order 4.a., order 4.d.; Waterfowl Plan p. 22

25. Revegetate or assess the need to revegetate Rush Creek side channel 14 in reach 4C for five years after rewatering. LADWP followed the recommendations of the Stream Scientists not to do any action on the channel. This item is now approved by SWRCB and is therefore considered completed in 2008. – *Stream Work*
Order 98-05 order 1; Stream Plan p. 68-71

26. LADWP and MLC were to cooperatively revegetate pine trees on areas of Rush Creek and Lee Vining Creek including disturbed, interfluve, and upper terrace sites targeted from reach 3B through 5A on Rush Creek. In 2005, remaining suitable areas were assessed resulting in a map showing those areas where planting pine trees may be successful and would add to habitat complexity. LADWP and MLC investigated locations suitable for planting by LADWP and MLC staff and volunteers. Acceptable Jeffrey Pine seedlings were procured by LADWP and were planted by MLC and volunteers on all available suitable sites. This item is

considered complete and is moved to Category B "Completed Items." However, MLC may continue to water these seedlings. MLC may also plant cottonwoods with volunteers as opportunities arise – Stream Work Order 98-05 order 1; Stream Plan p. 71-75

27. Rewater Rush Creek side channel 8 in reach 4B, completed March 2007 – *Waterfowl.* The further rewatering of Rush Creek side channel complex 8 in reach 4B was deferred by the Stream Scientists. Final review is being conducted by McBain and Trush. After presentation of the final review, LADWP followed the recommendations of the Stream Scientists and SWRCB has approved the plan *Order 98-05 order 4.a., order 4.d; Waterfowl Plan p. 22*

28. Rehabilitation of the Rush Creek Return Ditch, completed 2002 – *Stream Facility Modifications.* Since then, vegetation growth has slightly reduced ditch capacity. To restore maximum capacity of 380 cfs, the return ditch embankments were raised.
Order 98-05 order 1, order 1.c.; Stream Plan p. 85, appendix III

Category C – In-Progress Items

1. Placement by hand crews of large woody debris into Rush Creek on an opportunistic basis based on stream monitoring team recommendations – *Stream Work*
Order 98-05 order 1; order 1.d.; Stream Plan p. 67, 68

2. Placement by hand crews of large woody debris into Lee Vining Creek on an opportunistic basis based on stream monitoring team recommendations – *Stream Work*
Order 98-05 order 1; order 1.d.; Stream Plan p. 67, 68

3. Grazing moratorium for 10 years, assessed annually and status reported in May 2009 Monitoring Report. Grazing moratorium to continue until further notice. – *Stream Management*
Order 98-05 order 1; Stream Plan p. 83

4. Grant Lake Operation Management Plan (GLOMP) preparation for revisions – *Stream Management*
Order 98-05 order 2; GLOMP p. 103, 104

5. Waterfowl project funding – *Waterfowl*
Order 98-05 order 4.b.

6. Salt Cedar eradication reporting– *Waterfowl*
Order 98-05 order 4.e.; Waterfowl Plan p. 27

7. Stream monitoring team to perform duties – *Monitoring*
Order 98-05 order 1.b as revised by Order 98-07

8. Stream monitoring reporting to the SWRCB – *Monitoring*

Order 98-05 order 1.b as revised by Order 98-07; Order 98-07 order 1.b(2); Stream Plan p. 110

9. Development, approval, and finalization of stream monitoring termination criteria for Walker and Parker Creeks – *Monitoring Order 98-07*

10. Development, approval, and finalization of stream monitoring termination criteria for Lee Vining and Rush Creeks – *Monitoring*
    Order 98-07

11. Hydrology monitoring and reporting – *Monitoring*
    Order 98-05 order 4; Waterfowl Plan p. 27

12. Lake limnology and secondary producers monitoring and reporting – *Monitoring*
    Order 98-05 order 4; Waterfowl Plan p. 27, 28

13. Riparian and Lake fringing wetland vegetation monitoring and reporting – *Monitoring*
    Order 98-05 order 4; Waterfowl Plan p. 27, 28

14. Waterfowl monitoring and reporting – *Monitoring*
    Order 98-05 order 4; Waterfowl Plan p. 28; LADWP's 2004 "Mono Lake Waterfowl Population Monitoring Protocol" submitted to SWRCB on October 6, 2004

15. Testing the physical capability for Rush Creek augmentation up to 150 cfs from the Lee Vining Conduit through the 5-Siphon Bypass facility – *Stream Management*
    Order 98-05 order 2; GLOMP p. 82, 83

16. Evaluation of the effects on Lee Vining Creek of Rush Creek augmentation for diversions up to 150 cfs through the Lee Vining Conduit – *Monitoring*
    Order 98-05 order 1.b.

17. Sediment bypass for Parker Creek – *Stream Facility Modifications*
    Order 98-05 order 1.f.

18. Sediment bypass for Walker Creek – *Stream Facility Modifications*
    Order 98-05 order 1.f.

<u>Category D – Incomplete Items</u>

None

<u>Category E – Deferred Items</u>

1. Recommend an Arizona Crossing or a complete road closure at the County Road Lee Vining Creek, if and when Mono County plans to take action – *Stream Work*
   Order 98-05 order 1; Stream Plan p. 78-80

2. Fish screens on all irrigation diversions – *Stream Facility Modifications Order 98-05 order 1; Stream Plan p. 84*

3. Prescribed burn program – *Waterfowl Order 98-05 order 4.b.(3)c.; Waterfowl Plan p. 25, 26*

4. Rewatering of Rush Creek side channel 1A in reach 4A.– *Stream Work Order 98-05 order 1; Stream Plan p. 68-71*

5. Assessing the need to revegetate the areas affected by the side channel openings for Rush Creek side channel 1A in reach 4A – *Stream Work; Order 98-05 order 1; Stream Plan p. 68-71*

6. Assessing the need to revegetate the areas affected by the side channel openings for Rush Creek side channel 4Bii in reach 4B. – Stream Work Order 98-05 order 1; Stream Plan p. 68-71

7. Assessing the need to revegetate the areas affected by the side channel openings for Rush Creek side channel 8 in reach 4B.

8. Stream monitoring for 8-10 years to inform peak flow evaluation and recommendations including the need for a Grant Lake Reservoir Outlet – *Monitoring Order 98-05 order 1.b as revised by Order 98-07*

6. Completion Plans:

The following descriptions detail how LADWP plans to fulfill SWRCB requirements in the Mono Basin for each item above not categorized as complete or ongoing. This section will be reviewed annually by LADWP for revisions to reflect progress towards completion.

Category C – In-Progress Items

Item C1 – During walking surveys, large woody debris will be placed into Rush Creek and will continue to be done on an opportunistic basis based on recommendations made by the Monitoring Team. This item will remain "In-Progress" until the Monitoring Team indicates that no further work is required. At that time, this item will be considered complete and will be moved to Category B "Completed Items".

Item C2 – During walking surveys, large woody debris will be placed into Lee Vining Creek and will continue to be done on an opportunistic basis based on recommendations made by the Monitoring Team. This item will remain "In-Progress" until the Monitoring Team indicates that no further work is required. At that time, this item will be considered complete and will be moved to Category B "Completed Items".

Item C3 – The grazing moratorium in the Mono Basin was in effect until 2009. At this time LADWP does not intend to allow grazing on its lands in the Mono Basin and will continue the moratorium in 2019. This item will remain in the Category C “In Progress”.

Item C4 – The Grant Lake Operation Management Plan (GLOMP) includes instructions to “review for revisions” every five years until Mono Lake reaches 6,391 feet above mean sea level. Although no revisions have been finalized to date, the plan was continuously under review. GLOMP is expected to be revised and replaced with “Mono Basin Operations Plan” (MBOP) after the SWRCB amends LADWP Water Rights licenses. This item will remain in Category C “In-Progress Items” until the final operation/management plan is approved by SWRCB. It is expected that a final plan will be developed after the Water Board order. Once the plan is approved, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C5 – LADWP is to make available a total of $275,000 for waterfowl restoration activities in the Mono Basin. 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. USFS has requested funds for a project estimated at $100,000. MLC has requested that the remainder of the funds be applied toward the total cost of the Mill Creek Return Ditch upgrade which would provide benefits for waterfowl habitat. The Mill Creek Return Ditch rehabilitation is a component of a Federal Energy Regulatory Commission (FERC) settlement agreement. These funds will continue to be budgeted by LADWP until such a time that they have been utilized. Currently, this money has been tentatively been included in the Settlement Agreement as part of Administrative of Monitoring Accounts to be administered by a Monitoring Administration Team (MAT). Once the full $275,000 has been utilized, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C6 – Progress of the salt cedar eradication efforts is reported in the annual reports following the vegetation monitoring efforts. This item will continue to be in progress until notice from SWRCB is received that LADWP’s obligation for this in the Mono Basin is complete. Once this notice is received, this item will be moved to Category B “Completed Items”.

Item C7 – The stream monitoring team continues to perform their required duties in the Mono Basin. This item will continue to be in progress until notice from SWRCB is received that LADWP’s obligation for funding and managing the monitoring team in the Mono Basin is complete. Once this notice is received, this item will be moved to Category B “Completed Items”, and LADWP will implement an appropriate monitoring program for the vegetation, stream morphology waterfowl, and fisheries.

Item C8 – Progress of the restoration efforts is reported in the annual reports. This item will continue to be in progress until notice from SWRCB is received that

LADWP’s obligation for this in the Mono Basin is complete. Once this notice is received, this item will be moved to Category B “Completed Items”.

Item C9 – The Stream Scientists have submitted final recommendations for termination criteria on Walker and Parker Creeks in 2007 to the SWRCB. There has been no decision from SWRCB. Once the termination criteria are finalized by the Stream Scientists and approved by SWRCB, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C10 – The Stream Scientists have submitted final recommendations for termination criteria on Lee Vining and Rush Creeks in 2007 to the SWRCB. There has been no decision from SWRCB. Once approved by SWRCB, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C11 – LADWP will continue to monitor and report on the hydrology of the Mono Basin including regular Mono Lake elevation readings, stream flows, and spring surveys until SWRCB approves that all or portions of the hydrology monitoring is no longer required. Once this occurs, all or portions of this item will be considered complete and will be moved to Category B “Completed Items”. Any portions of this requirement that are deemed to be ongoing by the SWRCB will be moved to Category A “Ongoing Items”.

Item C12 – LADWP will continue to monitor and report on the Mono Lake limnology and secondary producers until SWRCB approves that limnological monitoring is no longer required. Once this occurs, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C13 – LADWP will continue to monitor and report on the vegetation status in riparian and lake fringing wetland habitats, which is done every 5 years until SWRCB approves that vegetation monitoring is no longer required. Once this occurs, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C14 – LADWP will continue to monitor and report on the waterfowl populations in the Mono Basin until SWRCB approves that waterfowl monitoring is no longer required. Once this occurs, this item will be considered complete and will be moved to Category B “Completed Items”.

Item C15 – Testing augmentation of Rush Creek flows with water from Lee Vining Creek through the use of the Lee Vining Conduit is possible and can occur as needed as demonstrated during peak runoff in June 2005. The augmentation has been tested up to 100 cfs and the orders call for maximum augmentation to be 150 cfs. This will only be possible if adequate runoff is available in Lee Vining Creek after the peak operation is complete. Once augmentation is successfully tested through 150 cfs, this item will be moved to Category B “Completed Items”.

Item C16 – Evaluation of the effects of Rush Creek augmentation on Lee Vining Creek needs to be completed to cover diversions up to 150 cfs. Once the evaluation is completed, this item will be moved to Category B “Completed Items”.

Item C17 – Sediment bypass for Parker Creek is now in trial implementation stage. Once a plan is finalized by SWRCB and becomes part of LADWP’s operation plans, this item will be moved to Category A “Ongoing Items”.

Item C18 – Sediment bypass for Walker Creek is now in trial implementation stage. Once a plan is finalized by SWRCB and becomes part of LADWP’s operation plans, this item will be moved to Category A “Ongoing Items”.

Category D – Incomplete Items

None

Category E – Deferred Items

Item E1 – Pending further action by Mono County to improve the county road crossing at Lee Vining Creek, LADWP will write a letter to Mono County recommending an Arizona crossing at that point. Once LADWP writes this letter, or the parties agree that this is unnecessary; this item will be moved to Category B “Completed Items”.

Item E2 – LADWP was to place fish screens on all of its irrigation diversions in the Mono Basin. Subsequently LADWP ended all irrigation practices and hence does not need to install fish screens. If at a later date LADWP resumes irrigation, fish screens will be installed and this item will be moved to Category A “Ongoing Items”.

Item E3 – LADWP began a prescribed burn program with limited success. LADWP requested to remove this item from the requirements and the SWRCB instead ruled that the prescribed burn program will be deferred until Mono Lake reaches 6,391 ft. Once Mono Lake reaches 6,391 ft. LADWP will reassess the prescribed burn. Based on results from the assessment, LADWP will either reinstate the program or request relief from the SWRCB from this requirement. If LADWP reinstates the program this item will be moved to Category C “In-Progress Items”, however if LADWP requests, and is granted relief from this SWRCB requirement, this item will be moved to Category B “Completed Items”.

Item E4 - Rewatering of Rush Creek side channel 1A in reach 4A. Final review was conducted by the Stream Scientists. After presentation of the final review, LADWP followed the recommendations of the Stream Scientists not to do any action on the channel and was awaiting final decision by SWRCB. This item was approved by SWRCB and was therefore considered completed in 2008. Further work on Channel 1A was to be considered in the future if deemed appropriate. In 2014, as part of the pending new license, it has been included to be done in the future. Until the SWRCB approves the Settlement Agreement and amends LADWP’s license, it will be placed in Category E – “Deferred Item”.

Item E5 - Assessing the need to revegetate the areas affected by the side channel openings for Rush Creek side channel 1A in reach 4A will occur for five years following rewatering. LADWP followed the recommendations of the Stream Scientists not to do any action on the channel and was awaiting final decision by SWRCB. This item was approved by SWRCB and was therefore considered completed in 2008. Until the SWRCB approves the Settlement Agreement and amends LADWP's license, it will be placed in Category E – "Deferred Item".

Item E6 - Assessing the need to revegetate the areas affected by the side channel openings for Rush Creek side channel 4Bii in reach 4B five years following rewatering (2007) occurred in the summer of 2012. The results from the assessment following the fifth year after rewatering was reported in Section 4 of the 2013 report. The final assessment concluded that satisfactory revegetation has occurred through natural processes and was considered complete and was moved to Category B "Completed Items". Until the SWRCB approves the Settlement Agreement and amends LADWP's license, it will be placed in Category E – "Deferred Item".

Item E7 - Assessing the need to revegetate the areas affected by the side channel openings for Rush Creek side channel 8 in reach 4B five years following rewatering (2007) occurred in the summer of 2012. The results from the assessment following the fifth year after rewatering were reported in Section 4 of the 2013 report. The final assessment concluded that satisfactory revegetation has occurred through natural processes and was considered complete and was moved to Category B "Completed Items". Until the SWRCB approves the Settlement Agreement and amends LADWP's license, it will be placed in Category E – "Deferred Item".

Item E8 – The stream monitoring team is to evaluate the restoration program after "no less than 8 years and no more than 10 years" from the commencement of the restoration program. This evaluation is to cover the need for a Grant Lake outlet, Rush Creek augmentation, and the prescribed stream flow regime. According to SWRCB Order Nos. 98-05 and 98-07, evaluation of LADWP's facilities to adequately provide proper flows to Rush Creek "*shall take place after two data gathering cycles but no less than 8 years nor more than 10 years after the monitoring program begins*". The Monitoring Team submitted final recommendation, on April 30, 2010. LADWP had 120 days after receiving the recommendation from the monitoring team to determine whether to implement the recommendation of the monitoring team. On July 28, 2010, LADWP submitted a Feasibility Report evaluating the recommendations. In September 2013, LADWP entered into a Settlement Agreement with the Stakeholders and this Agreement is pending SWRCB's approval via an amended Water Rights license. Until the SWRCB approves the Settlement Agreement and amends LADWP's license, it will be placed in Category E – "Deferred Item".

7. Ongoing Items Description:

See Section 5 for references where each requirement originates.

Category A – Ongoing Items

Item A1 – *Road closures*. Periodically LADWP personnel will visit all road closures performed by LADWP in accordance with SWRCB Order No. 98-05, Order 1, in the Lower Rush and Lee Vining Creek areas to assess their effectiveness. Where evidence exists that a road closure is ineffective, LADWP will improve the road closures through means such as additional barriers.

Item A2 – *Base flow releases*. LADWP normally will control flow releases from its facilities into Lower Rush, Parker, Walker, and Lee Vining Creeks according to agreed upon flow rate requirements as set forth in the SWRCB Decision 1631, Order Nos. 98-05 and Order 98-07, the Grant Lake Operations Management Plan, and any subsequent operations plans and decisions made by the SWRCB.

Item A3 – *Low winter flow releases*. Per the California Department of Fish and Wildlife recommendations, and SWRCB Order No. 98-05, order 2.b., LADWP will maintain winter flows into Lower Rush Creek below 70 cfs in order to avoid harming the Rush Creek fishery.

Item A4 – *Annual operations plan*. Per SWRCB Order No. 98-05, order 3, LADWP will distribute an annual operations plan covering its proposed water diversions and releases in the Mono Basin. Presently the requirement is to distribute this plan to the SWRCB and all interested parties by May 15 of each year.

Item A5 – *Notification of failure to meet flow requirements*. Per SWRCB Order No. 98-05, order 3, and SWRCB Decision 1631, order 4, if at the beginning of the runoff year, for any reason, LADWP believes it cannot meet SWRCB flow requirements, LADWP will provide a written explanation to the Chief of the Division of Water Rights by May 1, along with an explanation of the flows that will be provided. If unanticipated events prevent LADWP from meeting SWRCB Order No. 98-05 Stream Restoration Flow requirements, LADWP will notify the Chief of the Division of Water Rights within 20 days and provide a written explanation of why the requirement was not met. LADWP will provide 72 hours notice and an explanation as soon as reasonably possible for violation of SWRCB Decision 1631 minimum instream flow requirements.

Item A6 – *Grant storage targets*. LADWP will operate its Mono Basin facilities to maintain a target storage elevation in Grant Lake Reservoir between 30,000 and 35,000 acre-feet at the beginning and end of the runoff year. LADWP will seek to have 40,000 acre-feet in Grant Reservoir on April 1 each year at the beginning of wet and extreme wet years.

Item A7 – *Export release patterns to the Upper Owens River*. Per SWRCB Decision 1631, order 7, and SWRCB Order No. 98-05, order 2, LADWP will make exports from the Mono Basin to the Upper Owens River in a manner that will not have a combined flow rate below East Portal above 250 cfs. LADWP will perform ramping of exports at 20% or 10 cfs, whichever is greater, on the ascending limb, and 10% or 10 cfs, whichever is greater, on the descending limb of the hydrograph as measured at the Upper Owens River.

Item A8 – *Diversion targets from streams*. Per the 1996 GLOMP, diversion targets for exports from the Mono Basin will be divided between Rush, Lee Vining, Parker and Walker Creeks in the following manner. During all years except dry and extremely wet years, LADWP will seek to divert one-third to one-half of the export amount from Lee Vining Creek, with the remaining water coming from Rush Creek. Only during dry years when 16,000 acre-feet of export is permitted, LADWP will seek to divert from Parker and Walker Creeks. During extremely wet years, all exports will come from diversions off of Rush Creek. Parker and Walker Creeks are expected to be flow through after the SWRCB approves the Settlement Agreement and amends LADWP Water Rights licenses.

Item A9 – *Export amounts dependent on Mono Lake level*. LADWP export amounts follow those ordered by SWRCB Decision 1631, order 2.

Item A10 – *Year type designation and guidelines*. Per SWRCB Decision 1631, order 4, SWRCB Order No. 98-05, and GLOMP, LADWP will perform runoff year forecasts for the Mono Basin with preliminary forecasts being conducted on February 1, March 1, and April 1, with the forecast being finalized on or around May 1 if necessary. LADWP developed a draft May 1 forecast methodology without a need for May snow surveys. When Gem Pass snow pillow measures show an increase in water content between April 1 and May 1, the percentage change experienced by the pillow will be applied to all of the April 1st snow course survey measurements used in calculating the runoff. A slight adjustment to the calculation may be made for dry years. Additionally, the May 1st forecast will have measured April values.

Item A11 – *Dry and wet cycle contingencies for stream restoration flows and base flows*. During consecutive dry years LADWP will release channel maintenance flows (CMF) every other year. The CMF will commence in the second consecutive dry year. The channel maintenance flows for Rush Creek will be 100 cfs for five days, and for Lee Vining Creek it will be 75 cfs for five days. Ramping rates will be 10 cfs per day. The occurrence of a year type other than a dry year will terminate the dry year cycle. During consecutive wet years, LADWP will increase base flows above the minimum flow rate every other year. The increased base flows will commence in the second consecutive wet year. The occurrence of a year type other than a wet year will terminate the wet year cycle.

Item A12 – *Deviations from Grant Lake Operation Management Plan (GLOMP)*. LADWP must maintain operational flexibility to adjust or react to unpredictable circumstances.

Item A13 – *Ramping rates*. LADWP will continue to operate its Mono Basin facilities in order to provide SWRCB ramping flow requirements for Lee Vining, Parker, Walker, and Rush Creeks.

Item A14 – *Stream restoration flows and channel maintenance flows*. LADWP will continue to operate its Mono Basin facilities in order to provide peak flow requirements for Lee Vining, Parker, Walker, and Rush Creeks.

Item A15 – *Salt Cedar eradication*. LADWP will continue assisting in a Mono Basin wide effort to eradicate Salt Cedar (*Tamarisk*), and will continue to report on these efforts.

Item A16 – *Aerial Photography.* LADWP will capture aerial and/or satellite imagery of the Mono Basin (Stream Plan, 1" = 6,000' scale; SWRCB Order No. 98-05, Section 6.4.6(4), 1:6,000 scale) every five years or following an extreme wet year event, which resets the five year clock.

Item A17 – *Make basic data available to public*. Per SWRCB Order 98-05, Order 1.b., as revised by SWRCB Order No. 98-07, order 1.b(2), LADWP will continue to make all basic monitoring data available to the public.

Item A18 – *Operation of Lee Vining sediment bypass*. In order to bypass sediment past the Lee Vining diversion facility, LADWP will operate the Lee Vining Conduit control gate to assist with ramping flows towards peak with the intention of having it be in the completely open position while peak flows are passing the diversion facility. After peak flows have passed the facility, the Lee Vining Conduit control gate will slowly close assisting with ramping flows back down towards base flow condition.

Item A19 – *Operation of the Rush Creek augmentation from the Lee Vining Conduit when necessary*. At times when peak flow requirements in Rush Creek exceed facility capacities, and Grant Lake Reservoir is not spilling, LADWP will operate the Lee Vining Conduit 5-Siphon Bypass to bring water from Lee Vining Creek to Rush Creek to augment flows to the required levels.

Item A20 – Data from existing Mono Basin data collection facilities is available on a same-day basis on the LADWP.com internet web site. The data collection and reporting works, as with any other system, can experience periodic short term communication problems and/or technical difficulties, which may result in incorrect readings. LADWP will continue to monitor the data posting on a daily basis and will work to troubleshoot and correct problems as soon as possible. LADWP will continue to improve the data collection, computer, and communication systems as new technology(ies) become available.

Section 2

Mono Basin Operations

Section 2

Mono Basin Operations

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

May 2019

Los Angeles Department of Water and Power

Table of Contents **Page No.**

I. Introduction

Pursuant to State Water Resources Control Board (SWRCB) Decision 1631 and Order Nos. 98-05 and 98-07 (Orders), the Los Angeles Department of Water and Power (LADWP) undertakes certain activities in the Mono Basin in compliance with the terms and conditions of its water right licenses 10191 and 10192. In addition to restoration and monitoring activities covered in this report, LADWP also reports on certain required operational activities.

II. Summary of Mono Basin RY 2018-19 Operations

A. Rush Creek

The runoff from Rush Creek was approximately 44,367 AF which amounts to the total water delivered to Grant Lake Reservoir (GLR)'s 'Damsite'. The highest flow of 293.10 cfs occurred on June 16, 2018.

Rush Creek flows below 'the Narrows', which consist of Rush Creek releases (Return Ditch, Spill, and 5-Siphons augmentation) combined with Parker and Walker Creek flows, had an approximate total of 61,842 AF. This flow terminated into Mono Lake.

RY 2018 was forecasted as an NORMAL year type but because the level of GLR was low (18,931.5 AF storage) on April 1, 2018, peak flows followed Guideline C for a Dry-Normal II year: 250 cfs for 5 days.

1. Rush Creek Augmentation

To meet high flow targets for lower Rush Creek, LADWP must at times employ facilities in addition to the Mono Gate One Return Ditch (MGORD) which has a 380 cfs capacity limit. During wetter years, LADWP utilizes one or both of its additional facilities to release higher peak flows. These facilities include the 5-Siphons bypass, which can release up to 100 cfs from Lee Vining Creek, and the GLR Spillway which can release large reservoir spills into lower Rush Creek during the wetter years.

<u>5-Siphons Bypass</u>
RY 2018 was forecasted as an NORMAL year type but because the level of GLR was low (18,931.5 AF storage) on April 1, 2018, peak flows followed Guideline C for a Dry-Normal II year: 250 cfs for 5 days. The MGORD, at a maximum capacity of 380 cfs, were able to accommodate the prescribed peak flows, therefore 5-Siphons were not utilized.

<u>Grant Reservoir Spill</u>
Grant did not spill during RY 2018.

B. Lee Vining Creek

RY 2018 was forecasted as a NORMAL year type and as such, following Guideline 'D', with baseflows of 54 cfs from April 1 to September 30, and 40 cfs from October 1 to March 31, 2019. Flows above 54 cfs were diverted to Grant lake Reservoir.
Lee Vining Creek had its highest flow on April 8, 2018 at 253 cfs. Total runoff for the year was approximately 43,229 AF.

C. Dry Cycle Channel Maintenance Flows

RY 2018 was forecasted as a NORMAL year type, therefore dry cycle channel maintenance flows (CMF) were not required in accordance with the GLOMP.

D. Parker and Walker Creeks

Parker and Walker were operated as pass through for RY 2018.

Parker Creek had its highest flow on April 8, 2018 at 46.56 cfs. Total runoff for the year was approximately 9,495 AF.

Walker Creek had its highest flow on June 14, 2018 at 41.53 cfs. Total runoff for the year was approximately 4,905 AF.

E. Grant Lake Reservoir

Grant Lake began the runoff year at approximately 18,931.50 AF (7,100.01 ft AMSL). The reservoir did not spill during the RY 2018. Final storage volume by the end of the RY of March 31, 2019 was approximately 28,122.40 AF (7,111.04 ft AMSL).

F. Exports during RY 2018-19

During RY 2018, Mono Lake elevations were within the 6,381 ft – 6,382 ft range, allowing for up to 16,000 AF of exports per D1631. LADWP exported 15,937 AF total from the Mono Basin, which is below the allowed 16,000 AF.

G. Mono Lake Elevations during RY 2018-19

In RY 2018, Mono Lake elevations were as shown in the following table. The Lake elevation was at 6,381.9 ft AMSL at the beginning of the runoff year, and ended the runoff year at 6,381.9 ft AMSL.

RY 2018-19 Mono Lake Elevation Readings

April 1, 2018	6,381.9
May 1, 2018	6,381.9
June 1, 2018	6,382.1
July 1, 2018	6,382.1
August 1, 2018	6,382.1
September 1, 2018	6,381.8
October 1, 2018	6,381.5
November 1, 2018	6,381.3,
December 1, 2018	6,381.3
January 1, 2019	6,381.3
February 1, 2019	6,381.6
March 1, 2019	6,381.9
April 1, 2019	6,381.9

III. Proposed Mono Basin Operations Plan RY 2019-20

A. Forecast for RY 2019-20

The Mono Basin Operations Plan for RY 2019-20 through September 30, 2019 will follow the Temporary Urgency Change Petitions (TUCPs) for a "WET" year category, as approved by the SWRCB on April 16, 2019. (**Attachment 1**) The Mono Basin's April 1st forecast for Runoff Year (RY) 2019 for April to March period is 171,900 acre-feet (AF), or 144 percent of average using the 1966-2015 long term mean of 119,103 AF (**Attachment 2**). This value puts the year type within the "WET" category.

LADWP will submit a timely Temporary Change Petition application to the SWRCB for the Mono Basin Operations Plan from October 1, 2019 to March 31, 2019.

The following forecasts are subjected to change as operations for the second-half of the runoff year have not been defined.

B. Grant Lake Reservoir

Grant Lake Reservoir (GLR) storage volume was 28,041 AF, corresponding to a surface elevation of 7,110.95 feet above mean sea level (AMSL) at the start of the runoff year. Using the closest available representative historical inflow data (2005 runoff year at 147 percent of normal), and above specified flows, GLR's profile is projected to be as shown in **Attachment 3**. Forecasted scenarios will be relatively close only if this year's hydrology turns out to be similar to the hydrology of the selected historical runoff year. Operations are subject to change with variations in actual hydrology during the upcoming runoff year.

C. Expected Mono Lake Elevations during RY 2019-20

Mono Lake began this runoff year at 6,381.9 ft AMSL where it is forecasted to increase and end the runoff year at approximately 6,384.0 ft AMSL (**Attachment 4**).

ATTACHMENTS

Attachment 1

STATE OF CALIFORNIA
CALIFORNIA ENVIRONMENTAL PROTECTION AGENCY
STATE WATER RESOURCES CONTROL BOARD

DIVISION OF WATER RIGHTS

In the Matter of Licenses 10191 and 10192 (Applications 8042 and 8043)

Los Angeles Department of Water and Power

ORDER APPROVING TEMPORARY URGENCY CHANGES

SOURCES: Rush Creek, Lee Vining Creek, Parker Creek, and Walker Creek

COUNTY: Mono

BY THE DEPUTY DIRECTOR FOR WATER RIGHTS:

1.0 SUBSTANCE OF THE TEMPORARY URGENCY CHANGE PETITIONS

On January 24, 2019, the State Water Resources Control Board (State Water Board) received Temporary Urgency Change Petitions (TUCPs) pursuant to California Water Code section 1435 from the Los Angeles Department of Water and Power (LADWP) requesting approval of temporary changes to its water right Licenses 10191 and 10192 (Applications 8042 and 8043).

On March 22, 2019, the State Water Board received proposed amendments to the TUCPs from LADWP. With the amended TUCPs, LADWP requests authorization to temporarily deviate from Stream Restoration Flow requirements as outlined in the State Water Board's Decision 1631 (D-1631) and Order 98-05 for Rush, Lee Vining, Parker, and Walker Creeks and instead follow the Stream Ecosystem Flows (SEFs) in the Draft Amended Licenses 10191 and 10192. The purpose of the temporary changes to the flow requirements is to collect data, and to test and evaluate the effects on resources from the implementation of the SEFs. The proposed amendments to the TUCPs will cover the appropriate water-year type starting from the approval date of this Order until September 30, 2019.

The temporary flow changes and the amended TUCPs are supported by the California Trout, Inc. (CalTrout), the Mono Lake Committee (MLC), and the State Water Board-approved stream monitoring team (Stream Scientists).

The temporary flow modifications proposed by LADWP will not increase LADWP's annual export of 16,000 acre-feet[1] as specified in D-1631.

2.0 BACKGROUND

2.1 State Water Board Decision 1631, Orders WR 98-05 and WR 98-07, and Licenses 10191 and 10192

In Decision 1631 (D-1631), the State Water Board modified Licenses 10191 and 10192 for the purpose of establishing instream flow requirements below LADWP's points of diversion on four affected streams tributary to Mono Lake. The decision also established conditions to protect public trust resources at

[1] 16,000 acre-feet may be exported annually when Mono Lake elevation is at or above 6,380 feet and below 6,391 feet.

Mono Lake. State Water Board Orders WR 98-05 and WR 98-07 (Orders) amended D-1631. Pursuant to D-1631 and the subsequent Orders, LADWP is required to conduct fisheries studies and stream monitoring activities until the program (or elements thereof) is terminated by the State Water Board. LADWP has been conducting fisheries studies and stream monitoring for over 20 years. These activities are conducted by the Stream Scientists who: (a) oversee implementation of the stream monitoring and restoration program, and (b) evaluate the results of the monitoring program and recommend modifications as necessary. In the Stream Scientists' April 30, 2010 *Synthesis of Instream Flow Recommendations Report* (Synthesis Report), they recommended modification of the flow regime and other aspects of the Mono Basin stream monitoring and restoration program.

2.2 Description of the Temporary Urgency Changes

The basis of temporary changes to the flow requirements is to allow LADWP to collect data, and to test and evaluate the effects on resources from the implementation of the SEFs, as identified in the *Mono Basin Operations Plan Under The Amended TUCP*, dated March 22, 2019. The TUCPs request the following temporary changes:

1. Rush Creek - The Mono Basin's April 1st forecast for Runoff Year (RY) 2019-2020 is projected to be either an Extreme-Wet, Wet, or Wet/Normal water-year type. Rush Creek's SEFs will be set to the appropriate water-year type and follow either Table 1A for an Extreme-Wet, Table 1B for a Wet, or Table 1C for a Wet/Normal water-year type (see Tables on pages 6-8).
2. Lee Vining Creek – The SEFs for Lee Vining Creek will follow Table 2A for an Extreme-Wet, Wet, or Wet/Normal water-year type (see Table on page 9).
3. Parker Creek – All flow will be continuously bypassed.
4. Walker Creek - All flow will be continuously bypassed.

It has been noted that the current infrastructure may not allow LADWP to deliver the magnitude of flows and duration for Rush Creek's SEFs listed in Tables 1A, 1B, and 1C when flows exceed 380 cubic feet per second (cfs). LADWP also acknowledged that Lee Vining Creek's flows listed in Table 2A will be implemented to the extent that the current infrastructure and upstream operations allows and operate to ensure flows in Lee Vining Creek do not drop below the minimum specified flows as outlined in Table 2A. An exception to the flows in Table 2A will be made in September 2019 during fish monitoring activities where Lee Vining Creek flows will be set to 28 cfs for up to two weeks in order to ensure the safety of the Stream Scientists and LADWP biologists performing the fish monitoring activities.

LADWP will communicate with Mono Basin parties (MLC, CalTrout, California Department of Fish and Wildlife), the Stream Scientists, and the State Water Board during the TUCP's authorized period to coordinate and gain input as SEFs proceed. Specifically, a conference call will be scheduled within a reasonable time of the April runoff forecast to discuss final water year type, operations plan, address questions, and Stream Scientist input that may result from the operations plan. LADWP will also provide reasonable communication to update parties, answer questions, and address unforeseen challenges as SEFs are delivered according to the April 1 forecast for RY 2019-20.

3.0 COMPLIANCE WITH CALIFORNIA ENVIRONMENTAL QUALITY ACT

LADWP, as Lead Agency pursuant to the California Environmental Quality Act (CEQA), prepared a Notice of Exemption for the *Mono Basin Temporary Operation Petition to State Water Resources Control Board* on January 3, 2019. LADWP found that the change is categorically exempt from CEQA, as the project is for the use of existing facilities with negligible or no expansion of existing use, for the purpose of maintaining fish and wildlife habitat areas, maintaining stream flows, and protecting fish and wildlife resources. (14 Cal. Code Regs. § 15301(i).).

The State Water Board has reviewed the information submitted by LADWP and has determined that the petitions qualify for an exemption under CEQA. The State Water Board will issue a Notice of Exemption for the temporary urgency change petitions.

4.0 PUBLIC NOTICE OF TEMPORARY URGENCY CHANGE PETITIONS

On April 5, 2019, the State Water Board issued a public notice of the temporary urgency changes pursuant to Water Code section 1438, subdivision (a). The comment period expires on May 6, 2019. Pursuant to Water Code section 1438, subdivision (b)(1), LADWP is required to publish the notice in a newspaper having a general circulation and published within the counties where the points of diversion are located. LADWP published the notice on April 4, 2019 in the Mammoth Times. The State Water Board posted the notice of the temporary urgency changes and the TUCPs (and accompanying materials) on its website and distributed the notice through its electronic notification system. Pursuant to Water Code section 1438(a), the State Water Board may issue a temporary urgency change order in advance of the required notice period.

5.0 COMMENTS REGARDING THE TEMPORARY URGENCY CHANGE PETITIONS

On January 22, 2019, LADWP copied the initial TUCPs to interested parties including the California Department of Fish and Wildlife, CalTrout, MLC, and the Stream Scientists. On February 1, 2019, MLC commented on the proposed TUCPs. MLC recommended that the State Water Board approve implementation of the interim SEFs for 180 days with the option for renewal. MLC also recommended that all elements of draft Licenses 10191 and 10192 terms and conditions 11 (Stream Ecosystem Flows) including tables, 12 (Grant Lake Operations), and 15 (Annual Operations Plan) including the collaborative planning, Stream Scientists input, and monthly reporting elements be implemented. MLC stated that interim implementation of the SEFs in 2019 will benefit the restoration of Rush, Lee Vining, Walker, and Parker Creeks and help with Grant Lake Reservoir management as well.

On February 6, 2019, a Mono Lake stakeholders meeting/conference call was held at the State Water Board's office which initiated the discussion on the TUCPs and action items for the coordination and resubmittal of amended TUCPs. On March 14, 2019, LADWP discussed the proposed amendments to the TUCPs in a conference call with the MLC, CalTrout, and Stream Scientists and there was a consensus to support the amended TUCPs. On March 22, 2019, LADWP submitted the amended TUCPs to the State Water Board.

6.0 CRITERIA FOR APPROVING THE PROPOSED TEMPORARY URGENCY CHANGES

Water Code section 1435 provides that a permittee or licensee who has an urgent need to change the point of diversion, place of use, or purpose of use from that specified in the permit or license may petition for a conditional temporary change order. The State Water Board's regulations set forth the filing and other procedural requirements applicable to TUCPs (Cal. Code Regs., tit. 23, §§ 805, 806.) The State Water Board's regulations also clarify that requests for changes to permits or licenses other than changes in point of diversion, place of use, or purpose of use may be filed, subject to the same filing and procedural requirements that apply to changes in point of diversion, place of use, or purpose of use. (*Id.*, § 791, subd. (e))

Before approving a temporary urgency change, the State Water Board must make the following findings:

1. The Petitioner has an urgent need to make the proposed change;
2. The proposed change may be made without injury to any other lawful user of water;
3. The proposed change may be made without unreasonable effect upon fish, wildlife, or other instream beneficial uses; and
4. The proposed change is in the public interest.

(Wat. Code, § 1435, subd. (b)(1-4).)

6.1 Urgency of the Proposed Change

Under Water Code section 1435, subdivision (c), an "urgent need" means "the existence of circumstances from which the State Water Board may in its judgment conclude that the proposed temporary change is necessary to further the constitutional policy that the water resources of the state be put to beneficial use to the fullest extent of which they are capable and that waste of water be prevented . . . ." However, the State Water Board shall not find the need urgent if it concludes that the petitioner has failed to exercise due diligence in petitioning for a change pursuant to other appropriate provisions of the Water Code. (Ibid.)

In this case, there is an urgent need for the proposed change in the license conditions regarding fish flows for the purpose of furthering protection of public trust resources.

6.2 No Injury to Any Other Lawful User of Water

There are no known lawful users of water that will be affected by the proposed changes to instream flows. Accordingly, granting these TUCPs will not result in injury to any other lawful users of water

6.3 No Unreasonable Effect upon Fish, Wildlife, or Other Instream Beneficial Uses

As described above, MLC have indicated that the temporary urgency will benefit the restoration of Rush, Lee Vining, Walker, and Parker Creeks and help with Grant Lake Reservoir management. No other fish or wildlife resources are implicated by the proposed change; accordingly, the proposed change will not have unreasonable effects upon fish and wildlife resources.

6.4 The Proposed Change is in the Public Interest

The proposed change would assist LADWP in maintaining the fishery resources in good condition. Maintenance of the fishery is in the public interest.

In light of the above, I find in accordance with Water Code section 1435, subdivision (b)(4) that the proposed change is in the public interest, including findings to support change order conditions imposed to ensure that the change is in the public interest.

Pursuant to Water Code section 1439, the State Water Board shall supervise diversion and use of water under this temporary change order for the protection of all other lawful users of water and instream beneficial uses.

7.0 STATE WATER BOARD DELEGATION OF AUTHORITY

On June 5, 2012, the State Water Board adopted Resolution 2012-0029, delegating to the Deputy Director for Water Rights the authority to act on petitions for temporary urgency change. This Order is adopted pursuant to the delegation of authority in section 4.4.1 of Resolution 2012-0029.

8.0 CONCLUSIONS

The State Water Board has adequate information in its files to make the evaluation required by Water Code section 1435.

I conclude that, based on the available evidence:

1. The Petitioner has an urgent need to make the proposed changes;
2. The proposed changes will not operate to the injury of any other lawful user of water;

3. The proposed changes, with conditions set forth in this Order, will not have an unreasonable effect upon fish, wildlife, or other instream beneficial uses; and

4. The proposed changes are in the public interest.

ORDER

NOW, THEREFORE, IT IS ORDERED THAT: the petitions filed by the LADWP for temporary urgency changes in Licenses 10191 and 10192 are approved, and this approval is effective from the date of this Order to September 30, 2019. All existing terms and conditions in Licenses 10191 and 10192 remain in effect, except as temporarily amended by the following terms.

1. For protection of fish in Rush and Lee Vining Creeks, LADWP shall bypass flow below the point of diversion at the flows specified in the tables below for the appropriate water year type. The SEFs provided under this requirement shall remain in the stream channel and not be diverted for any other use.

2. LADWP shall submit to the Deputy Director for Water Rights on a monthly basis a written report that summarizes all activities conducted to ensure compliance with the requirements of this Order. The first monthly report is due at the end of the first complete month of this Order. LADWP shall submit a final report summarizing overall compliance with this Order no later than November 1, 2019.

3. This Order does not authorize any act that results in the taking of a threatened or endangered species, or any act that is now prohibited, or becomes prohibited in the future, under either the California Endangered Species Act (Fish and Game Code sections 2050 to 2097) or the federal Endangered Species Act (16 U.S.C.A. sections 1531 to 1544). If a "take" will result from any act authorized under this Order, the licensee shall obtain authorization for an incidental take permit prior to construction or operation. Licensee shall be responsible for meeting all requirements of the applicable Endangered Species Act for the temporary urgency change authorized under this Order.

4. The State Water Board shall supervise the diversion and use of water under this Order for the protection of legal users of water and instream beneficial uses and for compliance with the conditions. Petitioner shall allow representatives of the State Water Board reasonable access to the project works to determine compliance with the terms of this Order.

5. The State Water Board reserves jurisdiction to supervise the temporary urgency changes under this Order, and to coordinate or modify terms and conditions, for the protection of vested rights, fish, wildlife, instream beneficial uses and the public interest as future conditions may warrant.

6. The temporary urgency changes authorized under this Order shall not result in creation of a vested right, even of a temporary nature, but shall be subject at all times to modification or revocation in the discretion of the State Water Board. The temporary urgency changes approved in this Order shall automatically expire September 30, 2019, unless earlier revoked.

STATE WATER RESOURCES CONTROL BOARD

ORIGINAL SIGNED BY:

Erik Ekdahl, Deputy Director
Division of Water Rights

Dated: APR 16 2019

TABLE 1A: RUSH CREEK STREAM ECOSYSTEM FLOWS FOR EXTREME-WET YEARS

Hydrograph Component	Timing	Flow Requirement	Ramping Rate
Spring Baseflow	April 1 – April 30	40 cfs	Maximum: 10% or 10 cfs*
Spring Ascension	May 1 – May 15	40 cfs ascending to 80 cfs	Target: 5% Maximum: 25%
Spring Bench	May 16 – June 11	80 cfs	Maximum: 20%
Snowmelt Ascension	June 12 – June 22	80 cfs ascending to 220 cfs	Target: 10% Maximum: 20%
Snowmelt Bench	June 23 – August 10	220 cfs	Maximum Ascending: 20% Maximum Descending: 10% or 10 cfs*
Snowmelt Flood and Snowmelt Peak	Starting between June 23 and July 19 with the 5-day peak between June 29 and July 29	220 cfs ascending to 750 cfs, 750 cfs for 5 days, 750 cfs descending to 220 cfs	Target Ascending: 20% Maximum Ascending: 40% Maximum Descending: 10% or 10 cfs*
Medium Recession (Node)	August 11 – August 25	220 cfs descending to 87 cfs	Target: 6% Maximum: 10% or 10 cfs*
Slow Recession	August 26 – September 30	87 cfs descending to 30 cfs	Target: 3% 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 1B: RUSH CREEK STREAM ECOSYSTEM FLOWS FOR WET YEARS

Hydrograph Component	Timing	Flow Requirement	Ramping Rate
Spring Baseflow	April 1 – April 30	40 cfs	Maximum: 10% or 10 cfs*
Spring Ascension	May 1 – May 15	40 cfs ascending to 80 cfs	Target: 5% Maximum: 25%
Spring Bench	May 16 – June 11	80 cfs	Maximum: 20%
Snowmelt Ascension	June 12 – June 19	80 cfs ascending to 170 cfs	Target: 10% Maximum: 20%
Snowmelt Bench	June 20 – August 1	170 cfs	Maximum Ascending: 20% Maximum Descending: 10% or 10 cfs*
Snowmelt Flood and Snowmelt Peak	Starting between June 20 and July 8 with the 5-day peak between June 27 and July 19	170 cfs ascending to 650 cfs, 650 cfs for 5 days, 650 cfs descending to 170 cfs	Target Ascending: 20% Maximum Ascending: 40% Maximum Descending: 10% or 10 cfs*
Medium Recession (Node)	August 2 – August 15	170 cfs descending to 71 cfs	Target: 6% Maximum: 10% or 10 cfs*
Slow Recession	August 16 – September 13	71 cfs descending to 30 cfs	Target: 3% Maximum: 10% or 10 cfs*
Summer Baseflow	September 14 – 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 1C: RUSH CREEK STREAM ECOSYSTEM FLOWS FOR WET/NORMAL YEARS

Hydrograph Component	Timing	Flow Requirement	Ramping Rate
Spring Baseflow	April 1 – April 30	40 cfs	Maximum: 10% or 10 cfs*
Spring Ascension	May 1 – May 15	40 cfs ascending to 80 cfs	Target: 5% Maximum: 25%
Spring Bench	May 16 – June 11	80 cfs	Maximum: 20%
Snowmelt Ascension	June 12 – June 18	80 cfs ascending to 145 cfs	Target: 10% Maximum: 20%
Snowmelt Bench	June 19 – July 23	145 cfs	Maximum Ascending: 20% Maximum Descending: 10% or 10 cfs*
Snowmelt Flood and Snowmelt Peak	Starting between June 19 and July 1 with the 3-day peak between June 26 and July 10	145 cfs ascending to 550 cfs, 550 cfs for 3 days, 550 cfs descending to 145 cfs	Target Ascending: 20% Maximum Ascending: 40% Maximum Descending: 10% or 10 cfs*
Medium Recession (Node)	July 24 – August 4	145 cfs descending to 69 cfs	Target: 6% Maximum: 10% or 10 cfs*
Slow Recession	August 5 – September 1	69 cfs descending to 30 cfs	Target: 3% Maximum: 10% or 10 cfs*
Summer Baseflow	September 2 – 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 2A LEE VINING CREEK STREAM ECOSYSTEM FLOWS

Timing: April 1 – September 30						Year-type: Extreme/Wet, Wet, Wet/Normal, Normal, Dry/Normal II				
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	31	32	33	34
40	30	31	32	33	34	35	36	37	38	39
50	35	36	37	38	39	40	41	42	43	44
60	45	46	47	48	49	50	51	52	53	54
70	55	56	57	58	59	60	61	62	63	64
80	60	61	62	63	64	65	66	67	68	69
90	70	71	72	73	74	75	76	77	78	79
100	75	76	77	78	79	80	81	82	83	84
110	85	86	87	88	89	90	91	92	93	94
120	95	96	97	98	99	100	101	102	103	104
130	100	101	102	103	104	105	106	107	108	109
140	110	111	112	113	114	115	116	117	118	119
150	120	121	122	123	124	125	126	127	128	129
160	130	131	132	133	134	135	136	137	138	139
170	135	136	137	138	139	140	141	142	143	144
180	145	146	147	148	149	150	151	152	153	154
190	155	156	157	158	159	160	161	162	163	164
200	160	161	162	163	164	165	166	167	168	169
210	170	171	172	173	174	175	176	177	178	179
220	180	181	182	183	184	185	186	187	188	189
230	190	191	192	193	194	195	196	197	198	199
240	195	196	197	198	199	200	201	202	203	204
250	200									
251 cfs and greater	Licensee shall bypass inflow.									

Attachment 2

2019 EASTERN SIERRA
RUNOFF FORECAST
April 1, 2019

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 (1966 - 2015) (Acre-feet)
MONO BASIN:	**148,700**	**148%**	160%	135%	100,782
OWENS RIVER BASIN:	**432,000**	**145%**	158%	132%	298,151

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 (1966 - 2015) (Acre-feet)
MONO BASIN:	**171,900**	**144%**	158%	131%	119,103
OWENS RIVER BASIN:	**554,000**	**137%**	149%	124%	405,696

NOTE - Owens River Basin includes Long, Round and Owens Valleys (not incl Laws Area)

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

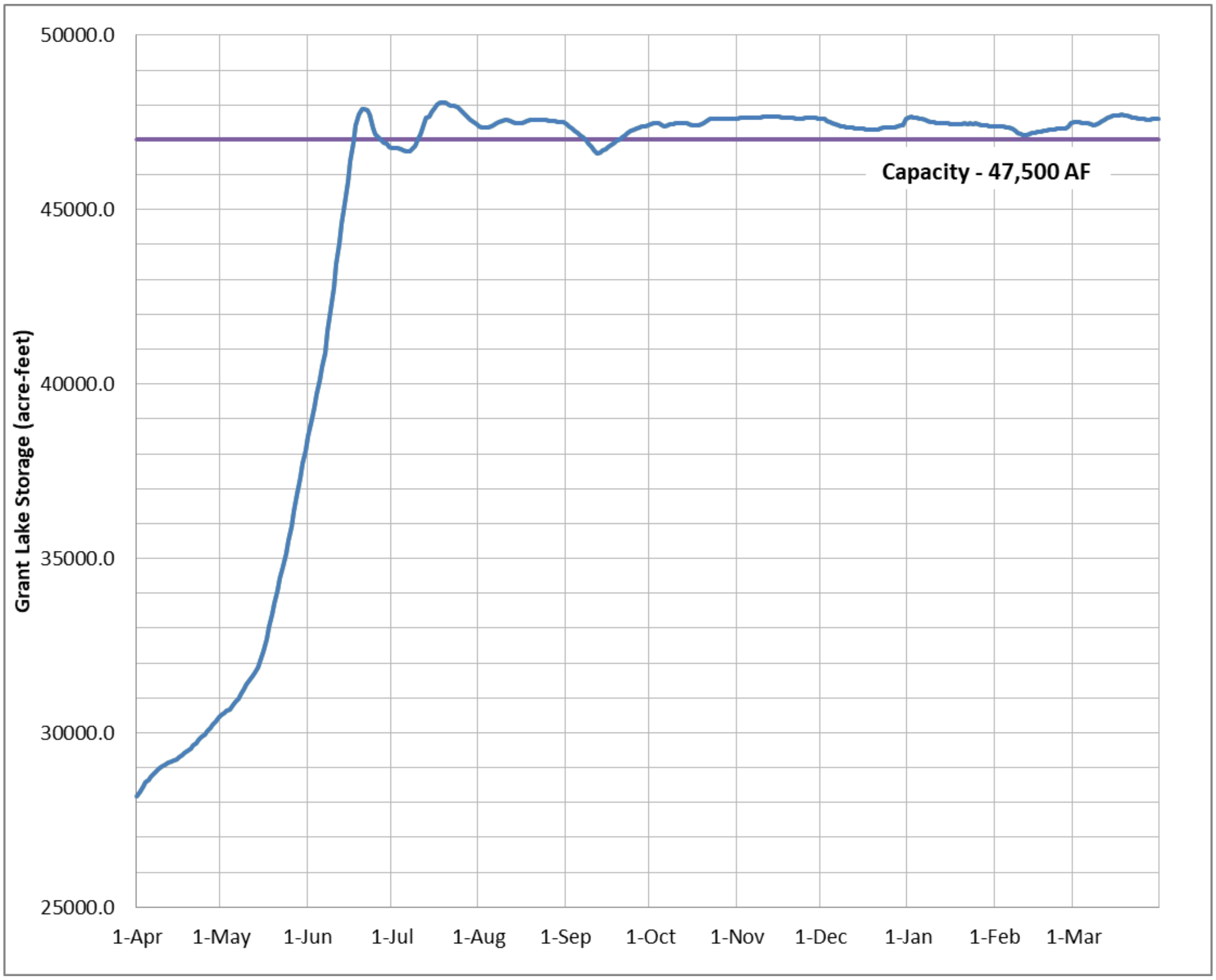
REASONABLE MAXIMUM - That runoff which is expected to occur if precipitation subsequent to the forecast is equal to the amount which is exceeded on the average once in 10 years.

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.

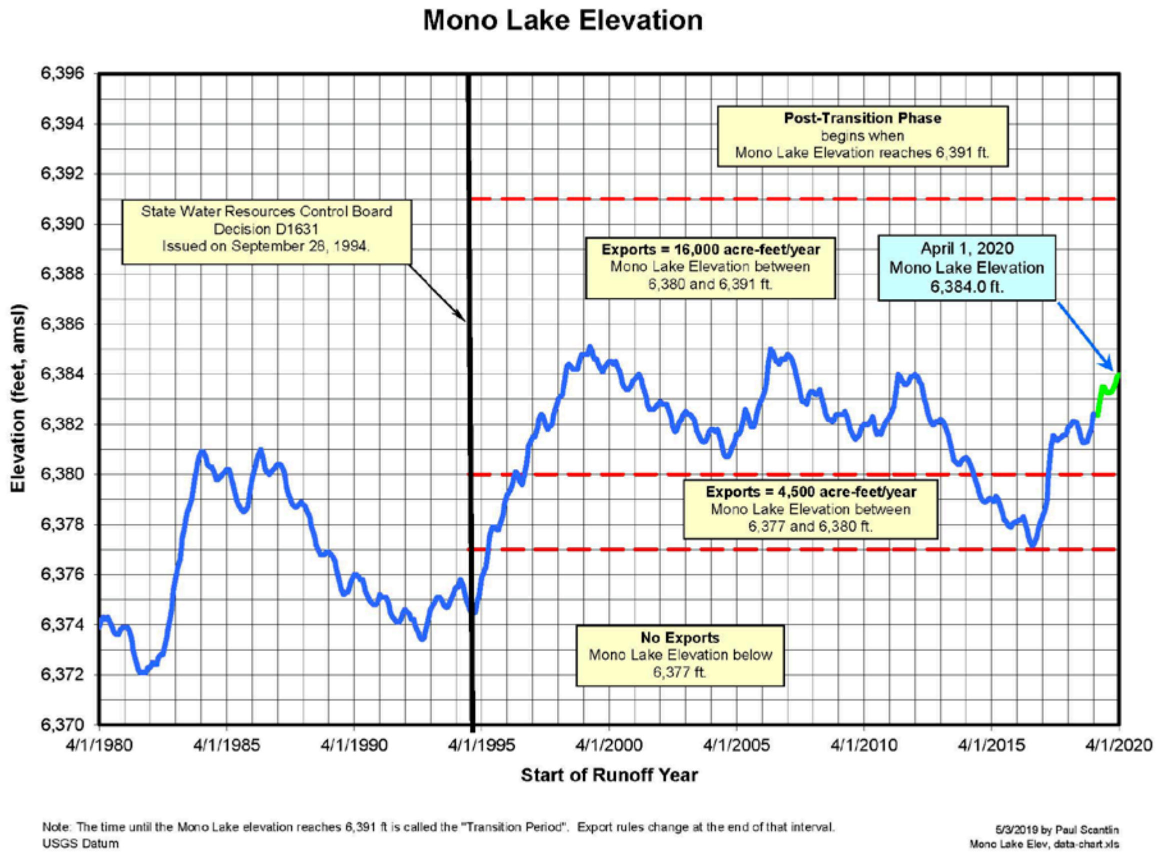
2019 Forecast.xls forecast 4/5/2019 12:44 PM

Attachment 3

RY 2019/20 Grant Lake Reservoir Storage Projection
Using 2005 (147% Year) Inflow (eSTREAM Release v3.2)



Attachment 4



Section 3

Mono Basin Fisheries Monitoring Report: Rush, Lee Vining, Parker, and Walker Creeks 2018

Mono Basin Fisheries Monitoring Report
Rush, Lee Vining, and Walker Creeks
2018

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

Date: April 2, 2019

Table of Contents

Executive Summary

This report presents results of the 22nd year of trout population monitoring for Rush, Lee Vining, and Walker Creeks pursuant to SWRCB's Water Right Decision 1631 (D1631) and the 20th 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 data collected in 2018 as mandated by the Orders and the Settlement Agreement.

The 2018 runoff year (RY) was 85% of normal and classified a "Normal" runoff year (RY) type, as measured on April 1st. The range of runoff that defines a Normal year is 82.5% - 107% (40-60% exceedence), thus RY 2018 was at the low end of the Normal range. The preceding six years included a record runoff of 206% in RY 2017 and five consecutive below "Normal" runoff 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). Annual electrofishing mark-recapture monitoring was conducted in the Lee Vining Creek main channel section and in three sections of Rush Creek – Upper Rush, Bottomlands and the MGORD. Multiple-pass depletion electrofishing was conducted in the Lee Vining Creek side channel and in Walker Creek. These data were used to generate population estimates, density estimates, standing crop estimates, condition factors, relative stock densities, and growth rates and apparent survival rates from PIT tag recaptures.

Population Estimates

The Upper Rush section supported an estimated 1,572 age-0 Brown Trout in 2018 compared to 612 age-0 fish in 2017 and 146 age-0 fish in 2016. This section supported an estimated 196 Brown Trout 125-199 mm in length in 2018 compared to a total catch of 31 fish in 2017 (insufficient numbers of recaptures prevented making a valid estimate in 2017). In 2018, Upper Rush supported an estimated 195 Brown Trout ≥200 mm in length compared to an estimate of 158 fish in 2017. In 2018, sufficient numbers of naturally-produced Rainbow Trout were sampled in the Upper Rush section to generate unbiased estimates for two of the three size classes. This section supported an estimated 319 Rainbow Trout <125 mm in length and an estimated 27 Rainbow Trout ≥200 mm in length.

The Bottomlands section supported an estimated 1,808 age-0 Brown Trout in 2018 versus 149 age-0 fish in 2017. This section supported an estimated 100 Brown Trout 125-199 mm in length in 2018 compared to 59 fish in 2017. The Bottomlands section supported an estimated 106 Brown Trout ≥200 mm in 2018 compared to 80 trout in 2017.

In 2018, insufficient numbers of age-0 Brown Trout were captured in the MGORD section of Rush Creek to generate a valid estimate; the total catch of trout <125 mm was 24 fish. Also in 2018, insufficient numbers of Brown Trout in the 125-199 mm size class were captured in the

MGORD to generate a valid estimate; the total catch equaled 34 fish. Brown Trout ≥200 mm in length accounted for 81% of the total catch in 2018 and the population estimate for this size class was 771 Brown Trout. The largest Brown Trout captured in the MGORD in 2018 was 550 mm in length; one of six Brown Trout caught that were >500 mm in length.

Lee Vining Creek's main channel section supported an estimated 192 age-0 Brown Trout in 2018, compared to an estimated 32 age-0 fish in 2017 and 118 age-0 fish in 2016. This section supported an estimated 71 Brown Trout 125-199 mm in length in 2018 compared to 13 fish in 2017 and 150 fish in 2016. Lee Vining Creek's main channel supported an estimated 14 Brown Trout ≥200 mm in 2018 versus 10 fish in 2017 and 50 fish in 2016.

Five Rainbow Trout (<125 mm) were captured in Lee Vining Creek's main channel in 2018. No Rainbow Trout in the 125-199 mm size class (probable age-1 fish) were captured in Lee Vining Creek's main channel during the past three sampling years.

The 2018 age-0 Brown Trout estimate for Walker Creek was 44 fish, compared to 66 fish in 2017. The 2018 population estimate for Brown Trout in the 125-199 mm size class equaled 86 fish (47 trout in 2017). Brown Trout ≥200 mm in length accounted for 26% of the total catch in 2018 and the population estimate for this size class was 45 Brown Trout. The largest Brown Trout captured in Walker Creek in 2018 was 274 mm in length.

In the Lee Vining Creek side channel, 10 Brown Trout were captured in two electrofishing passes during the 2018 sampling (23 fish in two passes during the 2017 sampling). The estimates for each size class were: <125 mm = three fish; 125-199 mm = seven fish; and ≥200 mm= no fish. No Rainbow Trout were captured in the side channel in 2018. This was the tenth consecutive year that no age-0 Rainbow Trout were captured in the Lee Vining Creek side channel and the eighth consecutive year the no age-1 and older Rainbow Trout were captured.

Densities of Age-0 Trout

In 2018, the Upper Rush section's estimated density of age-0 Brown Trout was 4,502 fish/ha and the Bottomlands section's estimated density of age-0 Brown Trout equaled 5,444 fish/ha. In Walker Creek, the 2018 density estimate of age-0 Brown Trout was 1,086 fish/ha.

The 2018 age-0 Brown Trout density estimate in the main channel of Lee Vining Creek was 1,394 fish/ha (a 500% increase from 2017's estimate of 232 fish/ha). In 2018, the age-0 Brown Trout density estimate in the Lee Vining Creek side channel equaled 59 fish/ha.

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

In 2018, the Upper Rush section's estimated density of age-1+ Brown Trout was 1,120 fish/ha and the Bottomlands section's estimated density of age-1+ Brown Trout equaled 620 fish/ha.

In the MGORD section of Rush Creek, the 2018 density estimate of age-1+ Brown Trout equaled 430 fish/ha. In Walker Creek, the 2018 density estimate of age-1+ Brown Trout was 3,235 fish/ha.

The 2018 age-1+ Brown Trout density estimate in the main channel of Lee Vining Creek was 617 fish/ha (an increase of 270% from the 2017 estimate of 167 fish/ha). In 2018, the Lee Vining Creek side channel's density estimate of age-1 and older Brown Trout was 138 fish/ha.

Standing Crop Estimates

In 2018, the estimated standing crop for Brown Trout in the Upper Rush section was 170 kg/ha and the estimated standing crop for Rainbow Trout was 19 kg/ha. The estimated standing crop for Brown Trout in the Bottomlands section of Rush Creek was 103 kg/ha in 2018. The estimated standing crop for Brown Trout in the MGORD section of Rush Creek was 95 kg/ha in 2018. The estimated standing crop for Brown Trout in Walker Creek was 245 kg/ha in 2018.

The Lee Vining Creek main channel in 2018 produced a total estimated standing crop of 70 kg/ha for Brown Trout. The Lee Vining Creek side channel produced a total Brown Trout standing crop estimate of 7 kg/ha in 2018.

Condition Factors

Condition factors of Brown Trout 150 to 250 mm in length in 2018 increased in the MGORD section of Rush Creek and in Walker Creek from their 2017 values and decreased in the four other sections from 2017 values (Upper Rush, Bottomlands, Lee Vining side channel, and Lee Vining main channel). In 2018, four sections (MGORD, Walker Creek, Lee Vining main channel and Lee Vining side channel) had Brown Trout condition factors ≥1.00.

Relative Stock Densities (RSD)

In the Upper Rush section, the RSD-225 equaled 39 for 2018, a large drop from the record RSD-255 value of 78 for 2017. This decrease was most likely influenced by greater numbers of fish, especially the numbers of fish smaller than 225 mm which comprised 61% of the trout ≥150 mm. The RSD-300 value was 9 in 2018, compared to 15 in 2017. This decrease was influenced by the higher numbers of fish ≤225 mm caught in 2018.

In the Bottomlands section of Rush Creek, the RSD-225 for 2018 equaled 36, a large drop from the record value of 65 for 2017. As in the Upper Rush section, the Bottomlands 2018 RSD-225 value was most likely influenced by greater numbers of fish, especially the numbers of fish smaller than 225 mm which comprised 64% of the trout ≥150 mm. The RSD-300 value was 6 in 2018. In 2018, nine Brown Trout ≥300 mm were captured in the Bottomlands section, most likely the result of a second year of good growth rates and higher survival rate.

In the MGORD, the RSD-225 value increased from 72 in 2015 to 74 in 2016 to 88 in 2017; this was the fourth consecutive increase since the low value of 42 in 2013. In 2017, the RSD-300 value was 27, an increase from a value of 21 in 2016. The RSD-375 value in 2017 was 11, the second consecutive season with a value of 11. In 2017, a total of 28 Brown Trout ≥300 mm in length were caught, including 11 fish ≥375 mm in length.

RSD values in Lee Vining Creek were generated for the main channel combined with the side channel and for the main channel only. The RSD-225 values for the main/side combined equaled 23 and main-only equaled 26 for 2017, both increases compared to the 2016 values. In 2017, one Brown Trout greater than 300 mm in length was captured in the Lee Vining Creek main channel, which resulted in a RSD-300 of 4 for the main channel and a RSD-300 of 3 for the main/side channels combined.

Introduction

Study Area

Between September 17th and 27th 2018, 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 which have been sampled between 2009 and 2017 (Figure 1). Aerial photographs of the sampling reaches are provided in Appendix A.

Hydrology

The 2018 runoff year (RY) was 85% of normal and classified a "Normal" runoff year (RY) type, as measured on April 1st. The range of runoff that defines a Normal year is 82.5% - 107% (40-60% exceedence), thus RY 2018 was at the low end of the Normal range. The preceding six years included a record runoff of 206% in RY 2017 and five consecutive below "Normal" runoff 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). Under the existing SWRCB orders and the Stream Restoration Flows (SRF), a Normal RY prescribes a Rush Creek summer baseflow of 47 cfs from April 1st to September 30th, a two-stage peak release of 380 cfs for five days, followed by 300 cfs for eight days, followed by baseflows of 30 to 44 cfs from October 1 through March 31. Management of Rush Creek flows and storage levels in Grant Lake Reservoir (GLR) were confounded by late-spring and summer rainfall and upstream reservoir management and maintenance by Southern Cal Edison; which resulted in additional inflow to GLR. After releasing the prescribed Rush Creek baseflow during most of July, LADWP increased flows throughout August to mid-September to lower GLR so that Rush Creek flows could be lowered to accommodate the fisheries sampling in September without a spill occurring (Figure 2). In Lee Vining Creek, the existing SWRCB orders require that the primary peak flow is passed downstream. The SRF summer baseflow in Lee Vining Creek below LADWP's point of diversion was 54 cfs or to pass all the flow if less than 54 cfs.

The peak discharges in Rush Creek at MGORD equaled 380 cfs for four days on June 6th-9th (red line on Figure 2). As previously, described, the summer baseflows of around 45 cfs were increased for 47 days (7/28/18 to 9/12/18) to an average of 78 cfs for the purpose of lowering GLR storage to accommodate flows in late September for the annual fisheries monitoring (Figure 2). Accretions from Parker and Walker creeks resulted in flow fluctuations through the spring and summer, and contributed to the peak of 442 cfs in Rush Creek below the Narrows on June 6th and a total of 13 days where flows exceeded 300 cfs (green line on Figure 2).

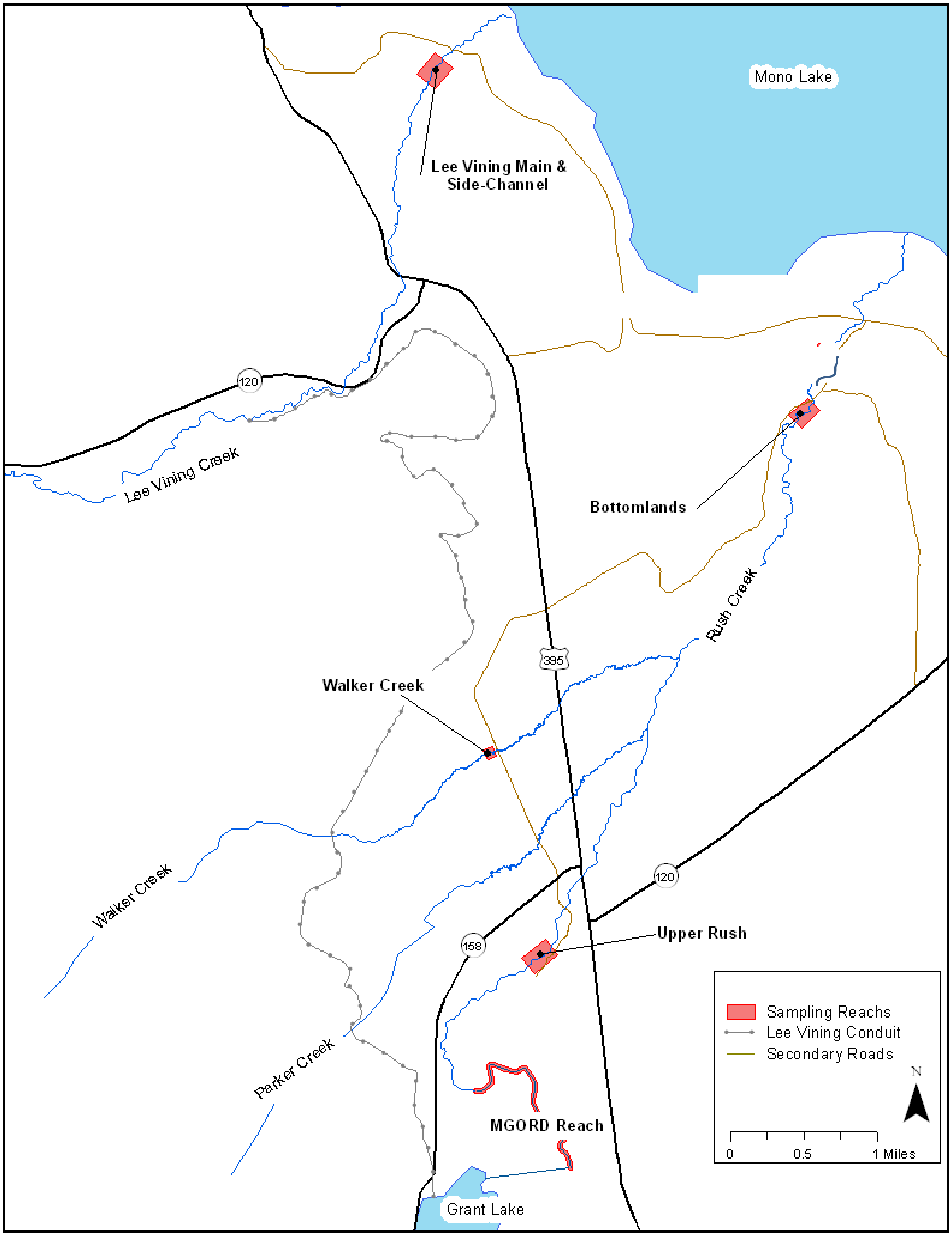


Figure 1. Annual fisheries sampling sites within Mono Basin study area, September 2018.

In 2018, multiple peaks occurred in Lee Vining Creek, with a peak of approximately 250 cfs passed downstream on April 8th (Figure 3). After diverting flow to GLR through mid-May to late-July, flow-through conditions were resumed to assist in lowering GLR for upcoming fisheries sampling. However, flows in Lee Vining Creek were approximately 45 cfs during the mark-run on 9/19/18 and wading conditions were unsafe; flows were subsequently lowered to 30 cfs for the recap-run conducted on 9/26/18 (Figure 3). In this year's report's Methods Evaluation section, we clarified the preferred flow for safely conducting the annual fisheries sampling in Lee Vining Creek as no more than 30 cfs.

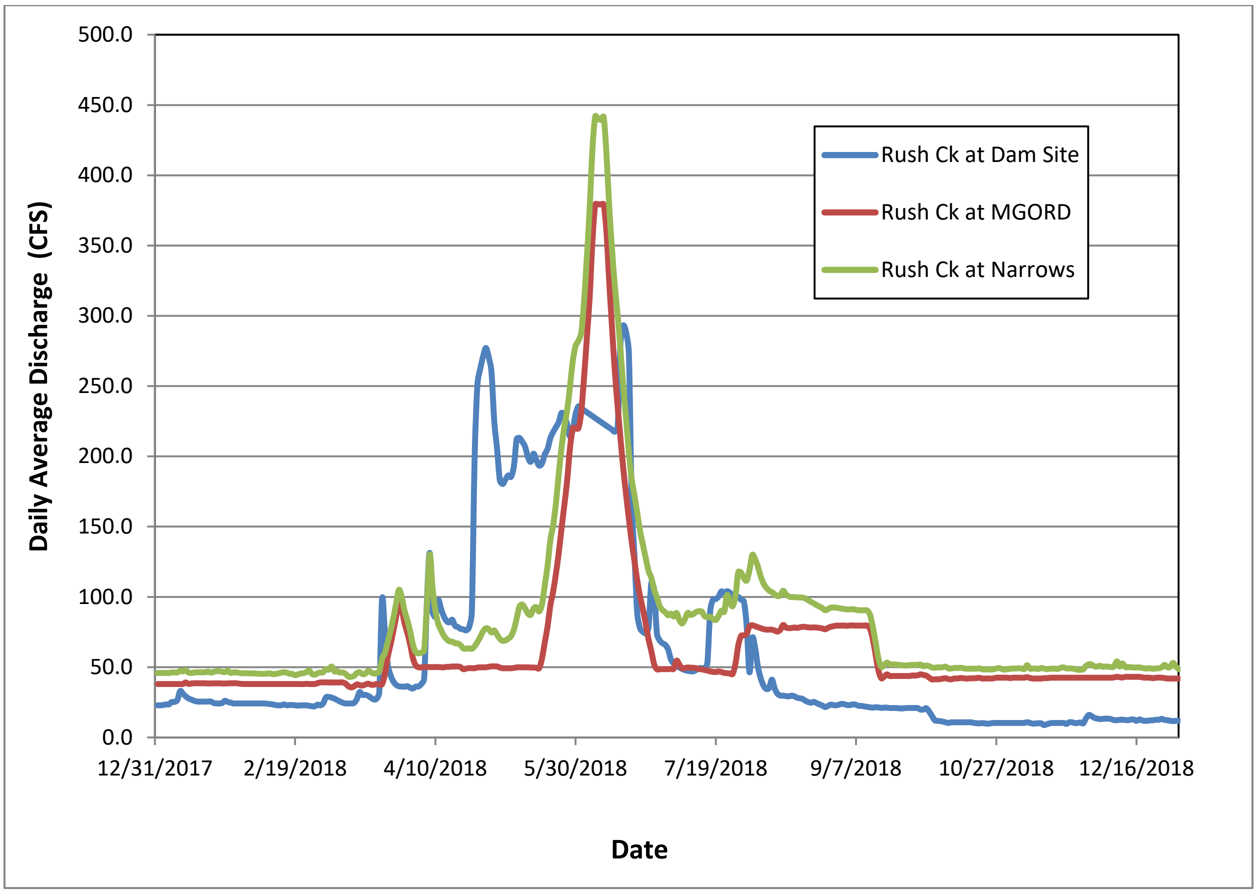


Figure 2. Rush Creek hydrographs between January 1st and December 31 of 2018.

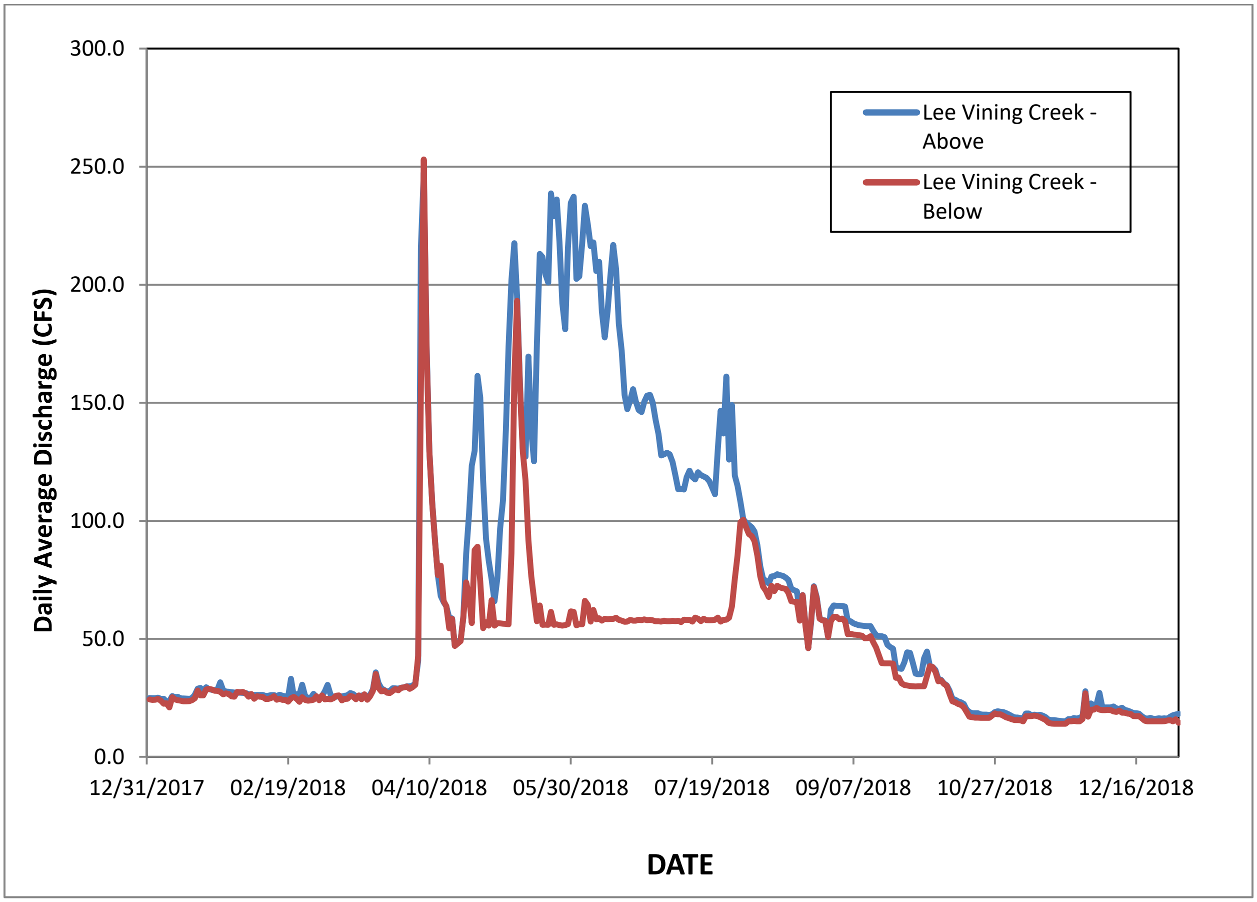


Figure 3. Lee Vining Creek hydrographs between January 1st and December 31st of 2018.

Grant Lake Reservoir

In 2018, storage elevation levels in GLR fluctuated from a low of 7,099.5 ft to a high of 7,128.5.0 ft (Figure 4). In 2018, GLR continued to fill throughout June and July due to frequent, and sometimes heavy, rainfall and reached its peak storage level on July 31, 2018. Also, Southern Cal Edison (SCE) was performing maintenance work on their dams and reservoirs upstream of GLR, so more runoff was allowed to flow-through instead of being stored by SCE.

Because of the previously mentioned conditions during RY2018, GLR's elevation was well 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). The 2018 summer water temperature monitoring documented cool water temperatures, suitable for fair to good growth of Brown Trout, at all Rush Creek locations downstream of GLR.

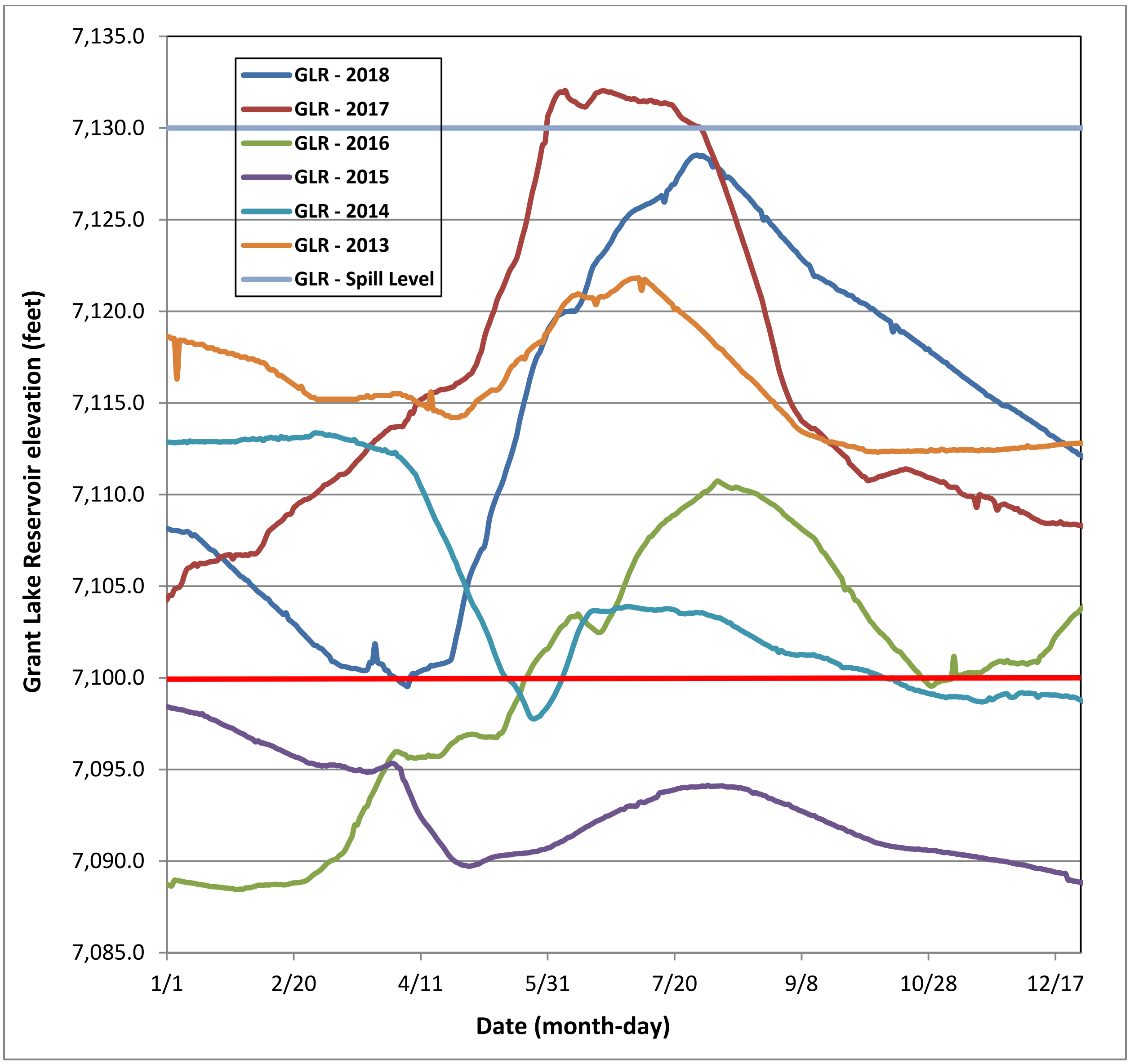


Figure 4. Grant Lake Reservoir's elevation between January 1st and December 31st 2013 - 2018.

Methods

The annual fisheries monitoring was conducted between September 17th and 27th of 2018. 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-recapture 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 2018, the field crew consisted of a barge operator, two anode operators, and four netters, two for each anode. A safety officer was also used during the 2018 sampling and this person walked the streambank and observed the in-stream operations. 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 LR-24 backpack electrofishers, four netters, and one bucket carrier who transported the captured trout. Again, a safety officer walked the streambank and observed the in-stream operations.

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

For the Walker Creek and Lee Vining Creek side channel (B-1 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. Again, a safety officer walked the streambank and observed the in-stream operations. 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 Bottom-lands and MGORD sections received anal fin clips and trout captured in the Upper Rush section received lower caudal fin clips. Before placing trout into the aerated recovery bucket, each fish was examined for a missing adipose fin. Trout missing their adipose fin were then scanned for their Passive Integrated Transponder (PIT) tag number. Any trout missing their adipose fin that failed to produce a tag number when scanned were recorded as having “shed” the PIT tag; in most instances these fish were retagged. Partially regenerated adipose fins of fish with PIT tags were reclipped for ease of future identification. Once recovered, fish were then moved from the recovery bucket to a live car to be held until the day's sampling effort was completed; this was done to prevent captured fish from potentially moving downstream into the actively sampled section. At the end of the electrofishing effort, fish were released from the live cars back into the sub-sections they had been captured in. Fish were then provided a seven-day period to remix back into the section's population prior to conducting the recapture-run.

Processing trout during the recapture-run was similar to the mark-run. Trout were transferred in small batches to a five gallon bucket. They were then anesthetized, identified, and examined for the "mark" fin clip. Trout that were fin clipped were only measured to the nearest millimeter and placed in the recovery bucket. Trout that were not clipped during the "mark" run (i.e. new fish) were measured to the nearest millimeter "total length," weighed to the nearest gram, and examined for missing adipose fins. New trout missing adipose fins were then scanned for their PIT tag number then placed into recovery. Again, trout that failed to produce a tag number were recorded as having "shed" the PIT tag, and were usually re-tagged.

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 - 2018 field seasons.

All data collected in the field, were written on data sheets and entered into Excel spreadsheets using a field laptop computer. 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 and density.

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

or 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. 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.

Length-Weight Relationships

Length-weight regressions (Cone 1989 as cited in Taylor and Knudson. 2011) 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). 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

Termination Criteria Calculations and Analyses

Information regarding the proposed termination criteria, calculations, and analyses were conducted as described in past Annual Fisheries Reports (Taylor and Knudson 2012).

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 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 2018:

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

For the fisheries monitoring program, the year-long data sets were edited to focus on the 2018 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 temperature.
7. Number of bad thermal days based on daily average temperature.
8. Maximum diurnal fluctuations.
9. Average maximum diurnal fluctuation for 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 2017 are provided for comparisons (Table 1). In 2018, the Upper Rush and the Bottomlands sample sections were shortened so the block fences could be set at favorable locations to deal with changes in channel depths and velocities (Table 1). The Lee Vining Creek side channel carried more water in 2018, thus its length increased (Table 1). Between 2017 and 2018, several abandoned meanders were reconnected in Walker Creek, resulting in a longer channel length in 2018, yet average wetted width was narrower in 2018, resulting in a smaller wetted area (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 10-20, 2018. Values from 2017 are provided for comparisons.

Sample Section	Length (m) 2017	Width (m) 2017	Area (m^2) 2017	Length (m) 2018	Width (m) 2018	Area (m^2) 2018	Area (ha) 2018
Rush – Upper	430	7.4	3,182.0	406	8.6	3,491.6	0.3492
Rush - Bottomlands	452	7.1	3,209.2	437	7.6	3,321.1	0.3321
Rush – MGORD	2,230	7.6	16,948.0	2,230	8.4	18,732.0	1.8732
Lee Vining – Main	255	5.4	1,377.0	255	5.4	1,377.0	0.1377
Lee Vining - Side	177	2.2	389.4	195	2.6	507.0	0.0507
Walker Creek	169	2.6	439.4	193	2.1	405.3	0.0405

Trout Population Abundance

In 2018, a total of 776 Brown Trout ranging in size from 64 mm to 362 mm were captured in the Upper Rush section (Figure 5). For comparison, in 2017 a total of 373 Brown Trout were captured and in 2016 a total of 182 Brown Trout were captured in this section. In 2018, age-0 Brown Trout comprised 67% of the total catch (compared to 58% in 2018 and 41% in 2016). The Upper Rush section supported an estimated 1,572 age-0 Brown Trout in 2018 (including morts) compared to 612 age-0 Brown Trout in 2017 (a 157% increase). The estimated standard error of the population estimate for age-0 Brown Trout in 2018 was 11% (Table 2).

In 2018, the 112 Brown Trout captured in the 125-199 mm size class comprised 14% of the total catch in the Upper Rush section (compared to 8% in 2017 and 19% in 2016). Upper Rush section supported an estimated 196 Brown Trout in the 125-199 mm size class in 2018 (including morts). The estimated standard error of the population estimate for 125-199 mm Brown Trout in 2018 was 14% (Table 2).

Brown Trout ≥200 mm in length comprised 18% of the Upper Rush total catch in 2018 (compared to 34% in 2017 and 40% in 2016). In 2018, Upper Rush supported an estimated 195 Brown Trout ≥200 mm in length compared to an estimate of 158 fish in 2017 (a 23% increase). Standard error of the estimate for this size class was 8% in 2018 versus 6% in 2017. In 2018, 24 Brown Trout ≥300 mm in length were captured in the Upper Rush section and these fish comprised 3% of the total catch (Figure 5).

A total of 168 Rainbow Trout were captured in the Upper Rush section comprising 17.8% of the section's total catch in 2018 (a total of 944 trout were caught). The 168 Rainbow Trout ranged in length from 55 mm to 323 mm and 137 of these were age-0 fish (Figure 6). Most of the Rainbow Trout appeared to be of naturally produced origin and sufficient numbers in two size classes (<125 mm and 125-199 mm) were marked and recaptured to produce unbiased estimates. In 2018, the Upper Rush section supported an estimated 319 Rainbow Trout <125 mm in length and an estimated 27 Rainbow Trout ≥200 mm in length (Table 2). The total catch of Rainbow Trout in the 125-199 mm size class was nine fish (Table 2).

Within the Bottomlands section of Rush Creek, a total of 699 Brown Trout were captured in 2018 (Table 2), which ranged in size from 54 mm to 360 mm (Figure 7). For comparison, 164 Brown Trout were captured in 2017 and 148 Brown Trout were captured in 2016. Age-0 Brown Trout comprised 80% of the total catch in 2018 versus 35% of the total catch in 2017. The Bottomlands section supported an estimated 1,808 age-0 Brown Trout in 2018 versus 149 age-0 fish in 2017 (a 12-fold increase). Estimated standard error for the 2018 population estimate of age-0 Brown Trout was 12% (16% in 2017) (Table 2).

Brown Trout 125-199 mm in length comprised 10% of the total catch in the Bottomlands section in 2018 versus 19% of the total catch in 2017. This section supported an estimated 100 Brown Trout 125-199 mm in length in 2018 compared to 59 fish in 2017 (a 69% increase). Estimated standard error for the population estimate of this size class was 16% in 2018 versus 7% in 2017 (Table 2).

Brown Trout ≥200 mm in length comprised of 10% of the total catch in 2018 (46% in 2017) with the largest trout 360 mm in length. The Bottomlands section supported an estimated 106 Brown Trout ≥200 mm in 2018 compared to 80 trout in 2017 (a 33% increase). Standard error for the estimate of this size class was 7% in 2018 versus 4% in 2017 (Table 2). In 2018, nine Brown Trout ≥300 mm were captured in the Bottomlands section; these fish were 300, 301, 302, 307 310, 317, 318, 319 and 360 mm in length (Figure 7).

Table 2. Rush Creek mark-recapture estimates for 2018 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							
9/17/2018 & 9/24/2018							
	0 - 124 mm	249	319	50	4	**1,572**	178
	125 - 199 mm	53	79	21	1	**196**	14
	≥200 mm	85	101	44	1	**195**	10
Upper Rush - **RBT**							
9/17/2018 & 9/24/2018							
	0 - 124 mm	68	87	18	0	**319**	54
	125 - 199 mm	4	7	2	0	**NP**	---
	≥200 mm	14	16	8	0	**27**	4
Bottomlands - BNT							
9/18/2018 & 9/25/2018							
	0 - 124 mm	266	337	49	4	**1,808**	42
	125 - 199 mm	45	45	20	0	**100**	16
	≥200 mm	54	34	17	0	**106**	7
MGORD - BNT							
9/20/2018 & 9/27/2018							
	0 - 124 mm	17	7	0	0	**NP**	42
	125 - 199 mm	20	16	2	0	**NP**	16
	≥200 mm	179	153	35	2	**771**	7
Lee Vining Creek							
Main Channel - BNT							
9/19/2018 & 9/26/2018							
	0 - 124 mm	37	60	11	0	**192**	42
	125 - 199 mm	24	22	7	0	**71**	16
	≥200 mm	9	11	7	0	**14**	7

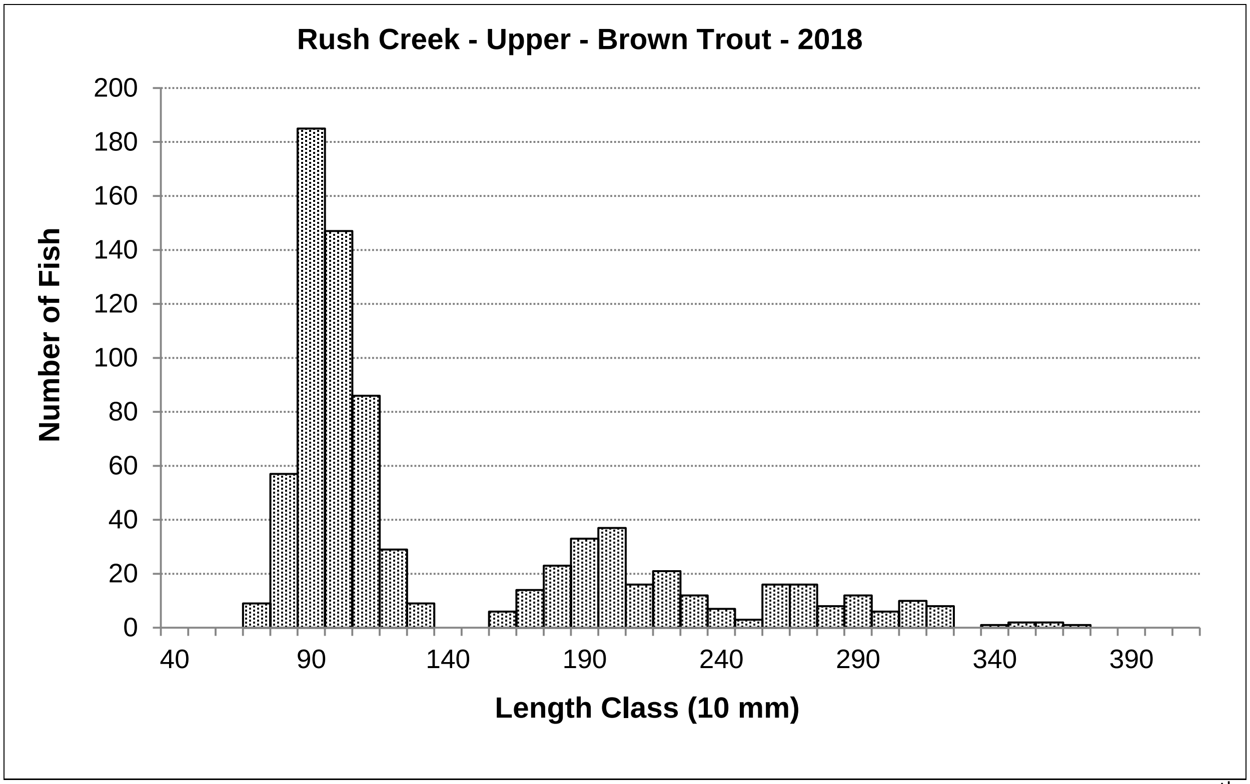


Figure 5. Length-frequency histogram of Brown Trout captured in Upper Rush, September 17th and 24th, 2018.

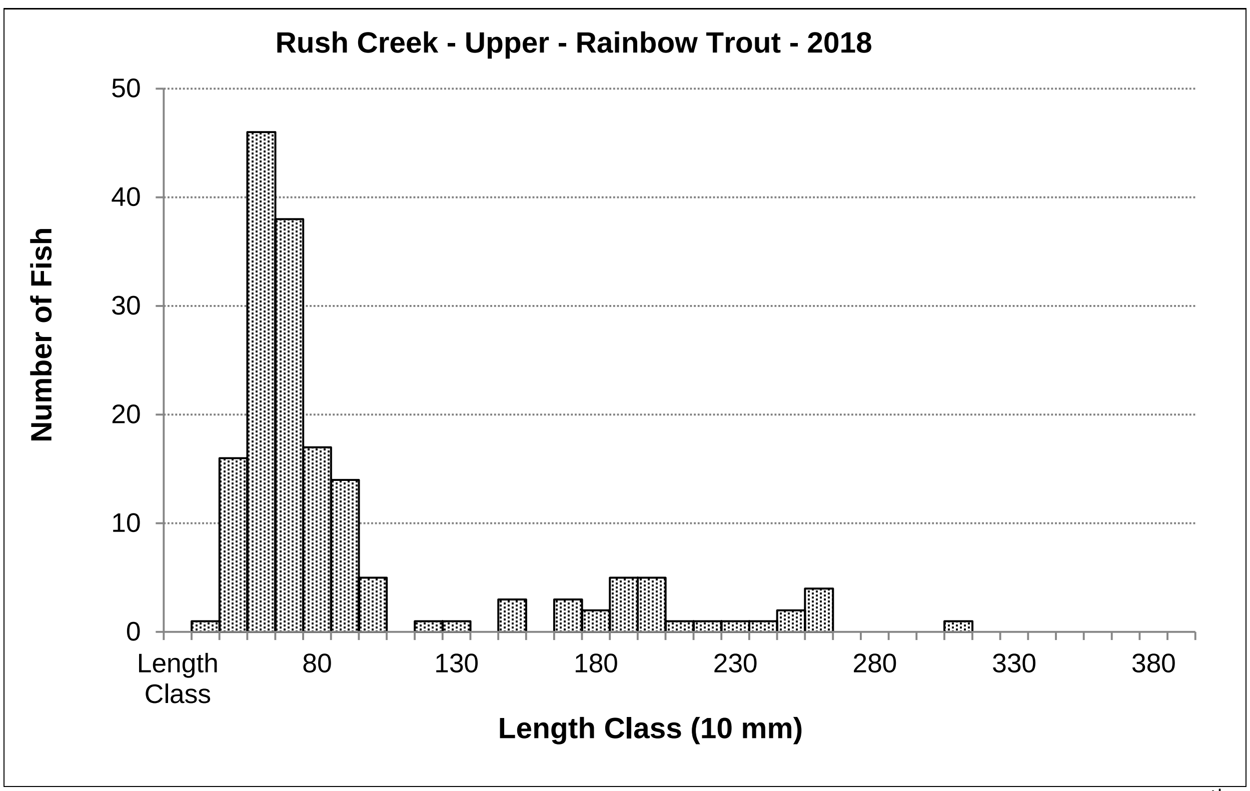


Figure 6. Length-frequency histogram of Rainbow Trout captured in Upper Rush, October 17th and 24th, 2018.

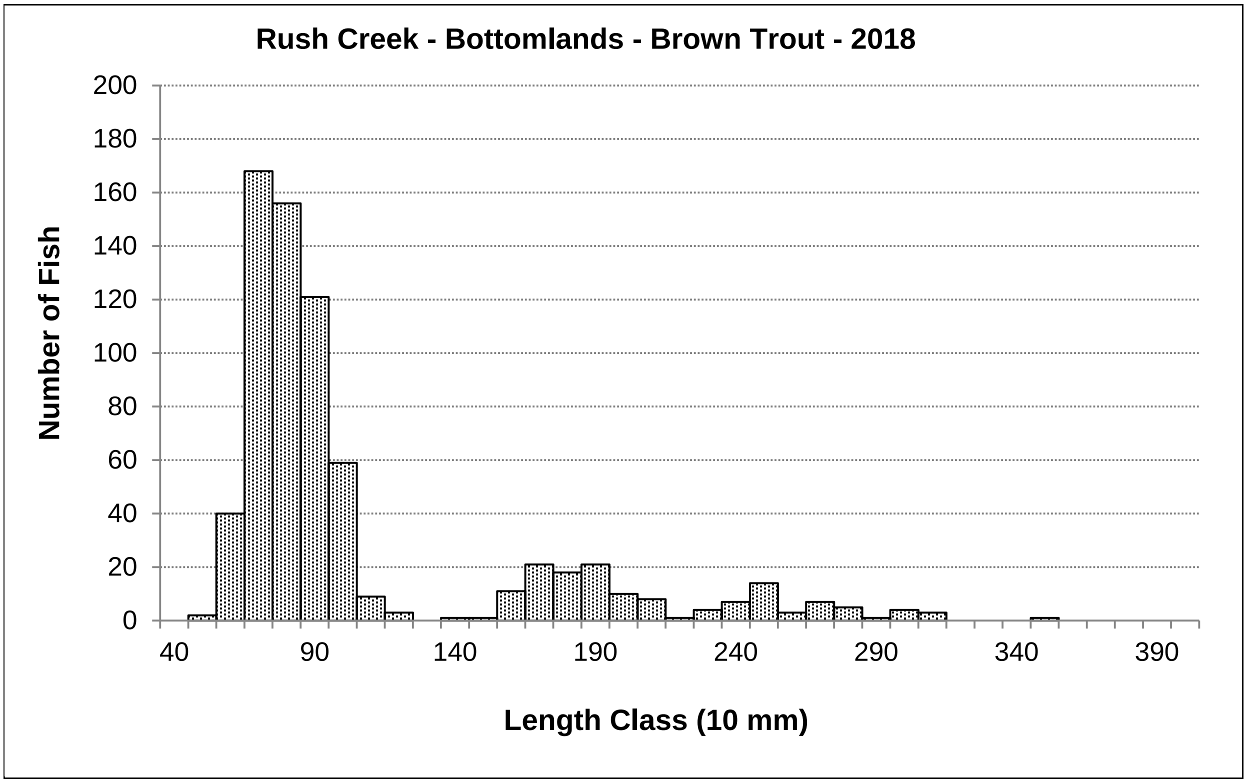


Figure 7. Length-frequency histogram of Brown Trout captured in the Bottomlands section of Rush Creek, September 18th and 25th, 2018.

Within the MGORD section of Rush Creek a total of 357 Brown Trout were captured during the mark and recapture electrofishing passes made in 2018. These Brown Trout ranged in size from 96 mm to 550 mm (Figure 8). Twenty-four Brown Trout <125 mm in length were captured in 2018 which comprised 9% of the total catch, compared to 22% of the total catch in 2017 and 2% of the total catch in 2016. No estimate was possible for this size class because we did not recapture any marked fish during the recapture electrofishing pass.

The 34 Brown Trout in the 125-199 mm size class comprised 10% of the total catch in the MGORD section in 2018 versus 17% of the total catch in 2017. Seven Brown Trout in this size class were 126 to 137 mm in length and were probably age-0 fish, due to the good growth typically exhibited by young fish in the MGORD. No population estimate was made for this size class due to the low number of clipped recaptures (2 fish).

Brown Trout ≥200 mm in length comprised of 81% of the total catch in the MGORD section during 2018 (61% in 2017 and 93% in 2016). In 2018, 66 Brown Trout ≥300 mm were captured in the MGORD (28 fish ≥300 mm in 2017 and 38 fish ≥300 mm in 2016). Fifteen Brown Trout ≥375 mm in length were captured in 2018 (11 fish in 2017 and 20 fish in 2016) and six of these fish were >500 mm in length (Figure 8). In 2018, the MGORD supported an estimated 771 Brown Trout ≥200 mm in length. Standard error for the estimate of this size class was 12%.

In 2018, 26 Rainbow Trout were captured in the MGORD section (Figure 9). In the previous five years, 39 Rainbow Trout were captured in 2017, eight in 2016, two in 2015, none in 2014, and nine in 2013. The Rainbow Trout captured in 2018 appeared to be a mix of hatchery origin and naturally-produced fish based on the conditions of fins (eroded fins on hatchery fish). Insufficient numbers of Rainbow Trout were caught during the recapture electrofishing to generate unbiased population estimates within the MGORD.

For the past 13 sampling years, electrofishing passes through the MGORD have produced the following total catch values (all size classes of Brown and Rainbow 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.

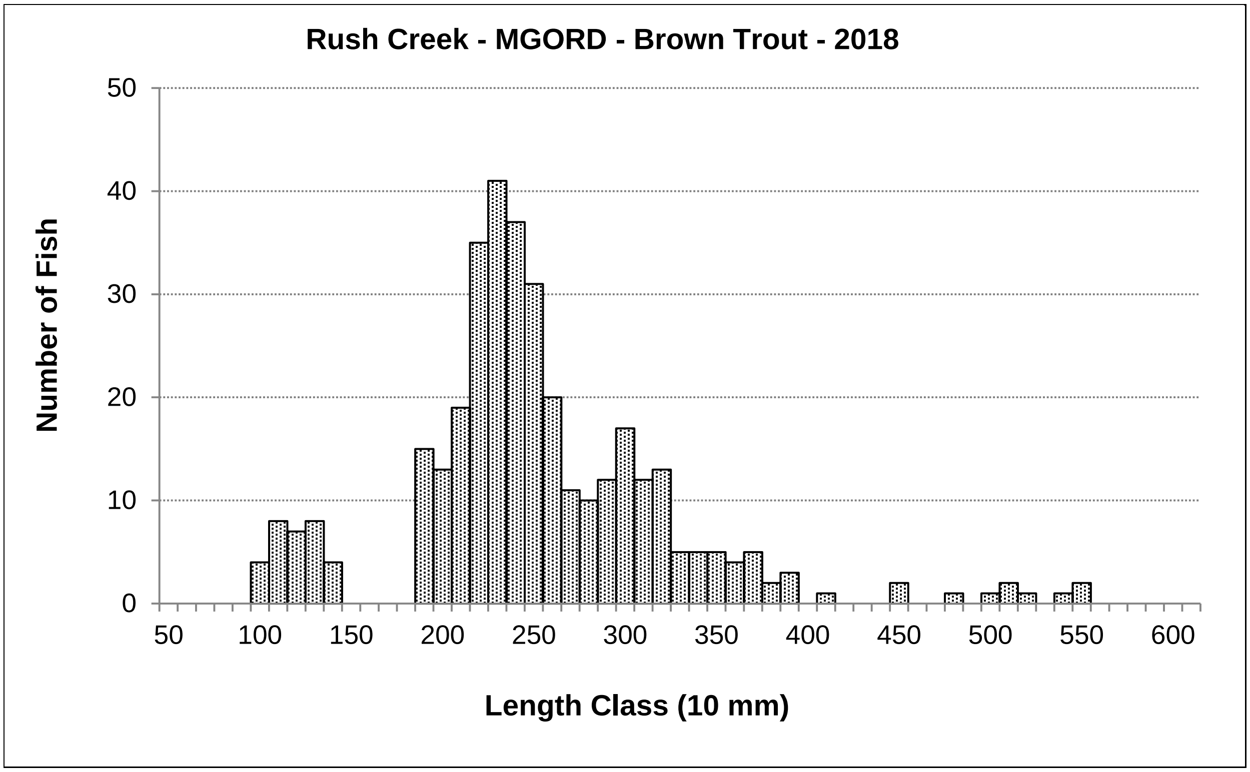


Figure 8. Length-frequency histogram of Brown Trout captured in the MGORD section of Rush Creek, September 20th and 27th, 2018.

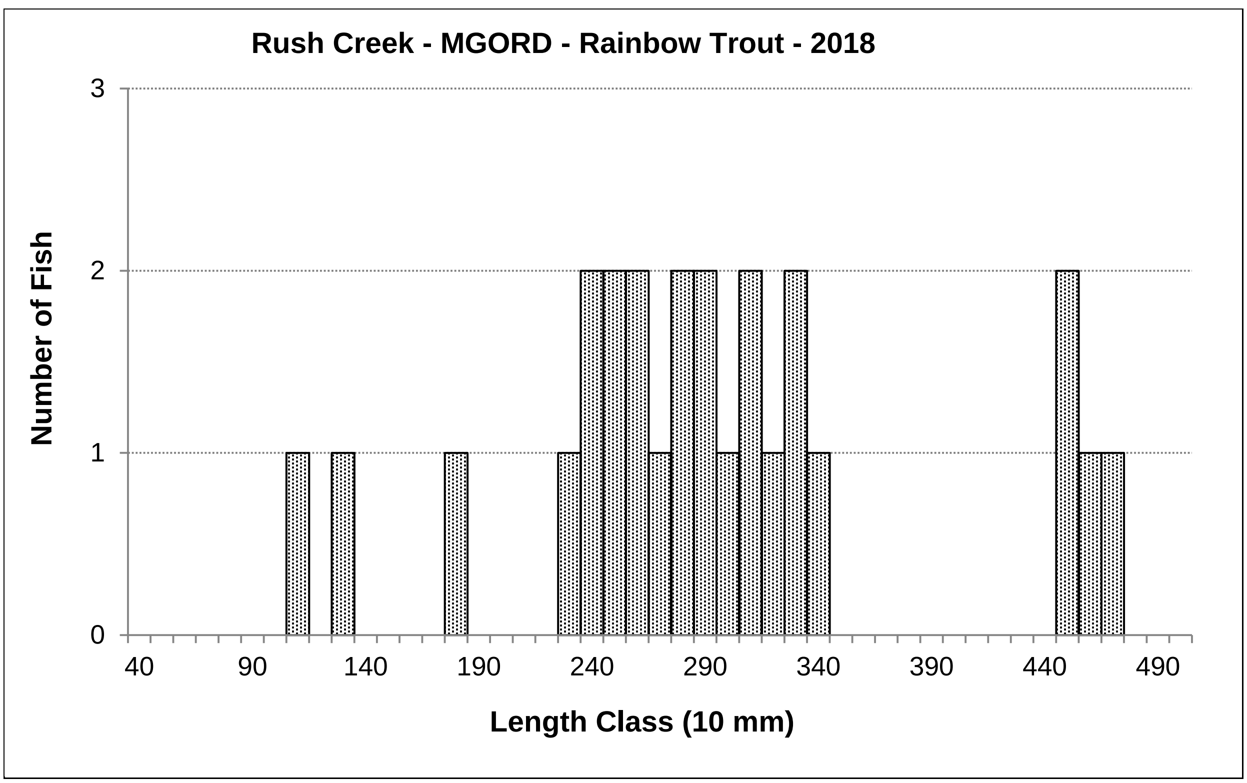


Figure 9. Length-frequency histogram of Rainbow Trout captured in the MGORD section of Rush Creek, September 20th and 27th, 2018.

Lee Vining Creek

In 2018, a total of 147 trout were captured in the Lee Vining Creek main channel section versus 55 trout trout in 2017, 246 fish in 2016, 422 fish in 2015 and 838 fish in 2012 (Table 3). Most (138 fish) of the trout captured in 2018 were Brown Trout and the nine Rainbow Trout were all age-0 fish (<125 mm in length). In 2018, Brown Trout ranged in size from 80 mm to 316 mm in length (Figure 10). Age-0 fish comprised 62% of the total Brown Trout catch in 2018, compared to 58% in 2017 and 28% in 2016. Lee Vining Creek's main channel section supported an estimated 192 age-0 Brown Trout in 2017, compared to an estimated 32 age-0 Brown Trout in 2017, a 500% decrease (Table 2). However, the 2018 estimate of 192 age-0 Brown Trout estimate was still 72% lower than the pre-drought estimate of 677 age-0 fish.

In 2018, Brown Trout 125-199 mm in length comprised 28% of the total Brown Trout catch in Lee Vining Creek's main channel section (versus 24% in 2017). This section supported an estimated 71 Brown Trout 125-199 mm in length in 2018 (Table 2) compared to 13 fish in 2017 (a 446% increase).

Lee Vining Creek's main channel supported an estimated 14 Brown Trout ≥200 mm in 2018 (versus 10 fish in 2017 and 50 fish in 2016) (Table 2). Two of the Brown Trout captured in 2018 were >300 mm in length (301 and 316 mm) (Figure 10).

No population estimate was generated for age-0 Rainbow Trout due to insufficient numbers of clipped fish (5 fish) and recaptures (one fish). No Rainbow Trout in the larger size classes (125-199 mm and ≥200 mm) were captured in the Lee Vining Creek main channel section in 2018.

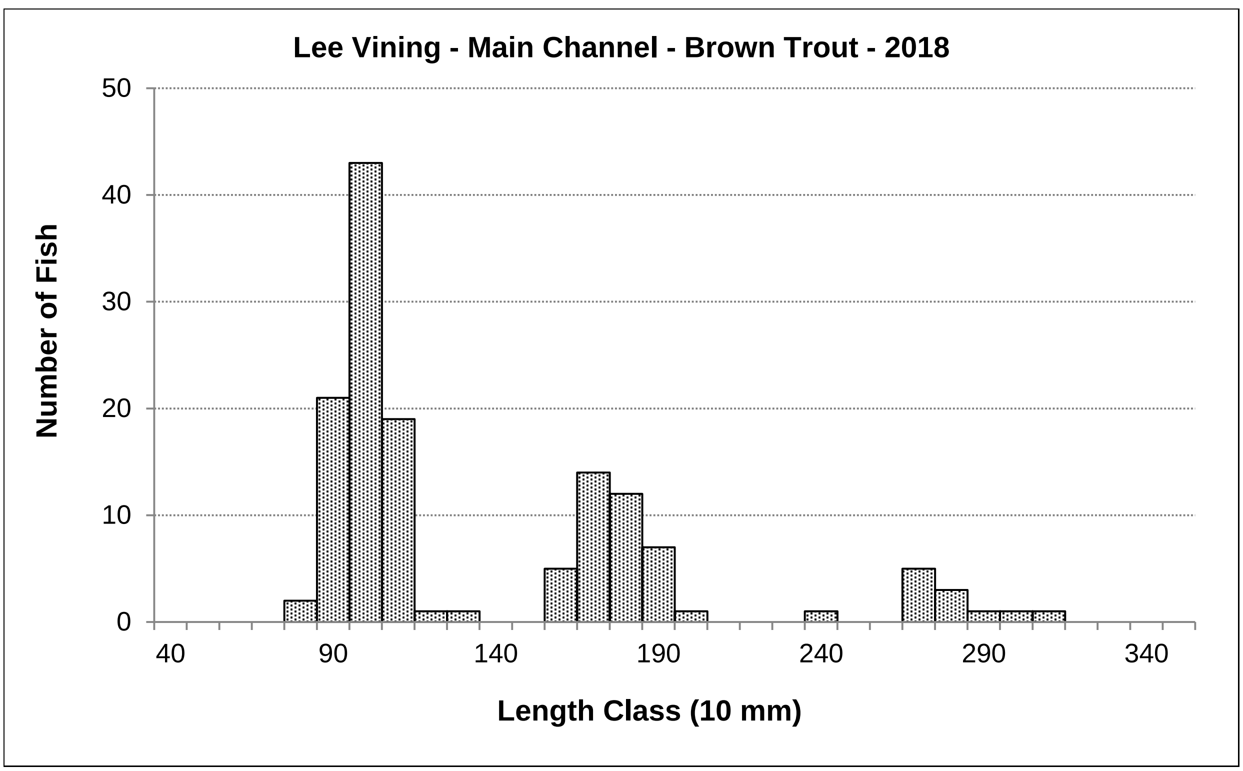


Figure 10. Length-frequency histogram of Brown Trout captured in the main channel section of Lee Vining Creek, September 19th and 26th, 2018.

In the Lee Vining Creek side channel, 10 Brown Trout were captured in two electrofishing passes made during the 2018 sampling (Table 3). Three age-0 fish were captured and the remaining seven fish were in the 125-199 mm size class (Figure 11). The estimates for the three size classes equaled the catch numbers (Table 3). No Rainbow Trout were captured in the side channel in 2018. This was the tenth consecutive year that no age-0 Rainbow Trout were captured in the Lee Vining Creek side channel and the eighth consecutive year that no age-1 and older Rainbow Trout were captured in the side channel.

Walker Creek

In 2018, 175 Brown Trout were captured in two electrofishing passes in the Walker Creek section (115 caught in 2017 and 312 in 2016) (Table 3). Forty-four of these captured fish, or 25%, were age-0 fish ranging in size from 84 mm to 112 mm in length (Figure 12). The 2018 estimated population of age-0 Brown Trout for this Walker Creek section was 44 fish, a 33% decrease from the 2017 estimate of 66 fish. For trout <125 mm in length, the estimated probability of capture during 2018 was 92% (Table 3).

Brown Trout in the 125-199 mm size class (86 fish) accounted for 49% of the total catch in 2018. The 2018 population estimate for Brown Trout in the 125-199 mm size class was 86 trout (an 83% increase from 2017 estimate) with an estimated probability of capture of 98% (Table 3).

Brown Trout ≥200 mm in length (45 fish) accounted for 26% of the total catch in 2018 (was 7% in 2017). The 2018 population estimate for this size class was 45 Brown Trout with a probability of capture of 100% because all 45 fish were caught on the first pass (Table 3). The largest Brown Trout captured in Walker Creek in 2018 was 274 mm in length (Figure 12).

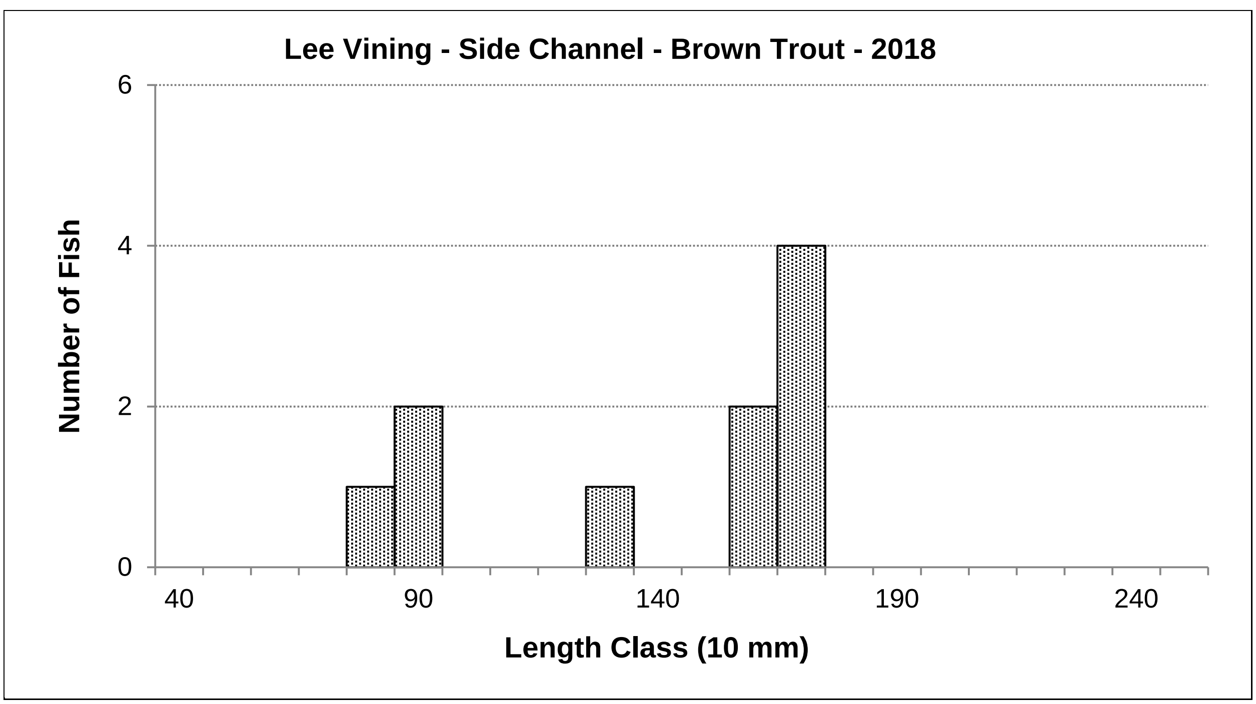


Figure 11. Length-frequency histogram of Brown Trout captured in the side channel section of Lee Vining Creek, September 21st, 2018.

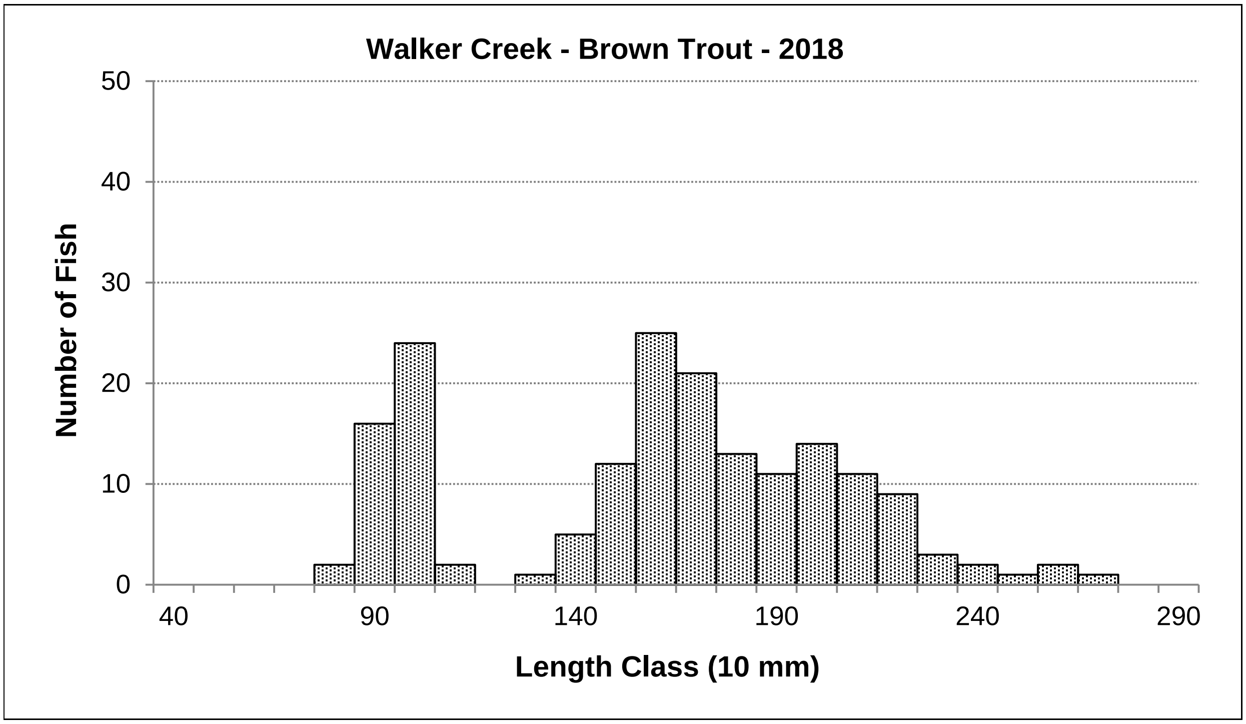


Figure 12. Length-frequency histogram of Brown Trout captured in Walker Creek, September 21st, 2018.

Table 3. Depletion estimates made in the side channel section of Lee Vining Creek and Walker Creek during September 2018 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 - 9/21/2018						
Brown Trout						
	0 - 124 mm	2	2	1	**3**	0.75
	125 - 199 mm	2	7	0	**7**	1.00
	200 + mm	2	0	0	**0**	N/A
Walker Creek - above old Hwy 395 - 9/21/2018						
Brown Trout						
	0 - 124 mm	2	40	4	**44**	0.92
	125 - 199 mm	2	84	2	**86**	0.98
	200 + mm	2	45	0	**45**	1.00

Catch of Rainbow Trout in Rush and Lee Vining Creeks

Beginning with the 2008 annual report, we have 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. In 2011, the last time GLR spilled significant amounts of water, hatchery-origin Rainbow Trout also spilled out of the reservoir. These spills resulted in Rainbow Trout accounting for 8% of the total catch in 2011, the highest we recorded in Rush Creek until 2017. For the sampling years since 2011; Rainbow Trout accounted for 5% of the total Rush Creek catch in 2012, 2% in 2013, 0.75% in 2014, 1.9% in 2015, and 2.5% in 2016. During the large snowmelt event of 2017, GLR spilled for 60 days and it appeared that fish originating from GLR came over the dam during these spills, as they likely did in 2011. For the 2017 sampling, Rainbow Trout comprised 10.9% of the total catch in Rush Creek (86 Rainbow Trout/787 total trout).

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. We suspect that Rainbow Trout numbers will decrease in the Upper Rush section as the Brown Trout population continues to rebound from the five years of drought.

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) and no rainbows in 2017. This large drop in Rainbow Trout numbers has occurred during the time period when CDFW shifted to stocking sterile catchable Rainbow Trout. We suggest 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.

Sufficient numbers of age-0 Rainbow Trout were captured in the main channel of Lee Vining Creek to generate population estimates for only four of the 18 years sampled (Table 4). Adequate numbers of age-1 and older Rainbow Trout were captured in the main channel to generate population estimates for eight of the 18 years sampled (Table 5). The side channel produced enough numbers of age-0 and age-1 and older Rainbow Trout to generate population estimates for six of the 18 years sampled (Tables 6 and 7). However, no age-0 Rainbow Trout have been caught in the side channel in the past 10 years and no age-1 and older Rainbow Trout have been caught in the past eight years (Tables 6 and 7).

Due to Rainbow Trout historically encompassing a large portion (10-40%) of the Lee Vining Creek trout population, an effort has been 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. While catch numbers are not statistically valid they were consistently lower than statistically valid estimates and allowed for comparison between all sampling years (Tables 4-7).

Table 4. Numbers of age-0 Rainbow Trout caught in Lee Vining Creek main channel section, 2000-2018.

Sample Year	Area of Sample Section (Ha)	Number of Trout on Marking Run	Number of Trout on Capture Run	Number of Recap Trout	Pop Estimate	Estimated Number of Trout per Hectare	Number of Trout Caught (Catch)	Catch per Hectare
2018	**0.1377**	**5**	**5**	**1**	**NP**	**NP**	**9**	**65**
2017	0.1377	0	0	0	0	0	0	0
2016	0.1352	0	0	0	0	0	0	0
2015	0.1224	0	0	0	0	0	0	0
2014	0.1403	4	4	2	NP	NP	6	43
2013	0.1454	19	12	5	40	275	26	179
2012	0.1279	155	138	67	318	2,494	226	1,773
2011	0.1428	1	0	0	NP	NP	1	7
2010	0.1505	0	0	0	0	0	0	0
2009	0.1505	4	4	0	NP	NP	8	53
2008	0.1377	17	31	9	57	414	39	283
2007	0.0884	42	56	22	106	1,199	76	860
2006	NS*	--	--	--	--	--	--	--
2005	0.0744	0	0	0	0	0	0	0
2004	0.0744	1	0	0	NP	NP	1	13
2003	0.0744	0	0	0	0	0	0	0
2002	0.0744	0	1	0	NP	NP	1	13
2001	0.0898	3	5	1	NP	NP	7	78
2000	0.0898	0	1	0	NP	NP	1	22

*NS stands for not sampled due to high flows

Table 5. Numbers of age-1 and older Rainbow Trout caught in Lee Vining Creek main channel section, 2000-2018.

Sample Year	Area of Sample Section (Ha)	Number of Trout on Marking Run	Number of Trout on Capture Run	Number of Recap Trout	Pop Estimate	Estimated Number of Trout per Hectare	Number of Trout Caught (Catch)	Catch per Hectare
2018	**0.1377**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
2017	0.1377	0	0	0	0	0	0	0
2016	0.1352	7	5	5	7	52	7	52
2015	0.1224	18	14	12	21	172	20	163
2014	0.1403	36	36	21	63	449	51	364

2013	0.1454	61	45	29	120	826	77	530
2012	0.1279	7	7	5	NP	NP	9	71
2011	0.1428	5	8	5	NP	NP	8	56
2010	0.1505	12	9	7	15	100	14	93
2009	0.1505	39	32	12	98	651	59	392
2008	0.1377	71	64	37	129	936	98	712
2007	0.0884	3	5	1	NP	NP	7	79
2006	NS*	--	--	--	--	--	--	--
2005	0.0744	3	3	0	NP	NP	6	81
2004	0.0744	2	2	2	NP	NP	2	27
2003	0.0744	5	6	5	NP	NP	6	81
2002	0.0744	10	10	7	14	188	13	175
2001	0.0898	9	8	4	NP	NP	13	145
2000	0.0898	1	3	0	NP	NP	4	45

Table 6. Numbers of age-0 Rainbow Trout caught in Lee Vining Creek side channel section, 2000-2018.

Sample Year	Area of Sample Section (Ha)	Number of Trout Caught on Pass #1	Number of Trout Caught on Pass #2	Number of Trout Caught on Pass #3	Pop Estimate	Estimated Number of Trout per Hectare	Number of Trout Caught (Catch)	Catch per Hectare
2018	**0.0507**	**0**	**0**	**--**	**0**	**0**	**0**	**0**
2017	0.0389	0	0	--	0	0	0	0
2016	0.0233	0	0	--	0	0	0	0
2015	0.0328	0	0	--	0	0	0	0
2014	0.0191	0	0	--	0	0	0	0
2013	0.0195	0	0	--	0	0	0	0
2012	0.0365	0	0	--	0	0	0	0
2011	0.0507	0	0	--	0	0	0	0
2010	0.0507	0	0	--	0	0	0	0
2009	0.0488	0	0	--	0	0	0	0
2008	0.0488	5	2	--	7	143	7	143
2007	0.0488	4	0	--	NP	NP	4	82
2006	0.0761	46	26	--	100	1,314	72	946
2005	0.0936	0	0	--	0	0	0	0
2004	0.0936	82	30	--	127	1,357	112	1,197
2003	0.0936	0	0	--	0	0	0	0
2002	0.0936	28	17	--	64	684	45	481
2001	0.1310	69	23	--	102	779	92	702
2000	0.0945	32	15	--	57	603	47	497

Table 7. Numbers of age-1 and older Rainbow Trout caught in Lee Vining Creek side channel section, 2000-2018.

Sample Year	Area of Sample Section (Ha)	Number of Trout Caught on Pass #1	Number of Trout Caught on Pass #2	Number of Trout Caught on Pass #3	Pop Estimate	Estimated Number of Trout per Hectare	Number of Trout Caught (Catch)	Catch per Hectare
2018	**0.0507**	**0**	**0**	**--**	**0**	**0**	**0**	**0**
2017	0.0389	0	0	--	0	0	0	0
2016	0.0233	0	0	--	0	0	0	0
2015	0.0328	0	0	--	0	0	0	0
2014	0.0191	0	0	--	0	0	0	0
2013	0.0195	0	0	--	0	0	0	0
2012	0.0365	0	0	--	0	0	0	0
2011	0.0507	0	0	--	0	0	0	0
2010	0.0507	1	0	--	1	20	1	20
2009	0.0488	15	0	--	15	307	15	307
2008	0.0488	3	1	--	4	82	4	82
2007	0.0488	6	0	--	NP	NP	6	123
2006	0.0761	5	0	--	NP	NP	5	66
2005	0.0936	7	2	--	9	96	9	96
2004	0.0936	5	0	--	NP	NP	5	53
2003	0.0936	13	0	--	NP	NP	13	139
2002	0.0936	29	4	--	33	353	33	353
2001	0.1310	38	3	--	41	313	41	313
2000	0.0945	9	0	--	NP	NP	9	95

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

Table 8. 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 2018 regression equations are in ***bold*** type.

Section	Year	N	Equation	r^2	P
Bottomlands	***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

Table 8 (continued).

Section	Year	N	Equation	r^2	P
Bottomlands	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	***2018***	***391***	$\boldsymbol{Log_{10}(WT) = 2.9173*Log_{10}(L) – 4.8237}$	***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
	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	***2018***	***350***	$\mathbf{Log_{10}(WT) = 3.0023*Log_{10}(L) – 5.0046}$	**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

Table 8 (continued).

Section	Year	N	Equation	r^2	P
	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 2018 decreased from 2017's values in four sections and increased slightly in two other sections (Figures 13 and 14). In 2018, four sections (MGORD, Walker Creek, Lee Vining main channel and Lee Vining side channel) had Brown Trout condition factors ≥1.00 (Figures 13 and 14).

Brown Trout in the Upper Rush section had a condition factor of 0.96 in 2018, a decrease from 1.04 in 2017 (Figure 13). The Upper Rush section has had Brown Trout condition factors ≥1.00 in 11 of 19 sampling seasons (Figure 13).

Brown Trout in the Bottomlands section of Rush Creek had a condition factor of 0.92 in 2018, a decrease from the value of 0.99 in 2017 (Figure 13). In 11 years of sampling, the Bottomlands section has failed to generate a Brown Trout condition factor ≥1.00 (Figure 13).

The MGORD's 2018 Brown Trout condition factor was 1.01, an increase from the 2017 value of 0.97. In 2018, condition factors for larger Brown Trout in the MGORD were also computed: fish ≥300 mm had a condition factor of 1.00 (0.99 in 2017) and fish ≥375 mm had a condition factor of 1.01 (1.02 in 2017).

In 2018, the condition factors for Brown Trout in Lee Vining Creek's main channel and side channel equaled 1.03 in both sections (Figure 14). The 2018 values were slight decreases from 2017 values (Figure 14). For the eighth year in a row, no Rainbow Trout were captured in the Lee Vining Creek side channel.

In Walker Creek, Brown Trout had a condition factor of 1.02 in 2018, an increase from 0.97 in 2017 (Figure 13). Brown Trout condition factors in Walker Creek have been ≥1.00 in 12 of the 19 sampling years (Figure 13).

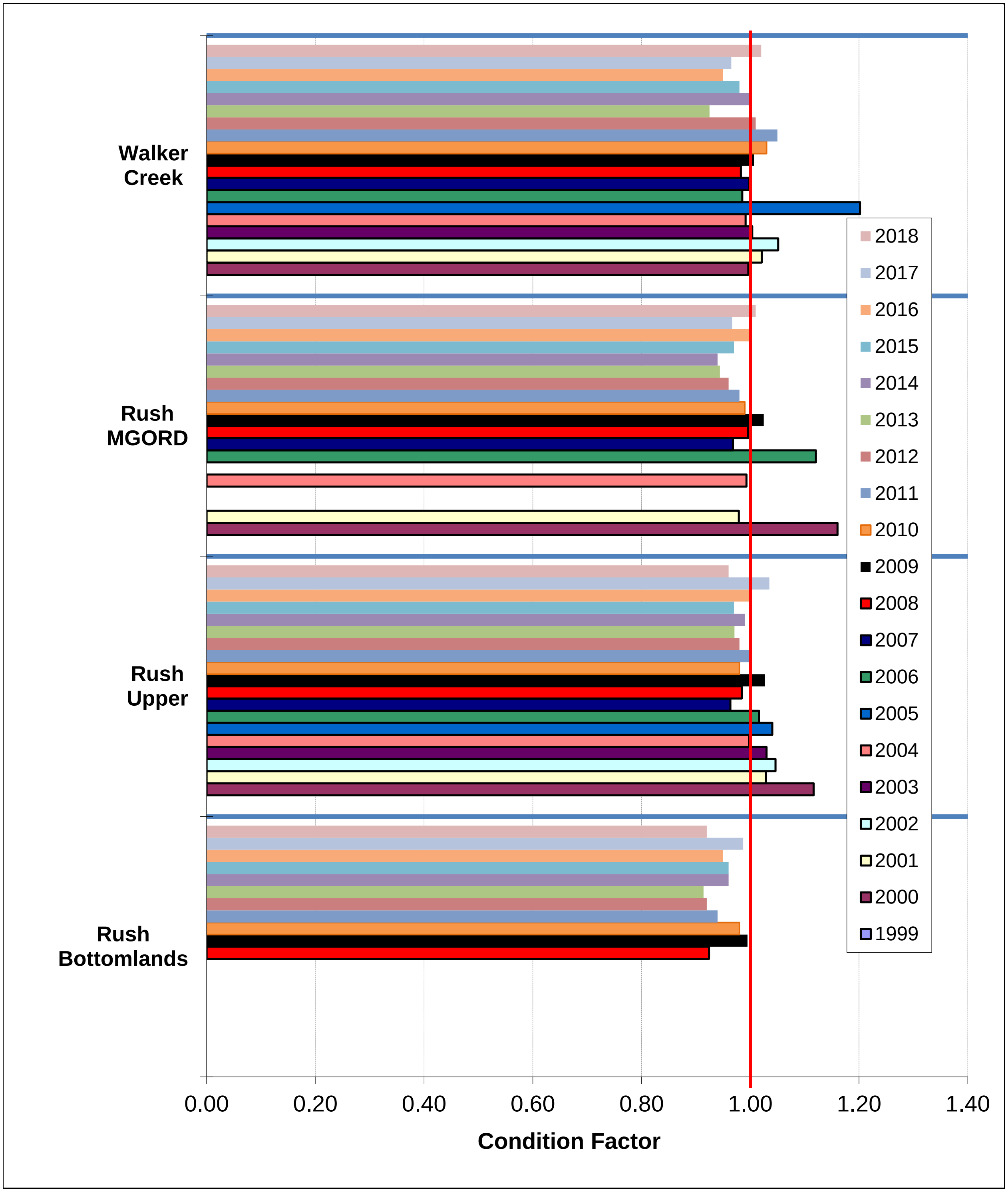


Figure 13. Condition factors for Brown Trout 150 mm to 250 mm in length from sample sections of Rush Creek and Walker Creeks from 1999 to 2018.

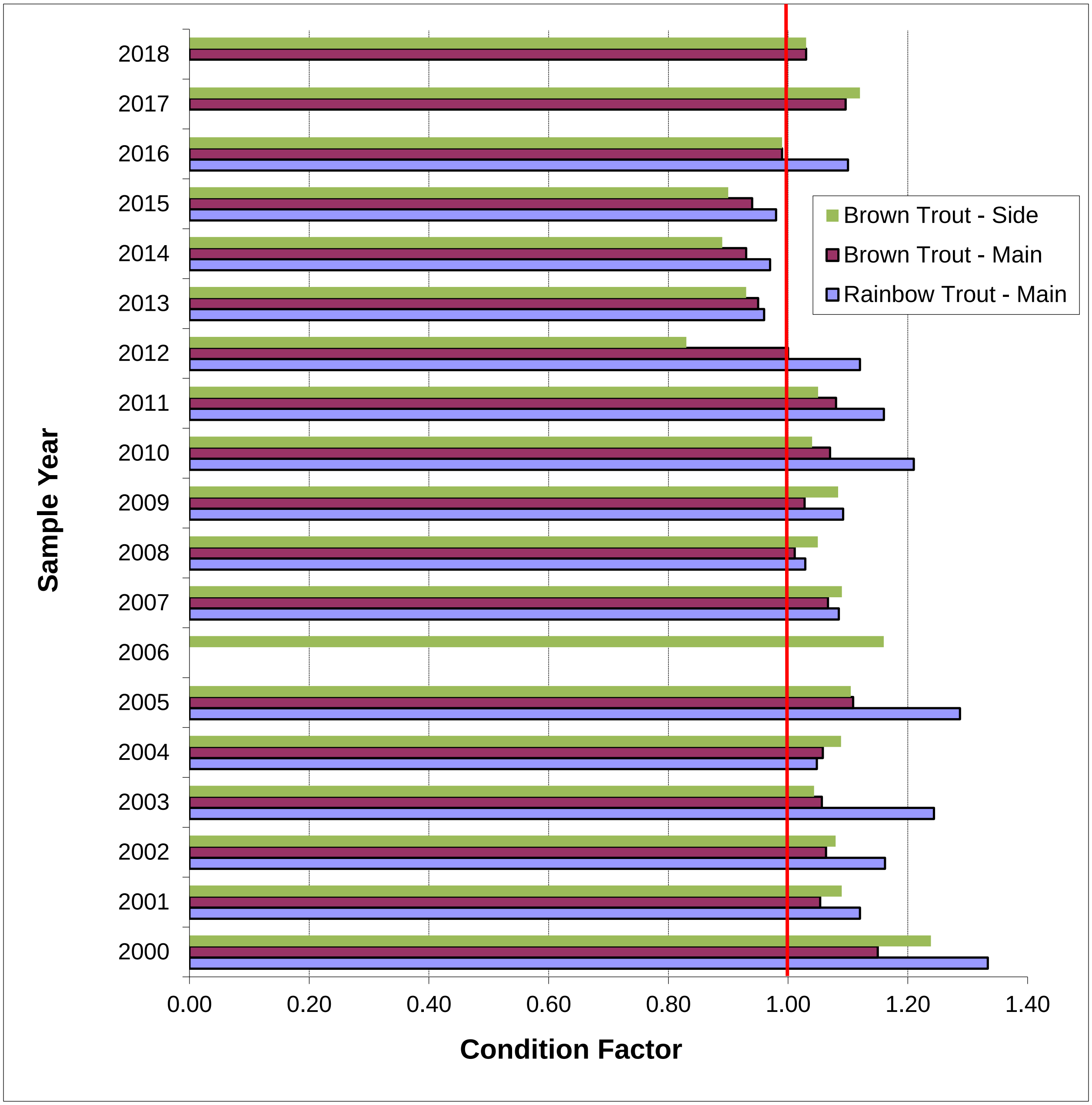


Figure 14. Comparison of condition factors for Rainbow Trout and Brown Trout 150 to 250 mm in length from the main channel and side channel sections of Lee Vining Creek from 2000 to 2018. Main channel was not sampled in 2006 due to high flows. No Rainbow Trout 150 to 250 mm in length were captured in 2017 and 2018.

Estimated Trout Densities Expressed in Numbers per Hectare

Age-0 Brown Trout

The Upper Rush section had an estimated density of 4,502 age-0 Brown Trout/ha in 2018, an increase of 134% from 2017's estimate of 1,923 age-0 trout/ha (Figure 15). After a 95% decrease during the five consecutive dry/below average RYs (2012-2016), age-0 Brown Trout density estimates have increased from the 2016 low of 439 fish/ha to 4,502 fish/ha (a nine-fold increase). The 2018 density estimate in the Upper Rush section was 20% lower than the 20-year average of 5,658 age-0 Brown Trout/ha.

The Bottomlands section of Rush Creek had a density estimate of 5,444 age-0 Brown Trout/ha in 2018, an increase of 1,073% from 2017's estimate of 464 age-0 trout/ha (Figure 15). After an 82% decrease during the five consecutive dry/below average RYs, age-0 Brown Trout density estimates have increased to the highest estimate generated for the Bottomlands section. When compared to the 11-year average of 2,093 age-0 Brown Trout/ha, the 2018 estimate was 160% higher.

In Walker Creek, the 2018 density estimate of 1,086 age-0 Brown Trout/ha decreased 28% compared to the 2017 estimate of 1,503 age-0 trout/ha (Figure 15). The 2018 density estimate was 69% lower than the 20-year average of 3,470 age-0 trout/ha (Figure 15).

In 2018, the estimated density of age-0 Brown Trout in the main channel section of Lee Vining Creek was 1,394 age-0 trout/ha, which was a 500% increase from the 2017 density estimate of 232 age-0 trout/ha (Figure 16). After a 96% decrease during the five consecutive dry/below average RYs, the age-0 Brown Trout density estimates increased 500%. The 2018 estimate was 14% lower than the 19-year average of 1,613 age-0 Brown Trout/ha (Figure 16).

In 2018, the age-0 Brown Trout density estimate in the side channel section of Lee Vining Creek was 59 age-0 trout/ha, which was an 86% decrease from the 2017 density estimate of 411 age-0 trout/ha (Figure 16).The 2018 estimate was 82% lower than the 20-year average of 336 age-0 Brown Trout/ha (Figure 16).

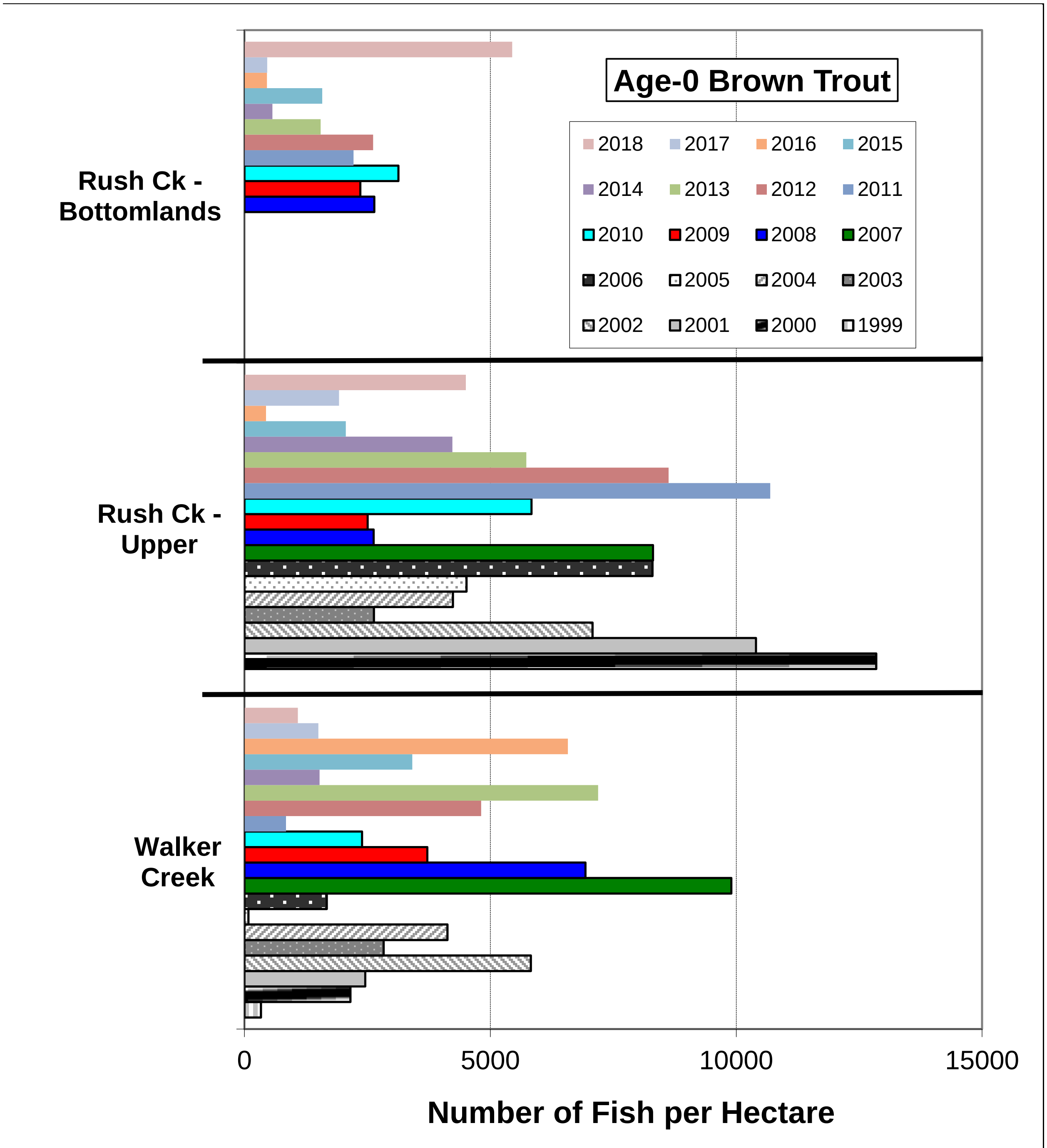


Figure 15. Estimated number of age-0 Brown Trout per hectare in Rush Creek and Walker Creek from 1999 to 2018.

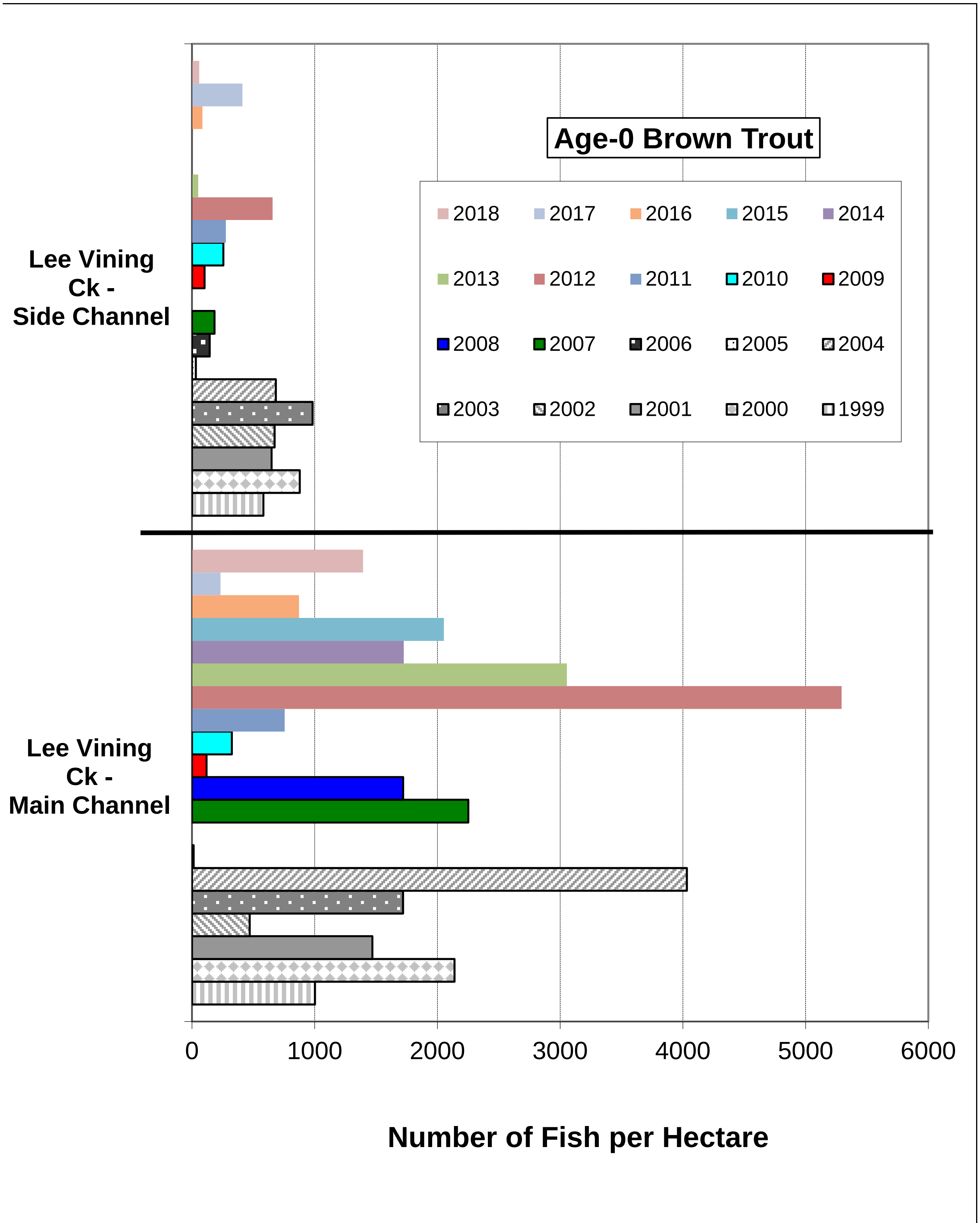


Figure 16. Estimated number of age-0 Brown Trout per hectare in Lee Vining Creek from 1999 to 2018.

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

The Upper Rush section had an estimated density of 1,120 age-1+ Brown Trout/ha in 2018, an increase of 86% from the 2017 estimate of 594 trout/ha (Figure 17). After a 75% decrease during the five consecutive dry/below average RYs, the age-1+ Brown Trout density estimates have increased by 126% during the 2017 and 2018 sampling seasons. The 2018 estimate was 17% lower than the 20-year average of 1,355 age-1+ Brown Trout/ha.

The estimated density of age-1+ Brown Trout in the Bottomlands section of Rush Creek during 2018 was 620 fish/ha in 2018, a 43% increase from the 2017 estimate of 433 age-1+trout/ha (Figure 17). After an 86% decrease during the five consecutive dry/below average RYs, the age-1+ Brown Trout density estimates have increased by 153% during the 2017 and 2018 sampling seasons. The 2018 density estimate of age-1+ Brown Trout/ha was 40% lower than the 11-year average of 1,034 age-1+ Brown Trout/ha.

The estimated density of age-1+ Brown Trout in the MGORD section of Rush Creek eqauled 430 age-1+ Brown Trout/ha in 2018, a 165% increase from the 2016 estimate of 162 age-1+trout/ha (Figure 17). The 2018 density estimate of age-1+ Brown Trout/ha was 4% lower than the average of 448 age-1+ Brown Trout/ha generated from the nine years where density estimates were produced for the MGORD section.

The 2018 density estimate for age-1+ Brown Trout for the Walker Creek section was 3,235 age-1+trout/ha which was a 158% increase from the 2017 estimate of 1,253 age-1+ trout/ha (Figure 17). The 2018 density estimate of age-1+ Brown Trout was 76% higher than the 20-year average of 1,822 age-1+ Brown Trout/ha.

The 2018 density estimate for age-1+ Brown Trout in the Lee Vining main channel section was 617 trout/ha, a 270% increase from the 2017 estimate of 167 age-1+ trout/ha(Figure 18). The 2018 estimate was still 75% lower than the 2013 estimate of 2,449 age-1+ Brown Trout/ha (Figure 18). The 2018 density estimate of age-1+ Brown Trout was 43% lower than the 19-year average of 1,091 age-1+ Brown Trout/ha.

In 2018, the side channel of Lee Vining Creek supported an estimated density of 138 age-1+ Brown Trout/ha, a decrease of 23% from the 2017 estimate of 180 age-1+ Brown Trout/ha (Figure 18). As discussed in last year's annual report, 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 9). 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 9). However, the total number of Brown Trout caught in 2018 was the second lowest for the past 12 years (Table 9).

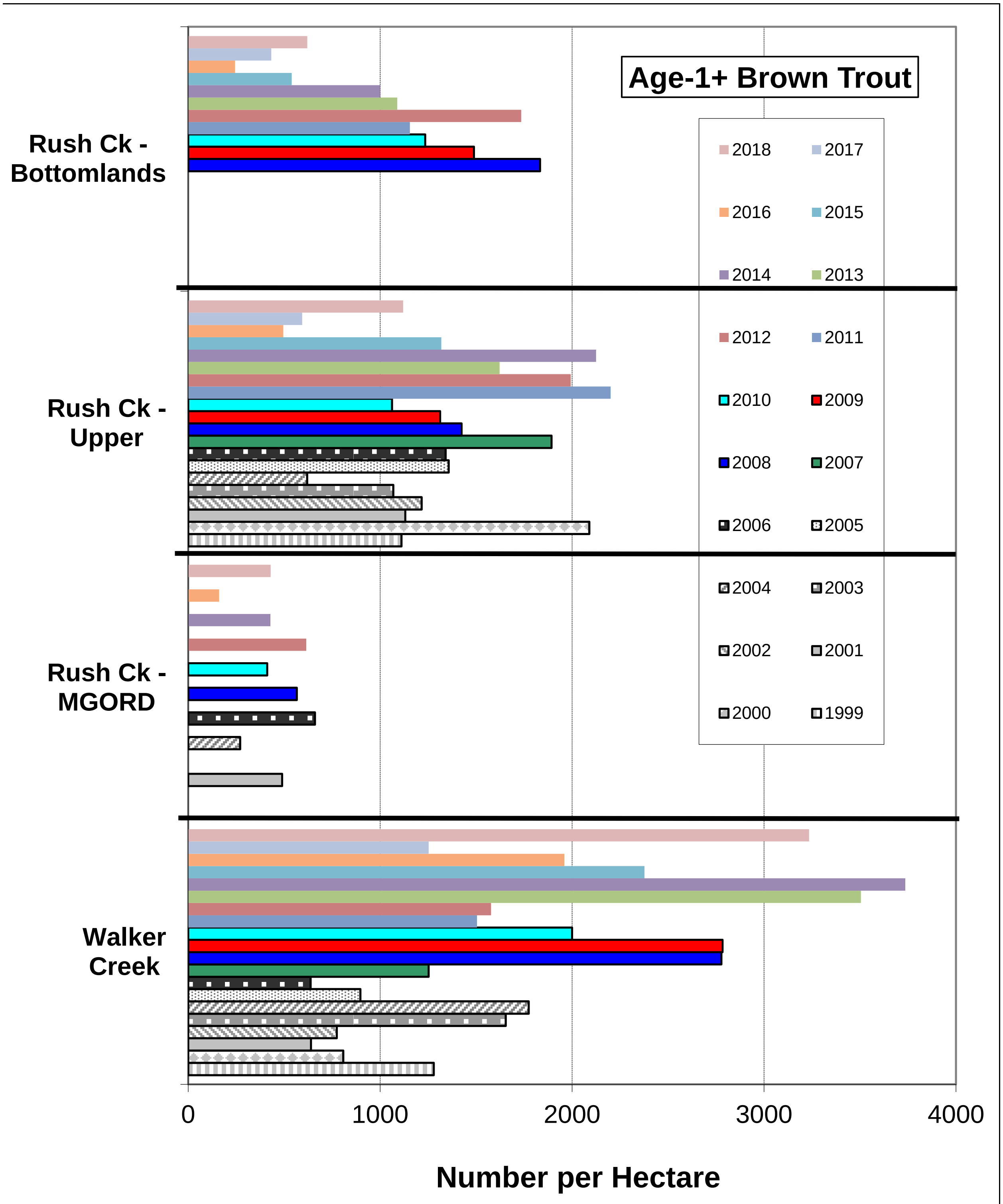


Figure 17. Estimated number of age-1 and older Brown Trout per hectare in sections of Rush and Walker Creeks from 1999 to 2018.

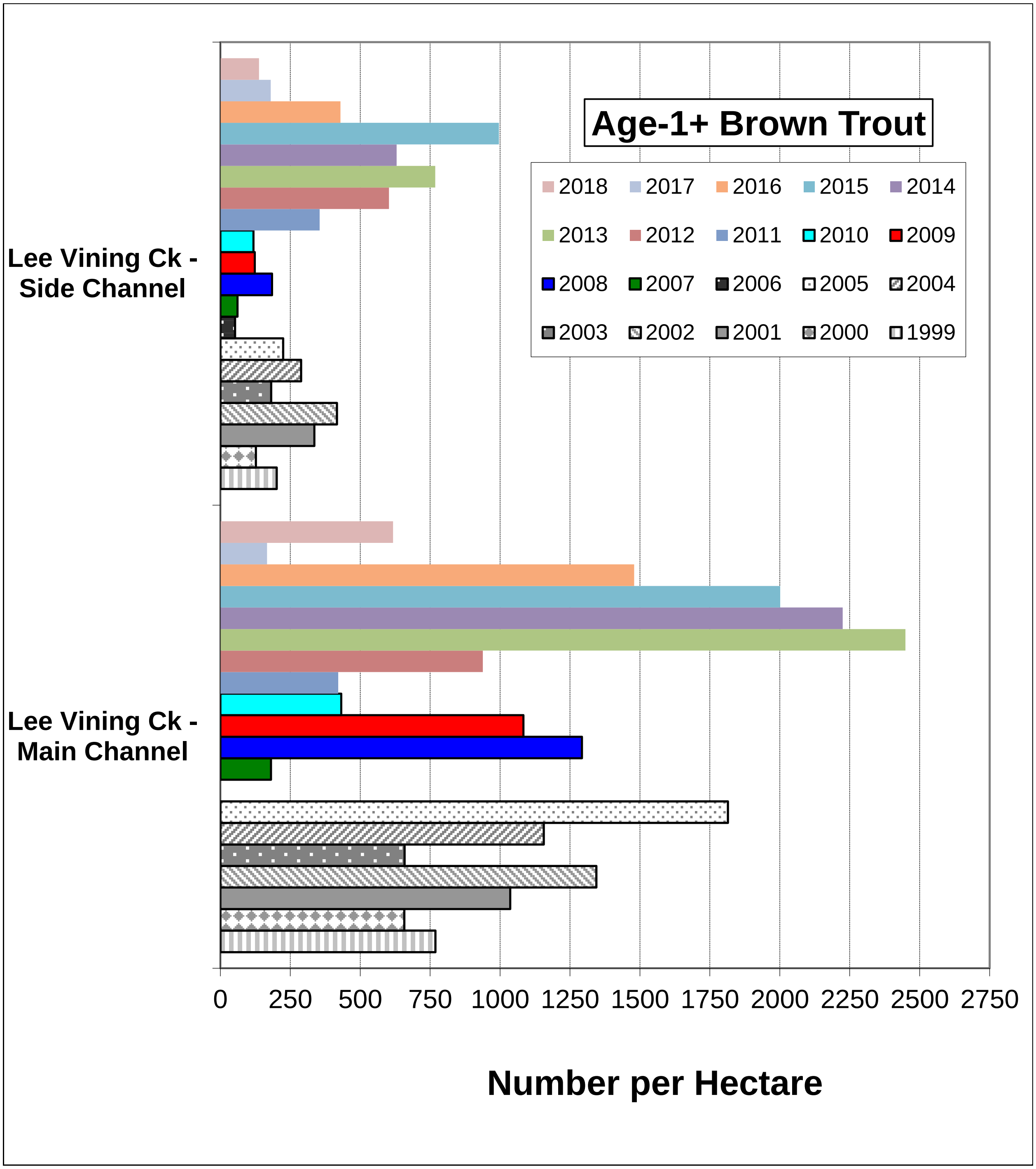


Figure 18. Estimated number of age-1 and older Brown Trout per hectare in sections of Lee Vining Creek from 1999 to 2018.

Table 9. Wetted surface area and total numbers of trout captured in the Lee Vining Creek side channel, from 2007 to 2018.

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**

Age-0 Rainbow Trout

In 2018, for the tenth consecutive year no age-0 Rainbow Trout were captured in the Lee Vining Creek side channel. In the Lee Vining Creek main channel, only nine age-0 Rainbow Trout were captured during the 2018 sampling.

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

In 2018, for the eighth consecutive year no age-1 and older Rainbow Trout were captured in the Lee Vining Creek side channel. In 2018, for the second time in 20 sampling years, no age-1 and older Rainbow Trout were captured in the Lee Vining Creek main channel.

Estimated Numbers of Trout per Kilometer

The Upper Rush section contained an estimated 4,835 Brown Trout/km (all size classes combined) in 2018, which was a 160% increase from the 2017 estimate of 1,863 Brown Trout/km (Table 10). The 2018 estimate was the second straight increase of the estimated numbers of Brown Trout/km since five years of declines during the drought (Table 10). The estimated density of age-1+ Brown Trout in 2018 was 963 fish/km, a 119% increase from the 2017 estimate of 440 age-1+ fish/km (Table 10).

The Bottomlands section contained an estimated of 4,608 Brown Trout/km (all size classes combined) in 2018, which was a 623% increase from the 2017 estimate of 637 fish/km (Table 10). The 2018 estimate was the second straight increase of the estimated numbers of Brown Trout/km since five years of declines during the drought. In 2018, the estimate of 471 age-1+ Brown Trout/km represented a 53% increase from the 2017 estimate of 308 age-1+ Brown Trout/km (Table 10).

The Lee Vining Creek main channel contained an estimated 1,189 Brown Trout/km (all size classes combined) in 2018, which was a 450% increase from the 2017 estimate of 216 fish/km (Table 11). In 2018, the estimate of 436 age-1+ Brown Trout/km represented a 384% increase from the 2017 estimate of 90 age-1+ trout/km (Table 11).

The Lee Vining side channel contained an estimated 51 Brown Trout/km in (all size classes combined) 2018, a 61% decrease from the 2017 estimate of 130 fish/km (Table 11). For age-1+ Brown Trout, the 2018 density estimate was 36 Brown Trout/km which was a 10% decrease from the 2017 density estimate 40 fish/km (Table 11).

The Lee Vining Creek main channel and the side channel estimates of total numbers of trout per kilometer were added in order to compare to the proposed termination criteria as discussed in the 2011 Annual Fisheries Report (Taylor and Knudson 2012). When combined, the two channels contained an estimated 637 Brown Trout/km in 2018, an increase of 254% from the 2017 estimate of 180 Brown Trout/km (Table 11). Age-1+ trout in these two channels contained a combined estimate of 204 fish/km in 2018, a 196% increase from 69 fish/km in 2017 (Table 11).

Table 10. 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 2007 to 2018.

Collection Location	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Rush Creek, Upper Rush	8,698 (1,621)	3,607 (1,267)	3,444 (1,186)	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)**
Rush Creek, Bottom-lands	N/A	3,579 (1,467)	2,961 (1,146)	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)**

Table 11. 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 2007 to 2018.

Collection Location	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Lee Vining, Main Channel	2,103 (148)	2,357 (1,204)	1,192 (1,023)	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**)
Lee Vining, Side Channel	129 (62)	103 (67)	133 (108)	103 (36)	159 (87)	257 (123)	131 (123)	95 (95)	100 (100)	97 (97)	130 (40)	**51 (36)**
LV Main + LV Side Additive Approach	1,116 (105)	1,230 (636)	663 (566)	311 (181)	443 (173)	2,668 (348)	2,588 (1,302)	1,662 (1,013)	1,591 (819)	860 (554)	180 (69)	**637 (204)**

Estimated Trout Standing Crops (kg/ha)

The total (Brown and Rainbow Trout) estimated standing crop in the Upper Rush section was 188 kg/ha in 2018, a 52% increase from the 2017 estimate of 123 kg/ha (Table 12 and Figure 19). Rainbow Trout comprised 18.7 kg/ha of the 2018 standing crop estimate. When compared to the 20-year average of 148 kg/ha, the 2018 standing crop estimate was approximately 27% greater (Figure 19).

The estimated standing crop for Brown Trout in the Bottomlands section of Rush Creek was 103 kg/ha in 2018, a 103% increase from 50 kg/ha in 2017 (Table 12 and Figure 19). When compared to the 11-year average of 81 kg/ha, the 2018 standing crop estimate was approximately 27% greater (Figure 19).

Although there is not a standing crop termination criterion for Walker Creek, an estimate was still generated for this annually-sampled section. The estimated standing crop for Brown Trout in Walker Creek was 245 kg/ha in 2018, a 188% increase from the 2017 estimate of 85 kg/ha (Table 12 and Figure 19). The 2018 standing crop estimate was the greatest value recorded in Walker Creek over the 20-year sampling period and the long-term average for this period is 137 kg/ha.

Although there is not a standing crop termination criterion for the MGORD section of Rush Creek, an estimate was still generated for this even-year sampled section. The estimated standing crop for Brown Trout in the MGORD was 95 kg/ha in 2018, a 132% increase from the 2016 estimate of 41 kg/ha (Figure 19). For the nine seasons between 2001 and 2018 that data were available, the long-term average standing crop for the MGORD is 88 kg/ha.

The estimated total standing crop for Brown Trout in the Lee Vining Creek main channel in 2018 was 70 kg/ha; an increase of 233% from the 2017 estimate of 21 kg/ha (Table 13 and Figure 20). The 2018 estimated standing crop of 70 kg/ha was 43% lower than the 19-year average of 122 kg/ha.

The estimated standing crop of Brown Trout in the Lee Vining Creek side channel was 7 kg/ha in 2018, which represented a 65% decrease from 2017's estimate of 20 kg/ha (Table 13 and Figure 20). The 2018 estimate was also the lowest recorded for this section (Figure 20). No Rainbow Trout were captured in the Lee Vining Creek side channel in 2018 and none have been sampled in the side channel section for eight consecutive years (2011-2017).

When estimates of standing crops were combined for the side and main channel section of Lee Vining Creek, the total was 53 kg/ha for 2018, a 152% increase from the 2017 estimate of 21 kg/ha (Table 13).

Table 12. Comparison of Brown Trout standing crop (kg/ha) estimates between 2032 and 2018 for Rush Creek sections. These six years cover four drier years of 2013-2016, followed by the extremely wet RY 2017 and the normal RY 2018.

Collection Location	2013 Total Standing Crop (kg/ha)	2014 Total Standing Crop (kg/ha)	2015 Total Standing Crop (kg/ha)	2016 Total Standing Crop (kg/ha)	2017 Total Standing Crop (kg/ha)	2018 Total Standing Crop (kg/ha)	Percent Change Between 2017 and 2018
Rush Creek – Upper	140	167	123	62	123	188*	+53%
Rush Creek - Bottomlands	55	52	59	34	50	103	+106%
Walker Creek	194	189	183	172	85	245	+188%

*includes 18.7 kg/ha of Rainbow Trout

Table 13. Comparison of total (Brown and Rainbow Trout) standing crop (kg/ha) estimates between 2013 and 2018 for the Lee Vining Creek sections. These six years cover four drier years of 2013-2016, followed by the extremely wet RY 2017 and the normal RY 2018.

Collection Location	2013 Total Standing Crop (kg/ha)	2014 Total Standing Crop (kg/ha)	2015 Total Standing Crop (kg/ha)	2016 Total Standing Crop (kg/ha)	2017 Total Standing Crop (kg/ha)	2018 Total Standing Crop (kg/ha)	Percent Change Between 2017 and 2018
Lee Vining Creek - Main Channel	184	140	150	113	21	70	+132%
Lee Vining Creek – Side Channel	26	30	45	31	20	7	-65%
Lee Vining Main/Side Channels Combined	165	126	145	101	21	53	-80%

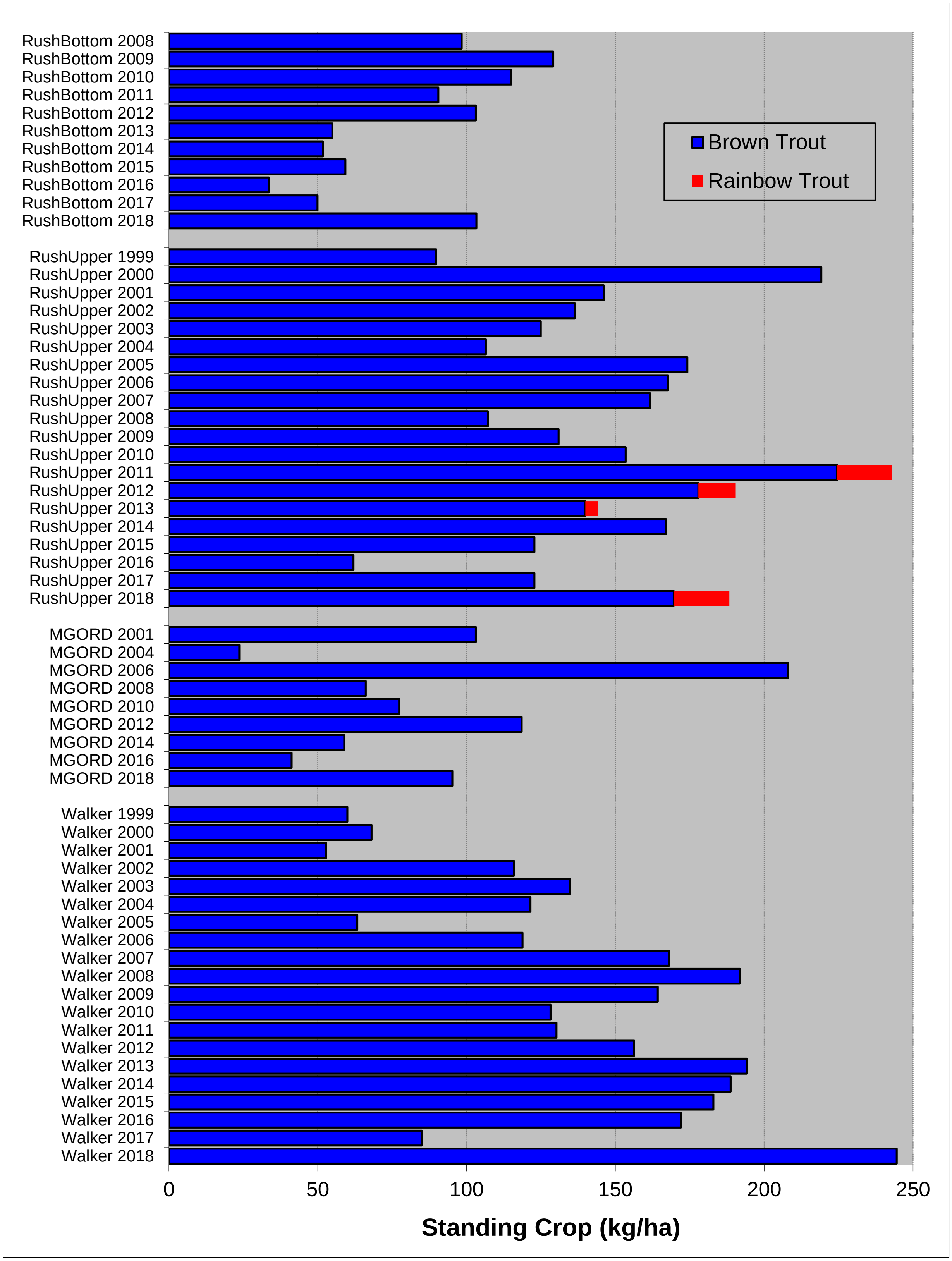


Figure 19. Estimated total standing crop (kilograms per hectare) of Brown Trout in Rush Creek sample sections from 1999 to 2018. NOTE: After 2001, MGORD estimates only made during even years.

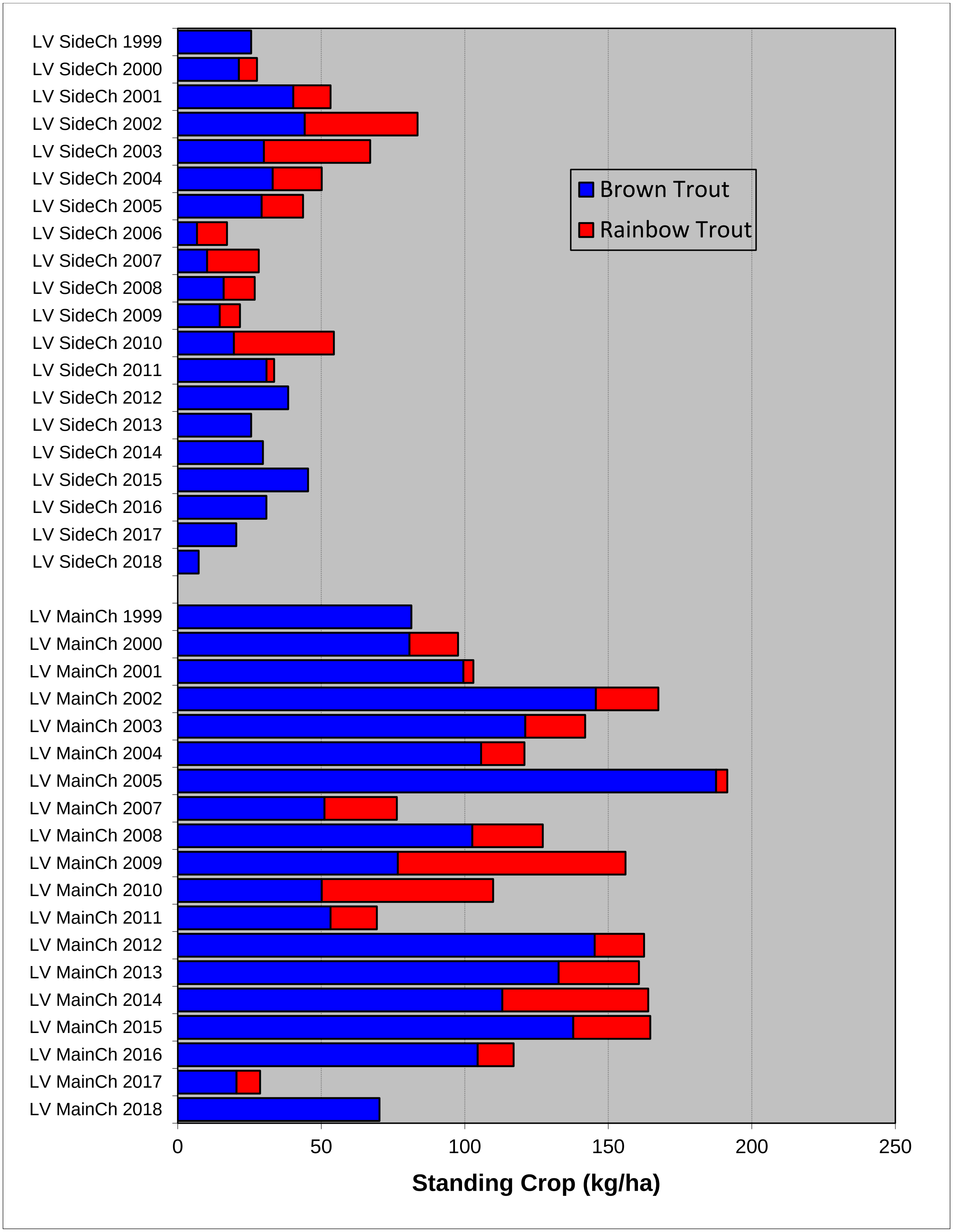


Figure 20. Estimated total standing crop (kilograms per hectare) of Brown Trout and Rainbow Trout (red) in Lee Vining Creek sample sections from 1999 to 2018.

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

In the Upper Rush section, the RSD-225 equaled 39 for 2018, a large drop from the record RSD-255 value of 78 for 2017 (Table 14). The 2018 RSD-225 value was most likely influenced by greater numbers of fish, especially the numbers of fish smaller than 225 mm which comprised 61% of the trout ≥150 mm (Table 14). The RSD-300 value was 9 in 2018, compared to 15 in 2017 (Table 14). This drop in RSD-300 value was influenced by the higher numbers of fish 225 mm and smaller caught in 2018, because although the RSD-300 dropped between 2017 and 2018, the actual numbers of Brown Trout >300 mm captured increased from 20 fish in 2017 to 24 fish in 2018 (Table 14). PIT tag recapture data from 2018 documented a second straight year of good growth rates in the Upper Rush section, which showed that some Brown Trout approached or exceeded 300 mm by age-2. Over 19 sampling years, a total of 136 Brown Trout ≥300 mm were captured in the Upper Rush Creek section, an average of 7.2 fish ≥300 mm per year (Table 14).

In the Bottomlands section of Rush Creek, the RSD-225 for 2018 equaled 36, a large drop from the record value of 65 for 2017 (Table 14). As in the Upper Rush section, the Bottomlands 2018 RSD-225 value was most likely influenced by greater numbers of fish, especially the numbers of fish smaller than 225 mm which comprised 64% of the trout ≥150 mm. The RSD-300 value was 6 in 2018. In 2018, we captured nine Brown Trout ≥300 mm in the Bottomlands section, most likely the result of a second year of more favorable summer water temperatures and higher survival rates (Table 14). Over the 11 sampling years, a total of 25 Brown Trout ≥300 mm were captured in the Bottomlands section, an average of 2.3 fish ≥300 mm per year (Table 14).

In the MGORD, the RSD-225 value decreased from 88 in 2017 to 70 in 2018; most likely due to larger numbers of trout <225 mm in length that were available for capture (Table 14). In 2018, the RSD-300 value was 20, the fourth consecutive year where the RSD-300 value has equaled or exceeded 20 in the MGORD (Table 14). The RSD-375 value decreased from 11 in 2017 to 5 in 2018; due to larger numbers of trout in the 150-374 mm size range (Table 14). The catch of Brown Trout ≥150 mm in the MGORD during the 2018 season was 326 fish, which included: 66 fish ≥300 mm in length and 15 fish ≥375 mm in length (Table 14). For sampling conducted between 2001 and 2012, the annual average catch of Brown Trout ≥300 mm equaled 180 fish/year; then for the past six sampling years the annual average catch of Brown Trout ≥300 mm equaled 41 fish/year (Table 14). This 77% decline in larger Brown Trout coincided with the five years of drier water-years and poor summer thermal regimes within the MGORD in 2012-2016; however numbers of larger (≥300 mm) Brown Trout experienced a modest increase in 2018 (Table 14).

RSD values in Lee Vining Creek were generated for the main channel combined with the side channel and for the main channel only (Table 15). The RSD-225 value for the main/side combined equaled 21 and main channel equaled 24 for 2018, these values represent slight decreases when compared to the 2017 values (Table 15). In 2018, two Brown Trout greater than 300 mm in length were captured in Lee Vining Creek main channel, which generated RSD-300 values of 4 for both the main/side combined and for the main channel only (Table 15).

Table 14. RSD values for Brown Trout in Rush Creek sections from 2000 to 2018.

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	**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	
Upper Rush	2013	336	288	45	3	0	14	1	
Upper Rush	2012	354	284	66	3	1	20	1	
Upper Rush	2011	498	381	110	6	1	23	1	
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	
Upper Rush	2007	282	210	61	9	2	26	4	1
Upper Rush	2006	233	154	69	10	0	34	4	
Upper Rush	2005	202	139	56	5	2	31	3	
Upper Rush	2004	179	112	64	2	1	37	2	
Upper Rush	2003	264	216	45	2	1	18	1	
Upper Rush	2002	220	181	35	1	2	18	2	1
Upper Rush	2001	223	190	27	6	0	15	3	
Upper Rush	2000	182	158	22	2	0	13	1	
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	
Bottomlands	2012	325	290	34	1	0	11	0	
Bottomlands	2011	267	218	46	3	0	18	1	
Bottomlands	2010	307	225	81	1	0	27	0	
Bottomlands	2009	379	321	56	1	1	15	1	
Bottomlands	2008	160	141	19	0	0	12	0	

Table 14 (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	**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 15. RSD values for Brown Trout in the Lee Vining Creek main channel + side channel sections from 2008-17. RSD values for Brown Trout in the main channel section from 2000-18.

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 & Side	**2018**	**57**	**45**	**10**	**2**	**0**	**21**	**4**
Main & Side	2017	30	23	6	1	0	23	3
Main & Side	2016	179	154	24	0	0	14	0
Main & Side	2015	227	206	21	0	0	9	0
Main & Side	2014	212	184	28	0	0	13	0
Main & Side	2013	327	309	17	1	0	6	0
Main & Side	2012	128	87	39	2	0	32	2
Main & Side	2011	78	46	26	5	1	41	1
Main & Side	2010	68	31	35	2	0	54	3
Main & Side	2009	192	159	32	1	0	17	1
Main & Side	2008	252	242	19	0	0	8	0
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

Table 15 (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
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
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

Termination Criteria (TC) Results based on 2014 – 2018 Data Sets

The Rush Creek sampling sections for years 2014 through 2018, failed to meet four of the five termination criteria for any of the three, three-year running averages. For the 2016-2018 three-year average, the Upper Rush section met three of the five termination criteria: condition factor, RSD-225 and RSD-300 (Table 16). This is the second consecutive annual TC analysis where three of the five termination criteria were met in the Upper Rush section (Table 16).

Table 16. Termination criteria analyses for the Upper Rush section of Rush Creek. Bold values indicate that an estimated value met a termination criterion.

Termination Criteria	2016 – 2018 Average	2015 – 2017 Average	2014 – 2016 Average
Biomass (≥175 kg/ha)	124	103	117
Density (≥3,000 trout/km)	2,488	1,699	2,603
Condition Factor (≥1.00)	**1.00**	**1.00**	0.99
RSD-225 (≥35)	**49**	**40**	18
RSD-300 (≥5)	**9**	**6**	2
Conclusion	**Met three of five TC**	**Met three of five TC**	**Met none of five TC**

For the 2016-2018 three-year average, the Bottomlands section met two of the five termination criteria: RSD-225 and RSD-300 (Table 17).

Table 17. Termination criteria analyses for the Bottomlands of Rush Creek. Bold values indicate that an estimated value met a termination criterion.

Termination Criteria	2016 – 2018 Average	2015 – 2017 Average	2014 – 2016 Average
Biomass (≥175 kg/ha)	62	48	48
Density (≥3,000 trout/km)	1,923	861	1,014
Condition Factor (≥1.00)	0.95	0.97	0.96
RSD-225 (≥35)	**41**	**36**	15
RSD-300 (≥5)	**5**	3	2
Conclusion	**Met two of five TC**	**Met one of five TC**	**Met none of five TC**

For the 2016-2018 three-year average, the MGORD met both the RSD-225 and RSD-375 termination criterion (Table 18).

Table 18. Termination criteria analyses for the MGORD section of Rush Creek. Bold values indicate that an estimated value met a termination criterion.

Termination Criteria	2016 – 2018 Average	2015 – 2017 Average	2014 – 2016 Average
RSD-225 (≥60)	**78**	**78**	**66**
RSD-300 (≥30)	23	24	18
RSD-375 (≥5)	**9**	**10**	**7**
Conclusion	**Met TC two of three RSD values**	**Met TC two of three RSD values**	**Met TC two of three RSD values**

For the 2016-2018 three-year average, the main and side channel sections of Lee Vining Creek together met one of the four termination criteria (Table 19).

Table 19. Termination criteria analyses for the Lee Vining Creek sample sections. Bold values indicate that an estimated value met a termination criterion.

Termination Criteria	**2016 - 2018 Average**	**2015 - 2017 Average**	**2014 - 2016 Average**
Biomass (≥150 kg/ha)	58	87	101
Density (≥1,400 trout/km)	559	877	1,371
Condition Factor (≥1.00)	**1.04**	**1.01**	0.97
RSD-225 (≥30)	19	16	12
Conclusion	**Met one of four TC**	**Met one of four TC**	**Met none of four TC**

PIT Tag Recaptures

PIT Tags Implanted between 2009 and 2018

Between 2009 and 2018, a total of 8,052 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 the Appendix B.

In 2018, a total of 993 trout received PIT tags and adipose fin clips in Rush and Lee Vining Creeks (Table 20). In addition, six recaptured adipose fin-clipped fish had shed their original tags and were re-tagged, thus a total of 999 PIT tags were implanted during the 2018 fisheries sampling (Table 20). Of the 999 trout tagged, 757 were age-0 Brown Trout and 153 were age-1 and older Brown Trout (Table 26). For Rainbow Trout, 81 age-0 fish and eight older fish were tagged (Table 20). The 148 age-1+ Brown Trout tagged in the MGORD section were no more than 250 mm in total length and were presumed to be age-1 fish (Table 20). Tagged and recaptured fish provided empirical information to estimate fish growth, tag retention and fish movements, and survivals.

Table 20. 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

In September of 2018, a total of 95 previously tagged trout (that retained their tags) were recaptured in Rush Creek (Appendix C). Thirty-nine of the recaptures occurred in the Upper Rush section (34 Brown Trout and five Rainbow Trout), followed by 20 recaptures in Walker Creek, 19 recaptures in the MGORD, and 17 recaptures in the Bottomlands section (Appendix C).

In September of 2018, a total of eight previously tagged Brown Trout (that retained their tags) were recaptured in the Lee Vining Creek main channel section (Appendix B). One previously tagged trout was recaptured in the Lee Vining Creek side channel section.

In the following text, growth between 2017 and 2018 will be referred to as 2018 growth rates. A 2018 trout refers to a fish recaptured in September of 2018. An age of a PIT tagged trout reflects the age during the sampling year. For instance, an age-1 trout in 2018 indicates that a trout had been tagged in October 2017 as age-0 and its length and weight were measured in September 2018 when it was recaptured. However, it should be noted that fish tagged in 2017 and recaptured in 2018 were at large for three weeks shorter than fish tagged and recaptured during most of the previous years (usually September to September).

Growth of Age-1 Brown Trout between 2017 and 2018

In 2018, a total of 47 known age-1 Brown Trout were recaptured that were tagged as age-0 fish in 2017, for an overall recapture rate of 15.7% (47/300 age-0 fish tagged in 2017). Of the 47 age-1 recaptures; 38 of these fish were from Rush Creek sections and nine fish were from the Lee Vining Creek main channel section. No age-0 fish were tagged in Walker Creek during the 2017 sampling. Thus, by creek, the age-1 recapture rates for 2018 were 29% in Lee Vining Creek (2.2% in 2017) and 14.1% in Rush Creek (19% in 2017 and 5% in 2016). These recapture rates suggest relatively high survival between age-0 and age-1 in Rush Creek, and a large improvement of survival rate in Lee Vining Creek in 2018 from the previous year.

In the Upper Rush section, 26 age-1 Brown Trout were recaptured in 2018 and the average growth rates of these trout were 83 mm and 56 g (Table 21). Compared to 2017 rates, the average growth rates of the 26 age-1 Brown Trout were lower by 49 mm and 73 g (Table 21). 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 21). The 2018 average growth rates were the lowest recorded for the past four years (Table 21).

In the Bottomlands section of Rush Creek, 10 age-1 Brown Trout were recaptured in 2018 and the average growth rates of these trout were 72 mm and 42 g (Table 21). Compared to 2017 rates, the growth rates of the 10 age-1 Brown Trout were lower by 46 mm and 54 g (Table 21). Growth rates of age-1 Brown Trout in the Bottomlands 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 21). The 2018 average growth rates were the lowest recorded for the past four years (Table 21).

In Walker Creek, no age-0 Brown Trout were tagged during the 2017 sampling, thus no tagged age-1 fish were available for recapture during the 2018 sampling.

In Lee Vining Creek, nine age-1 Brown Trout was recaptured in 2018 and the average growth rates of these trout were 103 mm and 77 g (Table 21). Compared to 2017 rates, the growth rates of the nine age-1 Brown Trout were lower by 7 mm and 15 g (Table 21). Growth rates of age-1 Brown Trout in Lee Vining Creek for the eight years of available data have averaged 84 mm in length and 50 g in weight (Table 21).

Growth of Age-2 Brown Trout between 2017 and 2018

In 2018, a total of 17 known age-2 Brown Trout were recaptured that were tagged as age-0 fish in 2016, for a recapture rate of 4.3% (17/394 age-0 fish tagged in 2016). All 17 of these fish were recaptured in Rush Creek and Walker Creek sections.

Within the Upper section of Rush Creek, six age-2 fish were recaptured in 2018 that had been tagged as age-0 fish in 2016 (Table 21). Between age-1 and age-2, the average growth rates of these six Brown Trout were 39 mm and 66 g (Table 21). The 2018 average growth rates were the lowest recorded for the past four years (Table 21).

In the Bottomlands section of Rush Creek, four age-2 fish were recaptured in 2018 that had been tagged as age-0 fish in 2016. Between age-1 and age-2, the average growth rates of these four Brown Trout were 39 mm and 55 g (Table 21). These four fish were the first recaptures of age-2 PIT tagged fish in the Bottomlands since September of 2015 (Table 21).

In Walker Creek, seven age-2 PIT tagged Brown Trout were recaptured in 2018 that had been tagged as age-0 fish in 2016; the average growth rates of these trout were 42 mm and 52 g (Table 21). Growth rates of age-2 Brown Trout in Walker Creek have averaged 40 mm in length and 38 g in weight for the eight years of available data (Table 21).

No age-2 PIT tagged trout were recaptured in the Vining Creek main channel during the September 2018 sampling. Only 42 age-0 Brown Trout were PIT tagged in Lee Vining Creek in 2016.

Growth of Age-3 Brown Trout between 2017 and 2018

In 2018, a total of six known age-3 Brown Trout were recaptured that were tagged as age-0 fish in 2015, for a recapture rate of 0.8% (6/738 age-0 fish tagged in 2015). Five of the six fish were recaptured in Walker Creek and the other fish was recaptured in the Upper Rush section. In Walker Creek, the average growth rates of age-3 fish between 2017 and 2018 averaged 25 mm and 37 g (Table 21). In Walker Creek, recaptures of PIT tagged age-3 fish have occurred in seven of the eight years that tagged age-3 fish were available for recapture, the most in any study section (Table 21).

The one PIT tagged age-3 Brown Trout recaptured in the Upper Rush section grew 11 mm in length and gained 40 g in weight between 2017 and 2018 (Table 21). This fish was the first age-3 fish recaptured in the Upper Rush section that provided a growth rate between age-2 and age-3 since 2014 (Table 21).

Table 21. Average growth (length and weight) of all Brown Trout recaptured from 2009 through 2018 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**
Upper Rush Creek	**Age 1**	89/51	81/50	83/48	72/33	67/35		90/55	105/77	132/129	83/56
	Age 2		58/70	54/73	43/42	41/42		64/69	99/176**	108/239	39/66
	Age 3				14/29		24/41				11/40*
	Age 4					12/-22					
	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
	Age 2		50/54	35/32	30/28	27/22	32/29*	62/62			39/55
	Age 3			13/14	17/16	11/9	35/31				
	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
	Age 2		66/95		77/110	33/34	35/29	47/40	47/49	77/128*	
	Age 3			34/92		23/48*	16/20*	27/32	42/75		
	Age 4				21/41*				25/47*		
	Age-5										
LV Main Channel Rainbow Trout	**Age 1**					78/47		80/35			
	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	
	Age 2		31/26	60/56	40/33	27/21	39/35		47/44	37/37	42/52
	Age 3			28/44	18/12	9/2	20/36	27/29		42/59*	25/37
	Age 4				7/2	2/-16*		28/45*			27/37*
	Age-5						0/-10*				

Growth of Age-4 Brown Trout between 2017 and 2018

In 2018, a single age-4 Brown Trout was recaptured in Walker Creek that was tagged as an age-0 fish in 2014, for a recapture rate of 2.4% (1/42 age-0 fish tagged in Walker Creek in 2014). This age-4 fish had a growth rate between 2017 and 2018 of 27 mm and 37 g (Table 21).

Growth of MGORD Brown Trout by size class between 2017 and 2018

Because the actual age at-time-of-tagging was unknown for most trout PIT tagged in the MGORD, determination of actual ages of recaptured trout was not possible. Thus, growth rate comparisons within the MGORD were based on size classes (Table 22). Due to the majority of the Brown Trout in the MGORD being larger sized, size classes were based on the RSD values for the MGORD. When evaluating growth rates by size classes, the size classes in Table 22 designate each fish's size class in 2017, not its size class at the time of recapture in 2018.

In 2018, a total of 17 PIT tagged Brown Trout were recaptured in the MGORD that were originally PIT tagged in the MGORD. Of these 17 recaptures, six fish had also been captured in 2017, thus one-year growth rates between 2017 and 2018 were calculated for these six fish (Table 22). In 2018, we also recaptured two PIT tagged Brown Trout in the MGORD that were originally tagged in the Upper Rush section.

One Brown Trout PIT tagged in the MGORD during the 2017 sampling within the <135 mm size class (presumed age-0) was recaptured within the MGORD in 2018 and this fish grew by 98 mm in length and 102 g in weight (Table 22). There was also an age-0 Brown Trout tagged in the Upper Rush section in 2017 that was recaptured in the MGORD in 2018. This trout's growth rates were 88 mm in length and 42 g in weight.

No tagged Brown Trout in the MGORD that were within the 135-225 mm size class and captured during the 2017 sampling were recaptured in 2018.

No tagged Brown Trout in the MGORD that were within the 226-300 mm size class and captured during the 2017 sampling were recaptured in 2018.

There were four PIT tagged Brown Trout captured in the MGORD during the 2017 sampling within the 301-375 mm size class (304, 338, 352, and 363 mm) that were recaptured in 2018. These four trout had average growth rates of 28 mm and 105 g between 2017 and 2018 (Table 22). Three of these fish had weight gains of 152, 154 and 154 g; and one fish lost 46 g.

There was one PIT tagged Brown Trout captured in the MGORD during the 2017 sampling within the >375 mm size class (501 mm) that was recaptured in 2018. This trout lost 10 mm in length and 370 g in weight (Table 22). This same fish had lost 148 g between 2016 and 2017.

Table 22. Average growth rates, length (mm) and weight (g), of all PIT tagged MGORD Brown Trout recaptured from 2009 through 2018 by size class. Note: *denotes only one fish recaptured.

Size Class (mm)	Average Annual Growth Length (mm)								
	2009-2010	2010-2011	2011-2012	2012-2013	2013-2014	2014-2015	2015-2016	2016-2017	2017-2018
0-134	121								98*
135-225	55	59	63			70*		90	
226-300	32	39	22	7		61	80	69	
301-375	20	17	9	12	30*	84*	74*	55*	28
>375	13	18	-1	10	17	69	34	24	-10
Size Class (mm)	**Average Annual Growth Weight (g)**								
	2009-2010	2010-2011	2011-2012	2012-2013	2013-2014	2014-2015	2015-2016	2016-2017	2017-2018
0-134	91								102*
135-225	85	90	78			155*		175	
226-300	53	81	34	2		203	184	172	
301-375	23	54	-5	49	178*	421*	365*	238*	105
>375	-10	134	-47	-2	283	718	208	94	-370

Growth of MGORD Brown Trout from non-consecutive years

Twelve of the 19 PIT tagged Brown Trout caught in the MGORD during the September 2018 sampling were last recaptured, measured and weighed in years prior to 2017; thus annual growth calculations were not possible. Seven of these 12 non-consecutive year recaptures were previously tagged or captured in 2016; thus two-year growth rates were calculated. On average, these fish grew by 99 mm in length and gained 255 g between 2016 and 2018. One of these Brown Trout had been initially tagged in Upper Rush in 2015 at age-0, was recaptured in Upper Rush in 2016 at age-1, and then was recaptured in the MGORD at age-3 in 2018.

Three of the non-consecutive year recaptures made in the MGORD during the 2018 sampling were Brown Trout tagged during the first two years of PIT tagging in 2009 and 2010 (Table 23). Fish #17016532 was initially tagged in 2009, escaped recapture for nine seasons, and was recaptured in 2018. During the nine years between captures, this fish gained an average of 51 g per year (Table 23). This Brown Trout was probably three or four years old when tagged in 2009, thus in 2018 it was most likely 12 or 13 years old.

Fish #20105641 was initially tagged in 2009, recaptured in 2010, was at large for seven years undetected, and was recaptured for a second time in 2018. During the seven years between 2010 and 2018, this fish gained an average of 169 g per year (Table 23). This Brown Trout was probably three years old when tagged in 2009, thus in 2018 it was most likely 12 years old.

Finally, fish #2345442 was initially tagged in 2010, recaptured in 2011 and 2012, was at large for five years undetected, and was recaptured for a third time in 2018. During the five years between 2012 and 2018, this fish gained an average of 215 g per year (Table 23). This Brown Trout was probably three or four years old when tagged in 2010, thus in 2018 it was most likely 11 or 12 years old.

Table 23. PIT tagged Brown Trout recaptured in the MGORD section of Rush Creek in September 2018, that were initially tagged in 2009 and 2010.

Last 8 Digits of PIT Tag #9851210-	Year of Capture	Total Length (mm)	Weight (g)	Difference in Length (mm)	Difference in Weight (g)
17016532	2009	426	924		
	2018	545	1,383	+119	+459
20105641	2009	338	364		
	2010	364	419	+26	+55
	2018	540	1,601	+176	+1,182
23454442	2010	384	575		
	2011	436	823	+52	+248
	2012	435	715	**-1**	**-108**
	2018	550	1,790	+115	+1,075

Movement of PIT Tagged Trout between Sections

From 2009 to 2018 just over 8,000 PIT tags were surgically implanted in Brown Trout and Rainbow Trout in the following annually sampled sections: Upper Rush, County Road, Bottomlands, MGORD, and Walker Creek. Most recaptures have occurred in the same sections where fish were originally tagged. Between 2010 and 2018, 38 Brown Trout were recaptured in stream reaches other than where they were initially tagged. The majority of movement between sections has occurred from the Upper Rush section upstream into the MGORD, and from the MGORD downstream into the Upper Rush section. We have also documented some limited movement between the Bottomlands and County Road sections. Up to 2013, no movement between other sections had been recorded. However in 2014, a large Brown Trout initially tagged in the MGORD was recaptured in the Bottomlands section.

In 2018, two Brown Trout recaptured in the MGORD had been tagged in the Upper Rush section. One of these fish moved upstream into the MGORD between age-0 and age-1 and the other fish moved between into the MGORD between age-1 and age-3.

PIT Tag Shed Rate of Trout Recaptured in 2018

In 2018, a total of 110 trout with adipose fin clips were recaptured and six of these fish failed to produce a PIT tag number when scanned with the tag reader (five shed tags were from Rush Creek recaptures and one was froma Lee Vining Creek recapture). Assuming that all these fish were previously PIT tagged, the 2018 calculated shed rate was 5.5% (6 shed tags/110 clipped fish recaptured). This rate was lower than the 2017 rate (9%), yet slightly higher than shed rates reported by other PIT tagging studies for juvenile trout: 3% for juvenile Brown Trout (Ombredane et al. 1998) and 3% for juvenile steelhead (Bateman and Gresswell 2006). 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).

Comparison of Length-at Age amongst Sample Sections

During 2018, four age-classes of PIT tagged Brown Trout were recaptured within four fisheries monitoring sections in Rush, Walker and Lee Vining creeks (Tables 24 and 25). Along with providing age-specific length information for each section, these data also allowed comparisons of length-at-age between sample sections and also between the years 2013-2018 (Tables 24 and 25). Again, the three weeks shy of a full year of growth between the October 2017 and September 2018 sampling events may have slightly influenced growth as measured in length.

In Upper Rush, the average length-at-age-1 in 2018 was 50 mm lower than the average length-at-age-1 in 2017 (Table 24). Similar to the three previous years, in 2018, age-1 Brown Trout in Upper Rush were larger than age-1 fish in the Bottomlands section (Table 24). In the Bottomlands section, the average length-at-age-1 in 2018 was 181 mm, 30 mm less than the 2017 average length-at-age-1 (Table 24).

In Upper Rush, seven PIT tagged age-2 Brown Trout were caught in 2018. The average length-at-age-2 of these seven Brown Trout was 274 mm, 39 mm less than the average length-at-age-2 in 2017 (Table 24). One of these seven fish was >300 mm at age-2. In the Bottomlands section, four age-2 Brown Trout were recaptured in 2018 and the average length-at-age-2 equaled 267 mm, 71 mm greater than the average length-at-age-2 of the three other years where data were available (Table 24).

In 2018, a single PIT tagged age-3 Brown Trout was recaptured in the Upper Rush sampling section and at 295 mm in length this fish was 57 mm greater than the average length-at-age-3 of the other two years where data were available (Table 24). In 2018, no age-4 or age-5 fish with PIT tags were captured in the Upper or Bottomlands sections of Rush Creek.

For Walker Creek in 2018, no age-1 Brown Trout were available for recapture (Table 24). In 2018, age-2 Brown Trout in Walker Creek were, on average, 10 mm longer than age-2 fish in 2017 (Table 24). In 2018, age-3 Brown Trout in Walker Creek were, on average, 10 mm shorter than age-3 fish in 2017 (Table 24). In 2018, one age-4 Brown Trout was recaptured in Walker Creek and this fish was 265 mm in length and was the largest confirmed age-4 fish we have recaptured in Walker Creek (Table 24).

In the Lee Vining Creek main channel the average length-at-age-1 for Brown Trout caught in 2018 was 183 mm (Table 25). In 2018, no previously tagged age-2 or age-3 Brown Trout were recaptured (Table 25).

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 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 appear to be a function of relatively low fish densities and mostly favorable summer water temperature conditions in

2018 (the second straight summer of favorable conditions). However, increasing densities of trout from 2017 to 2018 may have influenced the decline in growth rates observed between these two years.

Table 24. Size range of PIT tagged fish recaptured in 2013-2018 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	**2018** = 158-232 **2017** = 224-264 **2016** = 192-237 **2015** = 169-203	**2018** = 193 **2017** = 243 **2016** = 208 **2015** = 187
	Age-2	**2018** = 236-305 **2017** = 284-337 **2016** = 289* **2015** = 205-242	**2018** = 274 **2017** = 313 **2016** = 289* **2015** = 217
	Age-3	**2018** = 295 **2014** = 226-236 **2013** = 227-263	**2018** = 295 **2014** = 231 **2013** = 245
	Age-4	**2014** = 288 **2013** = 252-255	**2014** = 288 **2013** = 254
	Age-5	**2014** = 298	**2014** = 298
Bottomlands	Age-1	**2018** = 166-199 **2017** = 189-246 **2016** = 172-217 **2015** = 150-181	**2018** = 181 **2017** = 221 **2016** = 197 **2015** = 169
	Age-2	**2018** = 251-287 **2015** = 197-239 **2014** = 192 **2013** = 156-196	**2018** = 267 **2015** = 219 **2014** = 192 **2013** = 178
	Age-3	**2014** = 194 **2013** = 194-227	**2014** = 194 **2013** = 204
	Age-4	**2014** = 215-219	**2014** = 216
	Age-5	**2016** = 318	**2016** = 318
Walker Creek	Age-1	**2017** = 151-179 **2016** = 145-187 **2015** = 133-177	**2017** = 166 **2016** = 167 **2015** = 154
	Age-2	**2018** = 191-221 **2017** = 180-224 **2016** = 180-226 **2014** = 168-200 **2013** = 181-208	**2018** = 210 **2017** = 202 **2016** = 201 **2014** = 186 **2013** = 197
	Age-3	**2018** = 204-245 **2017** = 238 **2015** = 211-231 **2014** = 207-222 **2013** = 219-221	**2018** = 228 **2017** = 238 **2015** = 219 **2014** = 217 **2013** = 220
	Age-4	**2018** = 265 **2015** = 249 **2014** = 211 **2013** = 219	**2018** = 265 **2015** = 249 **2014** = 211 **2013** = 219
	Age-5	**2014** = 220	**2014** = 220

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

Table 25. Size range of PIT tagged fish recaptured in 2013-2018 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	**2018** = 170 -194 **2017** = 210 **2016** = 147-186 **2015** = 149-190	**2018** = 183 **2017** = 210 **2016** = 171 **2015** = 166
	Age-2	**2017** = 247 **2016** = 205-217 **2015** = 176-214 **2014** = 174-195 **2013** = 206-225	**2017** = 247 **2016** = 211 **2015** = 197 **2014** = 188 **2013** = 215
	Age-3	**2017** = 280-305 **2016** = 210-256 **2015** = 188-228 **2014** = 234-241 **2013** = 238-271	**2017** = 293 **2016** = 240 **2015** = 215 **2014** = 238 **2013** = 253
	Age-4	**2016** = 237	**2016** = 237
	Age-5	None captured in past five years	
Rainbow Trout in Lee Vining Main Channel	Age-1	**2016** = N/A **2015** = 140-177	**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 five years	
	Age-5	None captured in past five years	

Summer Water Temperature

Compared to the drought years of 2013-2016, the 2017 summer water temperatures in all sections of Rush Creek were a reprieve from four previous summers of stressful thermal conditions (Tables 26-29). Although RY 2018 was a normal year, GLR remained close to full due to rainfall and SCE's upstream maintenance operations, and this led to a second consecutive summer of mostly favorable water temperatures for Brown Trout growth and survival. In 2018, no Rush Creek monitoring locations had peak temperatures above 70^{o}F (Table 26).

Similar to the 2013-2017 annual reports, 2018 Rush Creek summer average daily water temperature data was classifiedbased 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 27). Development of these growth criteria were fully described in previous annual reports (Taylor 2013 and 2014). Using these growth prediction metrics, good potential growth days in 2018 varied from 23 to 58 days in Rush Creek out of the 92-day period from July 1 to September 30 (Table 27). The range of the number of good thermal days in 2018 was less than the 65 to 88 good thermal days recorded in 2017 (Table 27). For all Rush Creek monitoring locations, most of the remaining days in 2018 were classified as "fair" potential growth days; three days at Top of MGORD and two days at Bottom of MGORD were classified as poor growth days (Table 27).

As was done with the 2013 - 2017 data, the diurnal temperature fluctuations for July–September 2018 were characterized by the one-day maximum fluctuation that occurred each month and by monthly averages (Table 28). Also, for each temperature monitoring location, the highest average diurnal fluctuations over consecutive 21-day durations were determined (Table 28). The diurnal fluctuations throughout the summer of 2018 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 cooling of air temperatures (Table 28). Over the 21-day durations, these larger diurnal fluctuations were still below thresholds considered detrimental to trout growth during the summer of 2018 (Bell 2006).

The thermal window bounded by 66.2-71.6^{o}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 2018. 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^{o}F were summed by month and entire summer period (Table 29). The values from 2013 - 2017 were also included to better illustrate the variability that occurred at all the temperature monitoring locations (Table 29). The 2018 data show that all the temperature monitoring locations downstream of GLR experienced low numbers of hours bounded by the 66.2-71.6^{o}F thermal window (Table 29). In the MGORD, hourly water temperatures exceeded 66.2^{o}F less than 1% of the time and at the three downstream monitoring locations, hourly water temperatures of 66.2^{o}F were exceeded less than 10% of the time (Table 29). In 2018, the Rush Creek location Above Parker Creek had the most hours (182 hours) within the thermal window bounded by 66.2-71.6^{o}F (Table 29).

In 2018, 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 (Tables 26-29). 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 26-29).

Table 26. Summary of water temperature data during the summer of RY 2018 (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 2013-2017 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. Fluct.
Rush Ck. – At Damsite	2016 = 58.9 2017 = 58.1 **2018 = 59.7**	2016 = 58.3 2017 = 57.5 **2018 = 58.9**	2016 = 59.5 2017 = 58.7 **2018 = 60.4**	2016 = 0 2017 = 0 **2018 = 0**	2016 = 3.2 2017 = 2.1 **2018 = 2.4**	8/11/16 9/07/17 **8/22/18**
Rush Ck. – Top of MGORD	2013 = 63.1 2014 = 64.8 2015 = 64.4 2016 = 63.8 2017 = 57.0 **2018 = 60.7**	2013 = 62.7 2014 = 64.6 2015 = 64.1 2016 = 63.0 2017 = 56.5 **2018 = 59.6**	2013 = 63.7 2014 = 65.0 2015 = 64.8 2016 = 64.7 2017 = 58.1 **2018 = 61.9**	2013 = 0 2014 = 0 2015 = 0 2016 = 0 **2017 = 0** **2018 = 0**	2013 = 3.4 2014 = 3.9 2015 = 2.1 2016 = 6.5 2017 = 5.4 **2018 = 6.7**	7/09/13 8/13/14 7/03/15 7/07/16 9/07/17 **8/20/18**
Rush Ck. – Bottom MGORD	2013 = 63.2 2014 = 64.8 2015 = 64.4 2016 = 63.8 2017 = 57.1 **2018 = 61.0**	2013 = 60.9 2014 = 62.9 2015 = 62.3 2016 = 61.8 2017 = 56.5 **2018 =58.9**	2013 = 67.1 2014 = 68.5 2015 = 68.0 2016 = 66.9 2017 = 58.5 **2018 = 63.9**	2013 = 1 2014 = 20 2015 = 20 2016 = 1 **2017 = 0** **2018 = 0**	2013 = 9.0 2014 = 8.3 2015 = 8.4 2016 = 8.0 2017 = 6.4 **2018 = 8.7**	7/09/13 7/13/14 7/06/15 7/04/16 9/07/17 **7/05/18**
Rush Ck. – Old Highway 395 Bridge	2013 = 62.6 2014 = 64.0 2015 = N/A 2016 = 63.5 2017 = 59.0 **2018 = 60.9**	2013 = 58.8 2014 = 60.5 2015 = N/A 2016 = 60.1 2017 = 57.5 **2018 = 58.0**	2013 = 68.7 2014 = 69.8 2015 = N/A 2016 = 68.8 2017 = 61.0 **2018 = 65.3**	2013 = 40 2014 = 51 2015 = N/A 2016 = 47 2017 = 0 **2018 = 0**	2013 = 13.5 2014 = 13.3 2015 = N/A 2016 = 12.5 2017 = 7.6 **2018 = 10.9**	7/09/13 7/13/14 N/A 7/11/16 9/07/17 **7/10/18**
Rush Ck. – Above Parker	2016 = 63.2 2017 = 59.0 **2018 = 60.9**	2016 = 58.8 2017 = 57.2 **2018 = 57.2**	2016 = 69.4 2017 = 61.9 **2018 = 66.3**	2016 = 55 2017 = 0 **2018 = 0**	2016 = 13.7 2017 = 8.6 **2018 = 13.4**	7/11/16 9/08/17 **7/10/18**
Rush Ck. – below Narrows	2013 = 61.2 2014 = 63.2 2015 = 62.3 2016 = 61.7 2017 = 58.4 **2018 = 60.0**	2013 = 56.2 2014 = 57.1 2015 = 58.8 2016 = 56.9 2017 = 56.3 **2018 = 56.0**	2013 = 67.6 2014 = 69.4 2015 = 66.1 2016 = 68.3 2017 = 61.3 **2018 = 65.4**	2013 = 24 2014 = 46 2015 = 0 2016 = 34 2017 = 0 **2018 =0**	2013 = 16.3 2014 = 17.3 2015 = 11.5 2016 = 14.3 2017 = 8.2 **2018 = 12.4**	7/19/13 7/26/14 9/23/15 7/13/16 9/07/17 **7/10/18**
Rush Ck. – County Road	2013 = 61.4 2014 = 62.0 2015 = 62.1 2016 = 61.6 2017 = N/A **2018 = N/A**	2013 = 56.5 2014 = 56.7 2015 = 59.1 2016 = 56.0 2017 = N/A **2018 = N/A**	2013 = 66.6 2014 = 67.8 2015 = 65.5 2016 = 68.3 2017 = N/A **2018 = N/A**	2013 = 7 2014 = 24 2015 = 2 2016 = 32 2017 = N/A **2018 = N/A**	2013 = 14.7 2014 = 17.6 2015 = 9.2 2016 = 16.1 2017 = N/A **2018 = N/A**	8/02/13 7/26/14 7/28/15 7/11/16 N/A **N/A**

Table 27. Classification of 2013-2018 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%)**	2016 = 23 (25%) 2017 = 4 (4%) **2018 = 39 (42%)**	2016 = 0 2017 = 0 **2018 = 0**	2016 = 0 2017 = 0 **2018 = 0**
Rush Ck. – Top of MGORD	2013 = 14 (15%) 2014 = 5 (6%) 2015 = 7 (8%) 2016 = 10 (11%) 2017 = 66 (71%) **2018 = 47 (51%)**	2013 = 43 (47%) 2014 = 14 (15%) 2015 = 20 (22%) 2016 = 32 (35%) 2017 = 26 (29%) **2018 = 42 (46%)**	2013 = 17 (18%) 2014 = 25 (27%) 2015 = 5 (5%) 2016 = 17 (18%) 2017 = 0 **2018 = 3 (3%)**	2013 = 18 (20%) 2014 = 48 (52%) 2015 = 60 (65%) 2016 = 33 (36%) **2017 = 0** **2018 = 0**
Rush Ck. – Bottom MGORD	2013 = 11 (12%) 2014 = 6 (6%) 2015 = 8 (9%) 2016 = 9 (10%) 2017 = 67 (73%) **2018 = 48 (52%)**	2013 = 38 (41%) 2014 = 11 (12%) 2015 = 20 (22%) 2016 = 31 (34%) 2017 = 25 (27%) **2018 = 42 (46%)**	2013 = 20 (22%) 2014 = 21 (23%) 2015 = 5 (6%) 2016 = 16 (17%) 2017 = 0 **2018 = 2 (2%)**	2013 = 23 (25%) 2014 = 54 (59%) 2015 = 59 (64%) 2016 = 36 (39%) 2017 = 0 **2018 = 0**
Rush Ck. – Old Highway 395 Bridge	2013 = 14 (15%) 2014 = 7 (8%) 2015 = N/A 2016 = 16 (17%) 2017 = 75 (82%) **2018 = 36 (39%)**	2013 = 41 (45%) 2014 = 25 (27%) 2015 = N/A 2016 = 24 (26%) 2017 = 17 (18%) **2018 = 56 (61%)**	2013 = 33 (36%) 2014 = 27 (29%) 2015 = N/A 2016 = 19 (21%) 2017 = 0 **2018 = 0**	2013 = 4 (4%) 2014 = 33 (36%) 2015 = N/A 2016 = 33 (36%) 2017 = 0 **2018 = 0**
Rush Ck. – Above Parker Ck.	2016 = 17 (18%) 2017 = 65 (71%) **2018 = 28 (30%)**	2016 = 26 (28%) 2017 = 27 (29%) **2018 = 64 (70%)**	2016 = 24 (26%) 2017 = 0 **2018 = 0**	2016 = 25 (27%) 2017 = 0 **2018 = 0**
Rush Ck. – Below Narrows	2013 = 17 (18%) 2014 = 13 (14%) 2015 = 24 (26%) 2016 = 22 (24%) 2017 = 75 (82%) **2018 = 46 (50%)**	2013 = 69 (75%) 2014 = 58 (63%) 2015 = 44 (48%) 2016 = 52 (57%) 2017 = 17 (18%) **2018 = 46 (50%)**	2013 = 6 (7%) 2014 = 18 (20%) 2015 = 22 (24%) 2016 = 16 (17%) 2017 = 0 **2018 = 0**	2013 = 0 2014 = 3 (3%) 2015 =2 (2%) 2016 = 2 (2%) 2017 = 0 **2018 = 0**
Rush Ck. – County Road	2013 = 17 (18%) 2014 = 17 (18%) 2015 = 25 (27%) 2016 = 24 (26%) 2017 = N/A **2018 = N/A**	2013 = 64 (70%) 2014 = 59 (65%) 2015 = 39 (42%) 2016 = 50 (54%) 2017 = N/A **2018 = N/A**	2013 = 8 (9%) 2014 = 14 (15%) 2015 =23 (25%) 2016 = 13 (14%) 2017 = N/A **2018 = N/A**	2013 = 3 (3%) 2014 = 2 (2%) 2015 = 5 (6%) 2016 = 5 (6%) 2017 = N/A **2018 = N/A**

Table 28. Diurnal temperature fluctuations in Rush Creek for 2018: 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: 2017 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 = 2.0°F (2.0) Ave = 1.0°F (1.5)	Max = 2.4°F (1.8) Ave = 1.5°F (1.1)	Max = 2.4°F (2.5) Ave = 1.6°F (1.0)	1.9 °F (1.5) Aug-22 – Sept 11
Rush Ck. – Top of MGORD	Max = 4.0°F (2.7) Ave = 2.6°F (0.9)	Max = 6.7°F (4.5) Ave = 3.0°F (2.4)	Max = 1.8°F (5.4) Ave = 1.0°F (1.8)	3.4°F (2.1) Aug 10 – 30
Rush Ck. – Bottom MGORD	Max = 8.7°F (3.3) Ave = 6.1°F (2.4)	Max = 7.2°F (5.0) Ave = 4.7°F (2.6)	Max = 5.3°F (6.4) Ave = 4.0°F (2.7)	6.5°F (3.0) July 7 – 22
Rush Ck. – Old Highway 395 Bridge	Max = 10.9°F (3.9) Ave = 8.4°F (3.0)	Max = 8.6°F (5.3) Ave = 6.7°F (3.4)	Max = 8.6°F (7.6) Ave = 6.6°F (4.2)	8.8°F (4.4) July 1 - 21
Rush Ck. – Above Parker Ck.	Max = 13.4°F (5.1) Ave = 10.3°F (4.1)	Max = 10.6°F (5.6) Ave = 8.4°F (4.3)	Max = 10.9°F (8.6) Ave = 8.6°F (6.0)	10.8°F (6.4) July 1 - 21
Rush Ck. – below Narrows	Max = 12.4°F (5.2) Ave = 9.6°F (4.6)	Max = 11.0°F (5.9) Ave = 9.0°F (4.4)	Max = 12.1°F (8.2) Ave = 9.7°F (6.1)	10.2°F (6.4) Sept 8 - 28

Table 29. 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 - 2018.

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**	2016 = 0 hrs 2017 = 0 hrs **2018 = 0 hrs**	2016 = 0 hrs 2017 = 0 hrs **2018 = 0 hrs**	2016 = 0 hrs 2017 = 0 hrs **2018 = 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**	2013 = 4 hrs (0.5%) 2014 = 96 hrs (13%) 2015 = 205 hrs (28%) 2016 = 127 hrs (17%) 2017 = 0 hrs **2018 = 6 hrs**	2013 = 0 hrs 2014 = 0 hrs 2015 = 0 hrs 2016 = 0 hrs 2017 = 0 hrs **2018 = 0 hrs**	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%)**
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**	2013 = 229 hrs (31%) 2014 = 248 hrs (33%) 2015 =282 hrs (38%) 2016 = 268 hrs (36%) 2017 = 0 hrs **2018 = 1 hr (0.01%)**	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%)**	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%)**
Rush Ck. – Old 395 Bridge	2013 = 181 hrs (24%) 2014 = 287 hrs (39%) 2016 = 216 hrs (29%) 2017 = 0 hrs **2018 = 17 hrs (2%)**	2013 = 228 hrs (31%) 2014 = 248 hrs (33%) 2016 = 263 hrs (35%) 2017 = 0 hrs **2018 = 32 hrs (4%)**	2013 = 73 hrs (10%) 2014 = 117 hrs (16%) 2016 = 53 hrs (7%) 2017 = 3 hrs (0.4%) **2018 = 33 hrs (5%)**	2013 = 482 hrs (22%) 2014 = 639 hrs (29%) 2016 = 532 hrs (24%) 2017 = 3 hrs = (0.1%) **2018 = 82 hrs (4%)**
Rush Ck. – Above Parker Creek	2016 = 240 hrs (32%) 2017 = 0 hrs **2018 = 70 hrs (9%)**	2016 = 269 hrs (36%) 2017 = 0 hrs **2018 = 68 hrs (9%)**	2016 = 65 hrs (9%) 2017 = 14 hrs (2%) **2018 = 44 hrs (6%)**	2016 = 574 hrs (26%) 2017 = 14 hrs (0.6%) **2018 = 182 hrs (8%)**
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%)**	2013 = 192 hrs (26%) 2014 = 193 hrs (26%) 2015 = 189 hrs (25%) 2016 = 222 hrs (30%) 2017 = 0 hrs **2018 = 42 hrs (6%)**	2013 = 55 hrs (7%) 2014 = 105 hrs (15%) 2015 = 0 hrs (0%) 2016 = 49 hrs (7%) 2017 = 0 hrs **2018 = 36 hrs (5%)**	2013 = 405 hrs (18%) 2014 = 542 hrs (25%) 2015 = 318 hrs (14%) 2016 = 438 hrs (20%) 2017 = 0 hrs **2018 = 114 hrs (5%)**
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**	2013 = 172 hrs (23%) 2014 = 195 hrs (26%) 2015 = 119 hrs (16%) 2016 = 233 hrs (31%) 2017 = N/A **2018 = N/A**	2013 = 42 hrs (6%) 2014 = 79 hrs (11%) 2015 = 0 hrs (0%) 2016 = 42 hrs (6%) 2017 = N/A **2018 = N/A**	2013 = 411 hrs (19%) 2014 = 496 hrs (23%) 2015 = 293 hrs (13%) 2016 = 487 hrs (22%) 2017 = N/A **2018 = N/A**

Discussion

The 2018 sampling year documented fish populations responding favorably in Rush Creek to better water conditions related to a second continuous year of high storage levels in GLR. Moderate peak flows in conjunction with cooler summer water temperatures in 2018 appeared to facilitate a continued recovery of the trout populations from the previous five years of drought. Population estimates of age-0 and age-1 and older trout increased for a second consecutive year and apparent survival rates were also relatively high for a second straight year. Thus, this report's Discussion is focused on the trout populations' response to the Normal RY2018, mostly favorable summer water temperatures and the resulting increases in densities of fish.

2018 Summer Water Temperature and Trout Growth Rates

Before discussing the 2018 growth rates, the issue of the three weeks of less "time-at-large" between the October 2017 and September 2018 sampling should be acknowledged. Several researchers have documented that Brown Trout growth is greatest during the spring and fall months and lower in summer and winter months (Brown 1945; Swift 1961; Jensen and Berg 1995). In another study of Brown Trout residing in seven Spanish streams, growth varied during the year, peaking between March and September and then gradually decreasing during the fall to a winter minimum (Nicol and Almodovar 2004). Weight gains of age-2 Brown Trout in water temperatures between 52^{o}F and 59^{o}F that were fed to satiation had average growth rates of 2.8% per week during the fall months (Brown 1945). In regards to growth measured in length, age-1 Brown Trout grew 2.2-2.5% per week during the fall months (Swift 1961). Nicol and Almodovar (2004) recorded decreasing growth rates (in weight) during September and October, even though water temperatures were cooler and similar to temperatures documented during high growth periods during spring months, suggesting that decreasing photo-period influenced fall growth rates. Thus, the 49 weeks that previously PIT tagged fish were at large in Rush Creek between October 2017 and September 2018 may have resulted in about 8-10% reduced growth by weight and about 6-8% reduced growth by length. In addition, any comparisons between 2018 and 2017 growth rates needs to account for the extra 3.5 weeks of growth between 2016 and 2017 – thus, there's 7.5 weeks difference in time-at-large or potentially up to a 20-25% difference in weight due to this discrepancy.

The 2018 Brown Trout growth, as measured by weight gains of clipped or PIT tagged fish, between age-0 and age-1 in the Upper and Bottomlands sampling sections was average (Table 30). In the Upper Rush section, the weight gain of age-1 fish was 56 g, same as the 10-year average (Table 30). Similarly, in the Bottomlands section, the 2018 weight gain of age-1 Brown Trout was 41 g, within one gram of the 10-year average (Table 30). The Upper Rush section's age-2 recaptures gained an average of 66 g between 2017 and 2018; a growth rate 32% lower than the average growth rate (97 g) for the eight years of available tag return data (Table 30). Even accounting for the 49 weeks (versus 52 weeks) that tagged fish were at large, the average to below average growth rates documented in 2018 suggest that a combination of increasing densities of fish was an important factor, even with mostly favorable summer water temperatures. Even in the direct comparison between 2017 and 2018, and accounting for the 20-25% potential discrepancy due to varying time-at-large, Brown Trout weight gains were considerably lower in 2018; 56% to 72% less than in 2017 (Table 30).

Table 30. Annual growth rate (g) for PIT tagged or fin-clipped age-0 to age-1 and 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-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
	10-yr Average	*56.4*	*41.0*	
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
	8-yr Average	*97.1*	*40.3*	

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 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. 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, 13 years (2006-2018) 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 strongly suggest that density-dependent growth of age-0 fish does occur in the Upper Rush section (Figure 21). In the past two years, average weights of age-0 Brown Trout sampled from the Upper Rush section dropped from 12.3 g in 2017 to 8.6 g in 2018. Similarly, in the Bottomlands section average weights of age-0 Brown Trout dropped from 13.7 g in 2017 to 6.8 g in 2018; a 50% decrease in average weights when densities of age-0 fish increased tenfold.

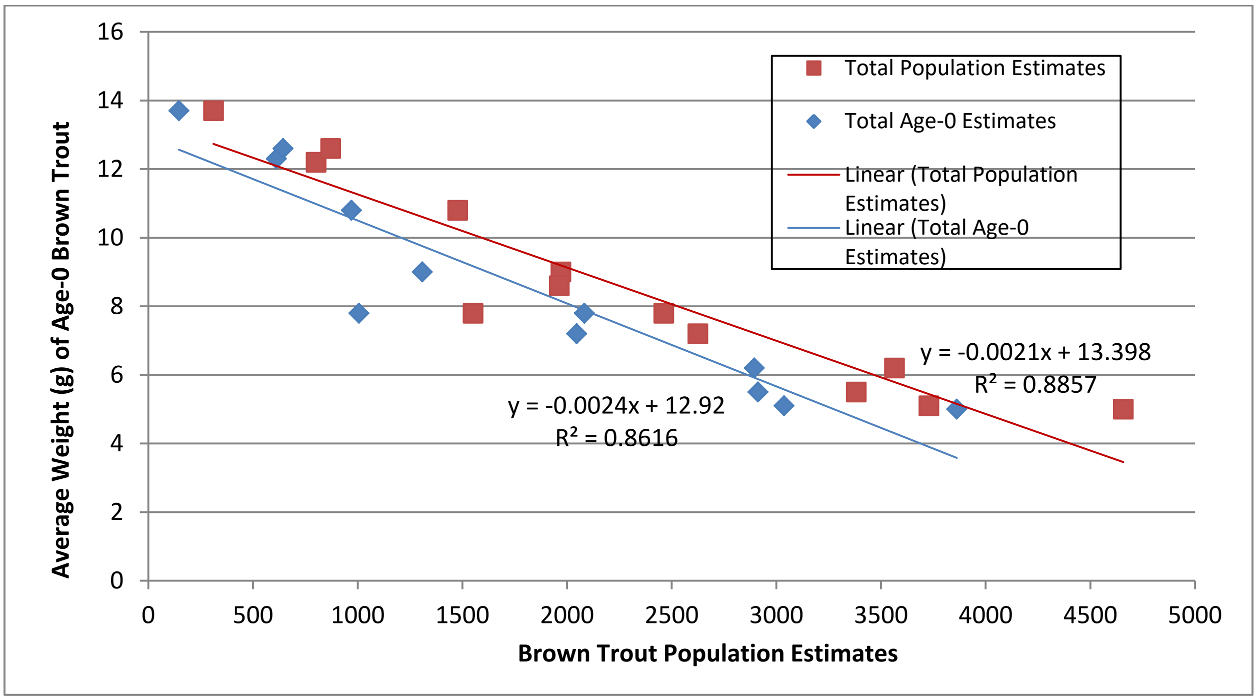


Figure 21. Relationship between average weights of age-0 brown trout and population estimates (age-0 and all trout) in the Upper Rush sampling section, 2006-2018.

Apparent Survival Rates

Apparent survival rates of age-1 Brown Trout were calculated with the following equation: [# age-1 recaps in 2018/capture probability of age-1 fish] ÷ [# age-0 tagged in 2017 - # shed tags]. For mark-recapture sections, capture probabilities were derived from the recapture run data: # of recaptures/# of captures. Compared to the 2017 survival rates, the 2018 apparent survival rates were about 50% lower in Upper Rush Creek and slightly lower in the Bottomlands section

(Table 31). In contrast, Lee Vining Creek's age-1 Brown Trout apparent survival rates increased dramatically from 4.8% in 2017 to 70.6% in 2018 (Table 31).

The 2016 and 2017 apparent survival rates were provided to show that rates flipped for all sections between the 2016 Dry RY and the 2017 Extreme Wet RY. During 2016 when flows were low and warm in Rush Creek, survival was low; and when flows were low and cool in Walker and Lee Vining creeks, survival was high. In sharp contrast, during 2017 when flows in all creeks were at or near record highs and water temperatures were cool, survival rates were high in Rush Creek and low in Walker and Lee Vining creeks. Then during the 2018 Normal RY (with a high storage level in GLR), the apparent survival rates were above 50% in the three sections where data were available. Walker Creek was excluded from the 2018 apparent survival analysis because no age-0 fish were PIT tagged in 2017 (Table 31).

Table 31. Apparent survival rates of age-1 Brown Trout in Rush, Walker and Lee Vining creeks in 2018. The 2016 values are in parentheses for comparisons.

Creek and Section	Capture Probability	No. Age-1 Recaps in 2018	No. Age-0 Tagged in 2017	No. Shed Tags	Apparent Survival Rate
Rush – Upper	0.27	26	192	0	2016 = 22.7% 2017 = 106% **2018 = 50.2%**
Rush - Bottomlands	0.44	10	34	0	2016 = 9.7% 2017 = 72.3% **2018 = 66.8%**
Walker Creek	N/A	N/A	N/A	N/A	2016 = 37.8% 2017 = 7.0%
Lee Vining Creek	0.32	9	31	0	2016 = 46.3% 2017 = 4.8% **2018 = 70.6%**

Age-0 Recruitment and Age-Class Structure in Rush and Lee Vining Creeks

The availability and location of spawning habitat in Rush Creek was a concern during the development of Decision 1631 and subsequent SWRCB Orders #98-05 and #98-07. The Mono Basin EIR noted that 55 redds were found between 1985 and 1989, primarily in the uppermost 0.85 miles of Rush Creek below GLR dam (page 3D-19). Section 5.4.2 of Decision 1631 (titled Flows for Providing Fishery Habitat) stated, "There is general agreement that adult habitat and spawning habitat in Rush Creek are limited." Much of the early instream flow recommendations centered on the stability of introduced spawning substrate. In contrast, our experience since 1999 after the fisheries sampling methods were established, was that annual recruitment of age-0 Brown Trout in the Rush Creek sections was variable, yet sufficient enough to translate into ample numbers of age-1 and older fish in subsequent years. Previous annual fisheries monitoring reports have shown that wide ranges in the numbers of age-0 Brown Trout produced in 2000-2004 eventually translated into similar numbers of age -1 and older fish (Hunter et al. 2004 - 2007). We also stated in the Synthesis Report that "In Rush Creek, ample recruitment of age-0 Brown Trout has occurred the past ten years" (McB&T and RTA 2010). During the five below-normal RY types, the numbers of age-0 Brown Trout declined in both annually sampled sections of Rush Creek. In the Upper Rush section, the population estimate of age-0 Brown Trout declined by 95% between 2012 and 2016. Age-0 Brown Trout in the Bottomlands section experienced an 83% decline in population estimates between 2012 and 2016. Between 2012 and 2015, the decreased fish numbers in Rush Creek were fairly steady and progressive. However, the paucity of age-0 Brown Trout in 2016 (only 46 were captured) suggested that the trout population had crashed after five years of drought, probably due to extremely low numbers of adult spawners and possible reduced egg viability due to warm water induced stress.

The 2017 population estimates of age-0 in the Rush Creek sections confirmed a 2% increase from the 2016 estimate in the Bottomlands section and a tripling of the age-0 estimate in the Upper Rush section. In last year's annual report we speculated that the continued recovery of Brown Trout population in Rush Creek would be contingent on continued favorable summer water temperatures in 2018 and that age-0 recruitment might be hindered by limited numbers of mature spawners as a lingering effect of the five years of drought and stressful summer water temperatures. Also, when the fisheries report was submitted in April of 2018, the runoff forecast was for a normal year and potentially lower storage levels in GLR, however the late rains and SCE's operations resulted in a nearly full GLR. As presented in this year's report, recruitment of age-0 Brown Trout in 2018 exceeded our supposition that availability of adult spawners may limit recruitment. Age-0 densities increased by 134% in the Upper Rush section and by >1,000% in the Bottomlands section.

Limited information was found concerning post-drought responses by stream dwelling trout populations. However, an assessment of naturally reproducing Rainbow Trout populations in Colorado on National Forest lands concluded that shortly after an extended period of drought (2000-2004), Rainbow Trout numbers were at stable, or increased, levels due to the fish's wide distribution across multiple watersheds (Adams et al. 2008). The state of Connecticut conducted pre and post drought fisheries sampling in 23 streams important to naturally-reproducing Brown Trout and concluded that recovery from drought conditions was mixed; in six streams, populations still declined and remained low, whereas populations in 17 streams improved, albeit recovery was "slow, not universal, and contingent on several consecutive years of favorable conditions" (Humphreys 2015). Thus, a continued rebound of trout populations in Mono Basin streams is dependent on this winter's snowpack and GLR level going into the 2019 RY.

As of mid-February 2019, the eastern Sierras had experienced an above normal winter and the snow pack near Mammoth was approximately 140% of normal. LADWP's 2019 Eastern Sierra forecast made on March 1st for the Mono Basin was 132% to 157% of normal. If RY 2019 remains above average (by April 1st) then Rush Creek below GLR will most likely experience a third consecutive summer of favorable water temperature conditions, which will translate into continued increasing population numbers, high recruitment of age-0 fish, and relatively high survival rates. We suspect that growth rates may continue to decrease or remain "average" as trout densities continue to increase. However, increased survival rates and favorable thermal conditions may allow the continued increase in numbers of larger Brown Trout in both the MGORD and the Upper Rush sections.

Methods Evaluation

In 2018, mark-recapture and depletion estimates were again used to produce population estimates on Rush, Lee Vining and Walker Creeks. As in past years, we started off cleaning the block fences twice a day, but several periods of windy conditions and falling leaves resulted in block fence failures. After the upstream fences at Upper Rush and the Lee Vining Creek main channel failed several times each, we implemented a more rigorous fence cleaning schedule. Fence failure has become more prevelant over the past seven or eight sampling years and for future sampling, it's recommended that LADWP dedicate a person whose primary job is to clean fences. When the annual fisheries sampling was conducted by the crew of consultants, a dedicated fence cleaner was valuable in keeping the fences up for the seven-day duration between mark and recapture electrofishing runs.

As in previous years, small variations in wetted channel widths were measured, which resulted in changes to sample section areas. Also, we moved the location of several block fences due to changes in channel depths and increased velocities. As previously mentioned, several abandoned meanders were reconnected in Walker Creek, resulting in a longer channel length in 2018, yet average wetted width was narrower, resulting in a smaller wetted area than the previous year. Thus, it is recommended that channel lengths and widths are re-measured annually.

The PIT tagging program was continued during the September 2018 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 towards its 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 **40 cfs** (± 5 cfs) and flow in Lee Vining Creek should not exceed **30 cfs** (± 5 cfs) during the annual sampling period. Allowances for flow variances to allow for safe wading conditions and effective sampling were included in the new Terms of Settlement.

References Cited

Adams, P., C. James and C. Speas. 2008. Rainbow Trout (*Onchorhynchus mykiss*): species conservation and assessment. Prepared for the Grand Mesa, Uncompahgre and Gunnison National Forests. 25 p.

Baerum, K.M., T. Haugen, P.M. Kiffney, E.M. Olsen, and L.A. Vollsetad. 2013. Interacting effects of temperature and density on individual growth performance in a wild population of Brown Trout. Journal of Freshwater Biology 58(7): 1329-1339.

Bateman, D.S. and R.E. Gresswell. 2006. Survival and growth of age-0 steelhead after surgical implantation of 23-mm passive integrated transponders. North American Journal of Fisheries Management 26: 545-550.

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

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

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

Bohlin, T., L.F. Sundström, J.I. Johnsson, J. Höjesjö and J. Petterson. 2002. Density-dependent growth in Brown Trout: effects of introducing wild and hatchery trout. J. of Animal Biology 71: 683-692.

Brown, M.E. 1945. The growth of Brown Trout (*Salmo trutta*): effects of temperatures on the growth of two-year old trout. Zoology Department, Cambridge. 11 p.

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

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

Grant, J.W.A. and I. Imre. 2005. Patterns of density-dependent growth in juvenile stream-dwelling salmonids. Journal of Fish Biology 67:100-110.

Humphreys, M. 2015. Wild trout research and management. Connecticut Inland Fisheries, Annual Performance Report. 12 p.

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

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

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

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

Jensen, J.W. and T. Berg. 1995. The annual growth rate cycle of Brown Trout. The Progressive Fish Culturist, 57(3): 231-237.

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

Ombredane, D., J. Bagliniere, and F. Marchand. 1998. The effects of passive integrated transponder tags on survival and growth of juvenile Brown Trout (*Salmo trutta* L.) and their use for studying movement in a small river. Hydrobiologica 371: 99-106.

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

Swift, D.R. 1961. The annual growth-rate cycle in Brown Trout (*Salmon trutta*) and its cause. Journal of Experimental Biology, 38: 595-604.

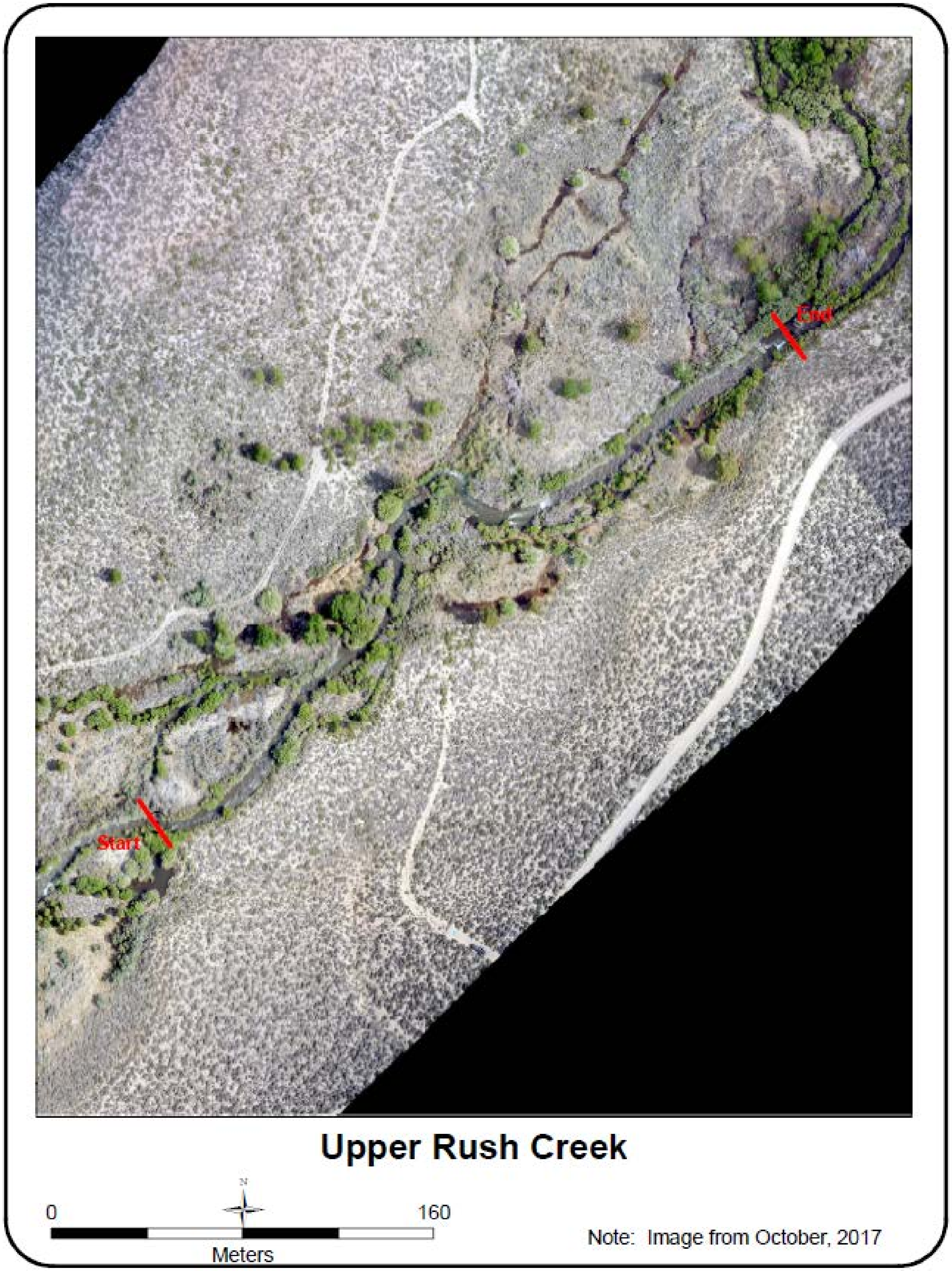
Taylor, R., and K. Knudson. 2012. Fisheries Monitoring Report for Rush, Lee Vining, Parker and Walker Creeks 2011. Los Angeles Department of Water and Power. 90 p.

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

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

Appendices for the 2018 Mono Basin Annual Fisheries Report

Appendix A: Aerial Photographs of Annual Sample Sites on Rush, Walker and Lee Vining Creeks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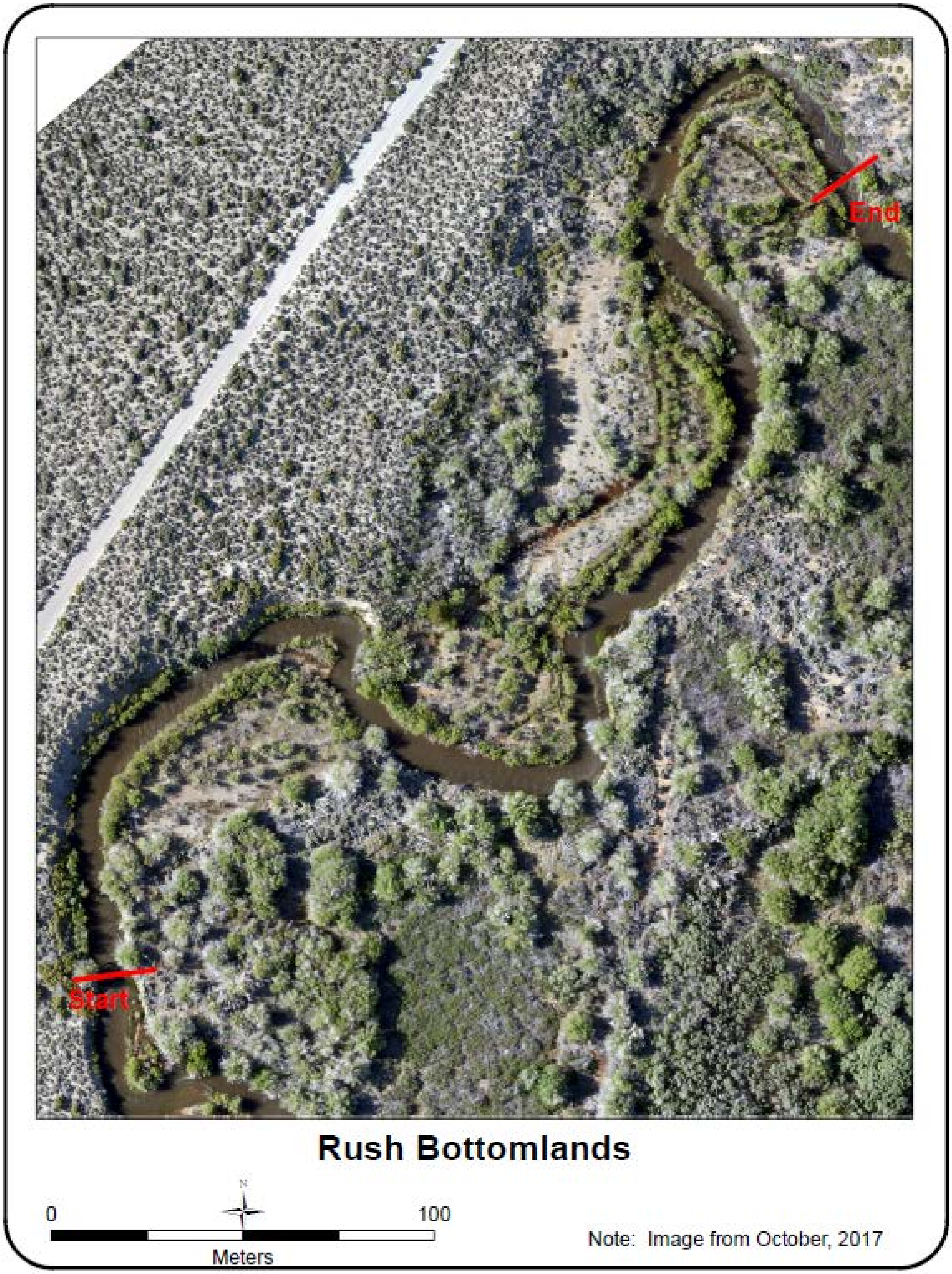
Meters
Note: Image from October, 2017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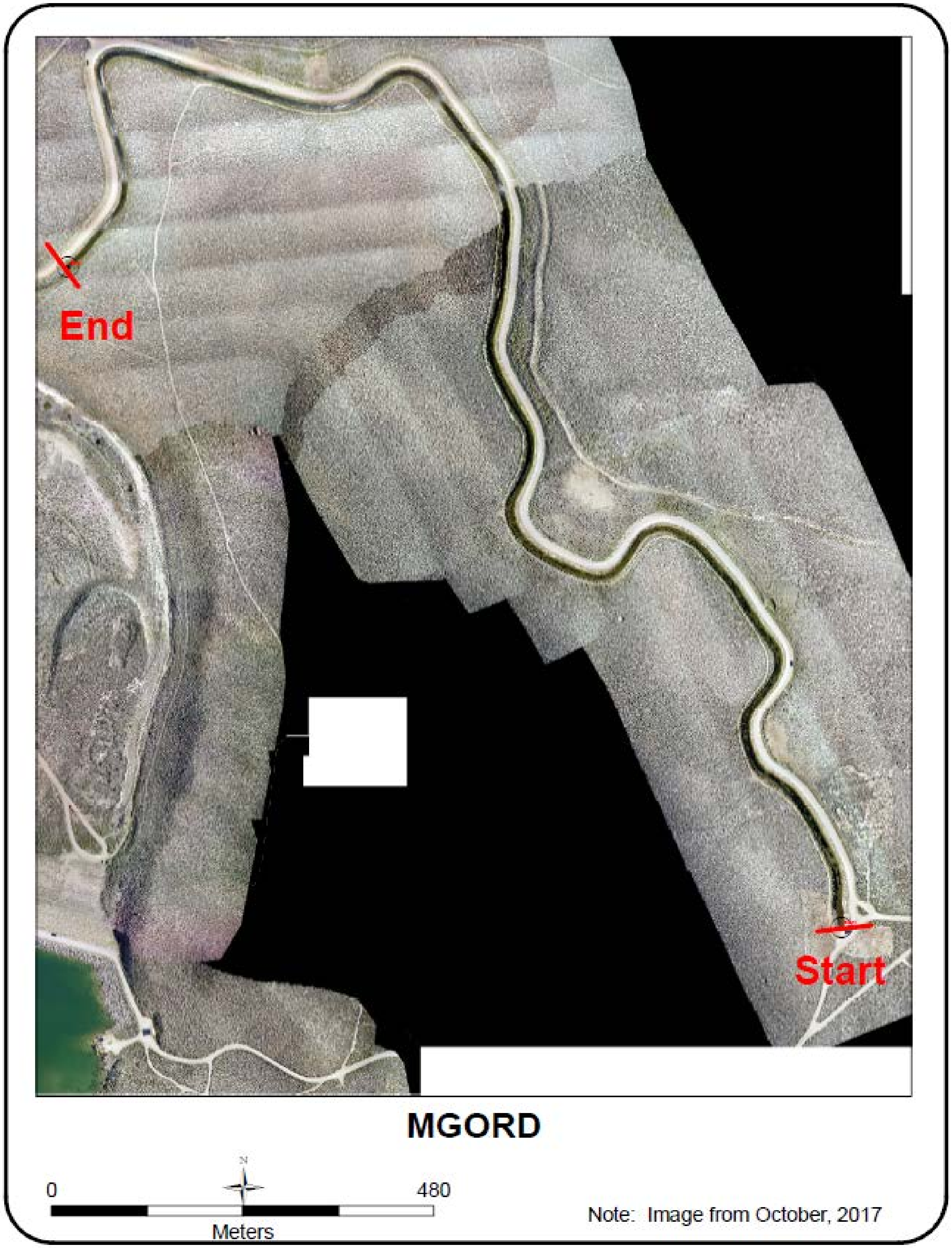
Meters
Note: Image from October, 2017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MGORD
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Meters
Note: Image from October, 2017


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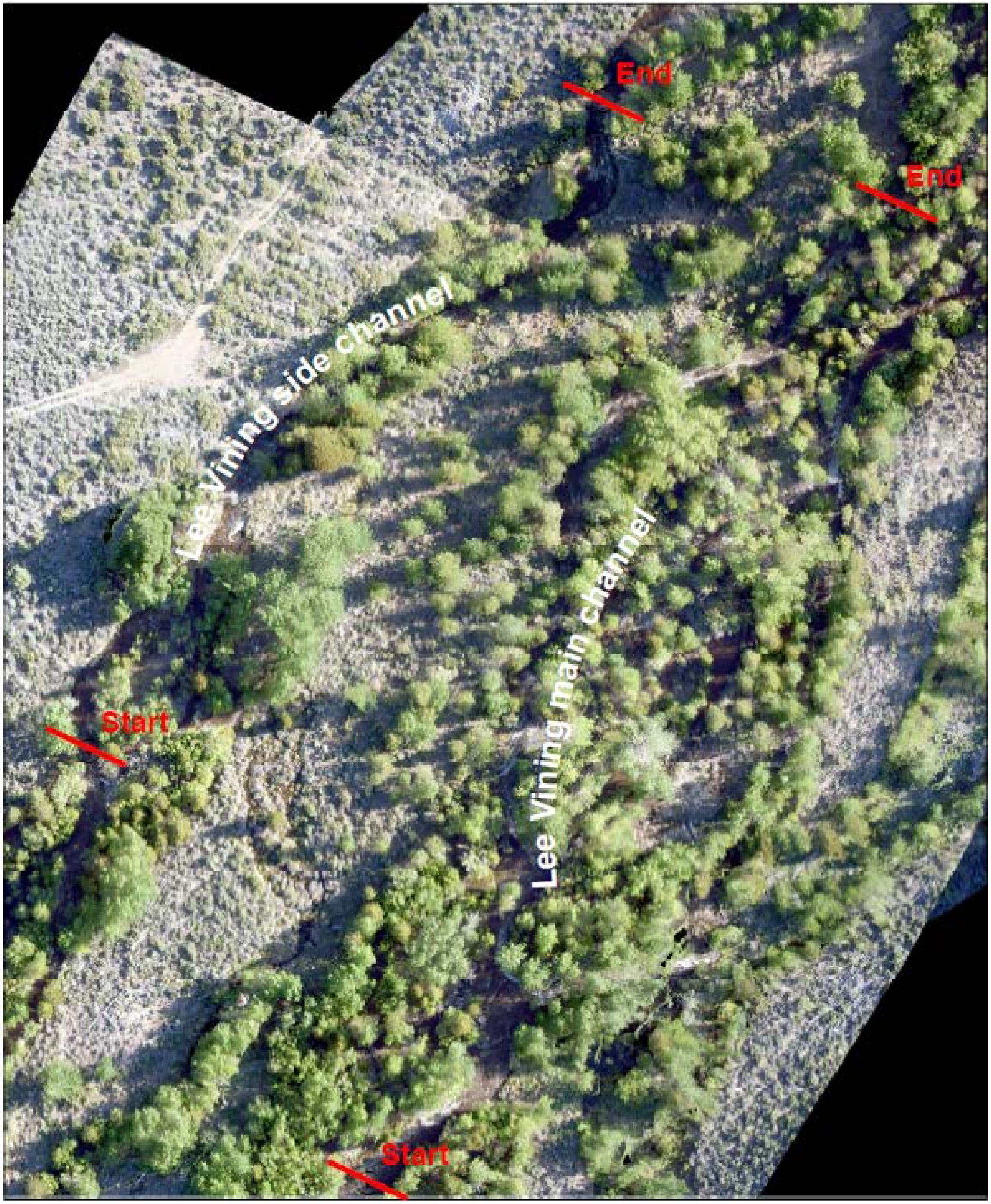
Meters
N
Note: Image from October, 2017



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 2017

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

Appendix C: Table of PIT-tagged Fish Recaptured during September 2018 Sampling

Appendix C. PIT tagged trout recaptured in Rush and Lee Vining Creeks, September 2018.

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2018 Recapture	Location of Initial Capture and Tagging
9/17/2018	BNT	295	243	989001004580732	UpperRush	Upper Rush
9/17/2018	BNT	256	156	989001006111210	UpperRush	Upper Rush
9/17/2018	BNT	270	211	989001006111242	UpperRush	Upper Rush
9/17/2018	BNT	230	118	989001006111437	UpperRush	Upper Rush
9/17/2018	BNT	169	50	989001006111450	UpperRush	Upper Rush
9/17/2018	BNT	188	65	989001006111480	UpperRush	Upper Rush
9/17/2018	BNT	183	56	989001006111490	UpperRush	Upper Rush
9/17/2018	BNT	190	68	989001006111504	UpperRush	Upper Rush
9/17/2018	BNT	236	128	989001006111512	UpperRush	Upper Rush
9/17/2018	BNT	168	51	989001006111561	UpperRush	Upper Rush
9/17/2018	BNT	185	61	989001006111562	UpperRush	Upper Rush
9/17/2018	BNT	191	63	989001006111572	UpperRush	Upper Rush
9/17/2018	BNT	210	79	989001006111578	UpperRush	Upper Rush
9/17/2018	BNT	214	92	989001006111596	UpperRush	Upper Rush
9/17/2018	BNT	158	37	989001006111626	UpperRush	Upper Rush
9/17/2018	BNT	195	74	989001006111642	UpperRush	Upper Rush
9/17/2018	BNT	203	78	989001006111691	UpperRush	Upper Rush
9/17/2018	BNT	165	44	989001006111698	UpperRush	Upper Rush
9/24/2018	BNT	270	187	989001006111204	UpperRush	Upper Rush
9/24/2018	BNT	305	280	989001006111232	UpperRush	Upper Rush
9/24/2018	BNT	285	187	989001006111244	UpperRush	Upper Rush
9/24/2018	BNT	293	250	989001006111251	UpperRush	Upper Rush
9/24/2018	BNT	169	47	989001006111497	UpperRush	Upper Rush
9/24/2018	BNT	205	87	989001006111503	UpperRush	Upper Rush
9/24/2018	BNT	182	61	989001006111511	UpperRush	Upper Rush
9/24/2018	BNT	220	77	989001006111527	UpperRush	Upper Rush
9/24/2018	BNT	208	84	989001006111534	UpperRush	Upper Rush
9/24/2018	BNT	190	65	989001006111542	UpperRush	Upper Rush
9/24/2018	BNT	232	121	989001006111555	UpperRush	Upper Rush
9/24/2018	BNT	188	63	989001006111592	UpperRush	Upper Rush
9/24/2018	BNT	184	58	989001006111595	UpperRush	Upper Rush
9/24/2018	BNT	194	69	989001006111600	UpperRush	Upper Rush
9/24/2018	BNT	199	72	989001006111629	UpperRush	Upper Rush
9/24/2018	BNT	189	64	989001006111687	UpperRush	Upper Rush
9/17/2018	**RBT**	201	80	989001006111537	UpperRush	Upper Rush
9/17/2018	**RBT**	235	135	989001006111565	UpperRush	Upper Rush
9/17/2018	**RBT**	192	67	989001006111664	UpperRush	Upper Rush

Appendix C. PIT tagged trout recaptured in Rush and Lee Vining Creeks, September 2018.

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2018 Recapture	Location of Initial Capture and Tagging
9/24/2018	**RBT**	190	71	989001006111637	UpperRush	Upper Rush
9/24/2018	**RBT**	183	56	989001006111646	UpperRush	Upper Rush
9/18/2018	BNT	278	185	989001004581290	Bottomlands	Bottomlands
9/18/2018	BNT	274	193	989001004581391	Bottomlands	Bottomlands
9/18/2018	BNT	251	145	989001006111224	Bottomlands	Bottomlands
9/18/2018	BNT	261	137	989001006111226	Bottomlands	Bottomlands
9/18/2018	BNT	179	51	989001006111418	Bottomlands	Bottomlands
9/18/2018	BNT	166	43	989001006111421	Bottomlands	Bottomlands
9/18/2018	BNT	193	72	989001006111436	Bottomlands	Bottomlands
9/18/2018	BNT	199	71	989001006111443	Bottomlands	Bottomlands
9/18/2018	BNT	186	59	989001006111452	Bottomlands	Bottomlands
9/18/2018	BNT	176	46	989001006111465	Bottomlands	Bottomlands
9/18/2018	BNT	171	43	989001006111476	Bottomlands	Bottomlands
9/25/2018	BNT	260	166	989001004581361	Bottomlands	Bottomlands
9/25/2018	BNT	287	222	989001006111291	Bottomlands	Bottomlands
9/25/2018	BNT	255	155	989001006111292	Bottomlands	Bottomlands
9/25/2018	BNT	177	55	989001006111422	Bottomlands	Bottomlands
9/25/2018	BNT	184	57	989001006111457	Bottomlands	Bottomlands
9/25/2018	BNT	175	47	989001006111469	Bottomlands	Bottomlands
9/20/2018	BNT	540	1601	985121020105641	MGORD	MGORD
9/20/2018	BNT	550	1790	985121023454442	MGORD	MGORD
9/20/2018	BNT	475	1321	989001001239659	MGORD	MGORD
9/20/2018	BNT	331	320	989001004580855	MGORD	**Upper Rush**
9/20/2018	BNT	337	421	989001004581259	MGORD	MGORD
9/20/2018	BNT	323	326	989001004581335	MGORD	MGORD
9/20/2018	BNT	345	413	989001006110908	MGORD	MGORD
9/20/2018	BNT	327	362	989001006110981	MGORD	MGORD
9/20/2018	BNT	387	540	989001006111290	MGORD	MGORD
9/20/2018	BNT	372	523	989001006111324	MGORD	MGORD
9/20/2018	BNT	491	1084	989001006111367	MGORD	MGORD
9/20/2018	BNT	345	434	989001006111378	MGORD	MGORD
9/20/2018	BNT	365	455	989001006111387	MGORD	MGORD
9/20/2018	BNT	227	122	989001006111645	MGORD	MGORD
9/27/2018	BNT	545	1383	985121017016532	MGORD	MGORD
9/27/2018	BNT	502	1395	989001004580768	MGORD	MGORD
9/27/2018	BNT	447	965	989001004581314	MGORD	MGORD
9/27/2018	BNT	365	456	989001006111401	MGORD	MGORD
9/27/2018	BNT	180	52	989001006111588	MGORD	**Upper Rush**

Appendix C. PIT tagged trout recaptured in Rush and Lee Vining Creeks, September 2018.

Date of Recapture	Species	Length (mm)	Weight (g)	PIT Tag Number	Location of 2018 Recapture	Location of Initial Capture and Tagging
9/21/2018	BNT	265	170	989001001953504	Walker Ck	Walker Creek
9/21/2018	BNT	245	141	989001004580857	Walker Ck	Walker Creek
9/21/2018	BNT	224	107	989001004580876	Walker Ck	Walker Creek
9/21/2018	BNT	246	155	989001004580887	Walker Ck	Walker Creek
9/21/2018	BNT	240	144	989001004580913	Walker Ck	Walker Creek
9/21/2018	BNT	204	88	989001004580937	Walker Ck	Walker Creek
9/21/2018	BNT	210	90	989001006111006	Walker Ck	Walker Creek
9/21/2018	BNT	224	98	989001006111007	Walker Ck	Walker Creek
9/21/2018	BNT	212	96	989001006111014	Walker Ck	Walker Creek
9/21/2018	BNT	199	88	989001006111045	Walker Ck	Walker Creek
9/21/2018	BNT	210	96	989001006111055	Walker Ck	Walker Creek
9/21/2018	BNT	187	59	989001006111057	Walker Ck	Walker Creek
9/21/2018	BNT	221	110	989001006111064	Walker Ck	Walker Creek
9/21/2018	BNT	211	106	989001006111072	Walker Ck	Walker Creek
9/21/2018	BNT	202	90	989001006111094	Walker Ck	Walker Creek
9/21/2018	BNT	222	124	989001006111138	Walker Ck	Walker Creek
9/21/2018	BNT	191	73	989001006111179	Walker Ck	Walker Creek
9/21/2018	BNT	195	80	989001006111191	Walker Ck	Walker Creek
9/21/2018	BNT	188	62	989001006111194	Walker Ck	Walker Creek
9/21/2018	BNT	216	95	989001006111199	Walker Ck	Walker Creek
9/19/2018	BNT	178	58	989001006111313	LV Main	LV Main
9/19/2018	BNT	180	58	989001006111326	LV Main	LV Main
9/19/2018	BNT	194	66	989001006111328	LV Main	LV Main
9/19/2018	BNT	184	61	989001006111334	LV Main	LV Main
9/19/2018	BNT	170	50	989001006111358	LV Main	LV Main
9/19/2018	BNT	194	74	989001006111604	LV Main	LV Main
9/19/2018	BNT	183	60	989001006111688	LV Main	LV Main
9/26/2018	BNT	184	63	989001006111699	LV Main	LV Main
9/21/2018	BNT	178	55	989001006111410	LV Side	LV Side

Section 4

RY 2018 Mono Basin Stream Monitoring Report

Mason London (left) and Emily Cooper (right) locating the riffle crest thalweg upstream of a newly formed point bar in Rush Creek, Mono Lake Basin, CA (Photo: 4 Oct., 2017, 11:35 AM).

RY2018 Mono Basin Stream Monitoring Report

Emily J. Cooper, HSU River Institute Research Scientist
Mason London, HSU River Institute Research Scientist
William J. Trush, Principal Investigator,
Co-Director Humboldt State University River Institute
Jim Graham, Co-Principal Investigator

Department of Environmental Science and Management
Humboldt State University
1 Harpst Street
Arcata, CA 95521

February 28, 2019

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Introduction

A river is at least as wide as its floodplain, including geomorphic terraces and the transitional zone along streambanks, known as the riparian corridor. Those interconnected features play a crucial role in the overall ecological functionality of a stream ecosystem. Healthy riparian corridors generally support tree, shrub and grass species that produce lush vegetation due to favorable water availability and soil quality. The Rush Creek Bottomlands is no different, serving many vital roles that maintain ecological and geomorphic functions of its mainstem channel and floodplain. Some functions include sediment trapping, filtration and buffering of water, the construction and maintenance of streambanks, floodwater and energy storage, recharging of groundwater, preservation of biological diversity, and the creation of primary productivity. A river ecosystem is "healthy" if it can perform the described ecological functions and maintain its capacity for self-renewal. Monitoring the condition of these functions and the species that rely on them is important for documenting ecosystem response to watershed and streamflow management.

This annual report summarizes findings from the Mono Lake basin RY2018 monitoring season in the Rush Creek Bottomlands (Figure 1). The primary goal of this research was to continue long-term monitoring that objectively evaluates ecological performance in response to instream flow recommendations described in the Synthesis Report (2010). Monitoring seasons from runoff years (RY) 2016, 2017, and 2018 focused on (1) geomorphic response by evaluating changes in stream channel morphology and (2) documenting riparian cottonwood and willow tree vigor under varied geomorphic settings affecting water availability. Additionally, RY2018 was the second year of collecting spectral imagery using remote sensing with an Unmanned Aerial Vehicle (UAV) for geomorphic monitoring tasks and alternatively measuring cottonwood and willow tree vigor.

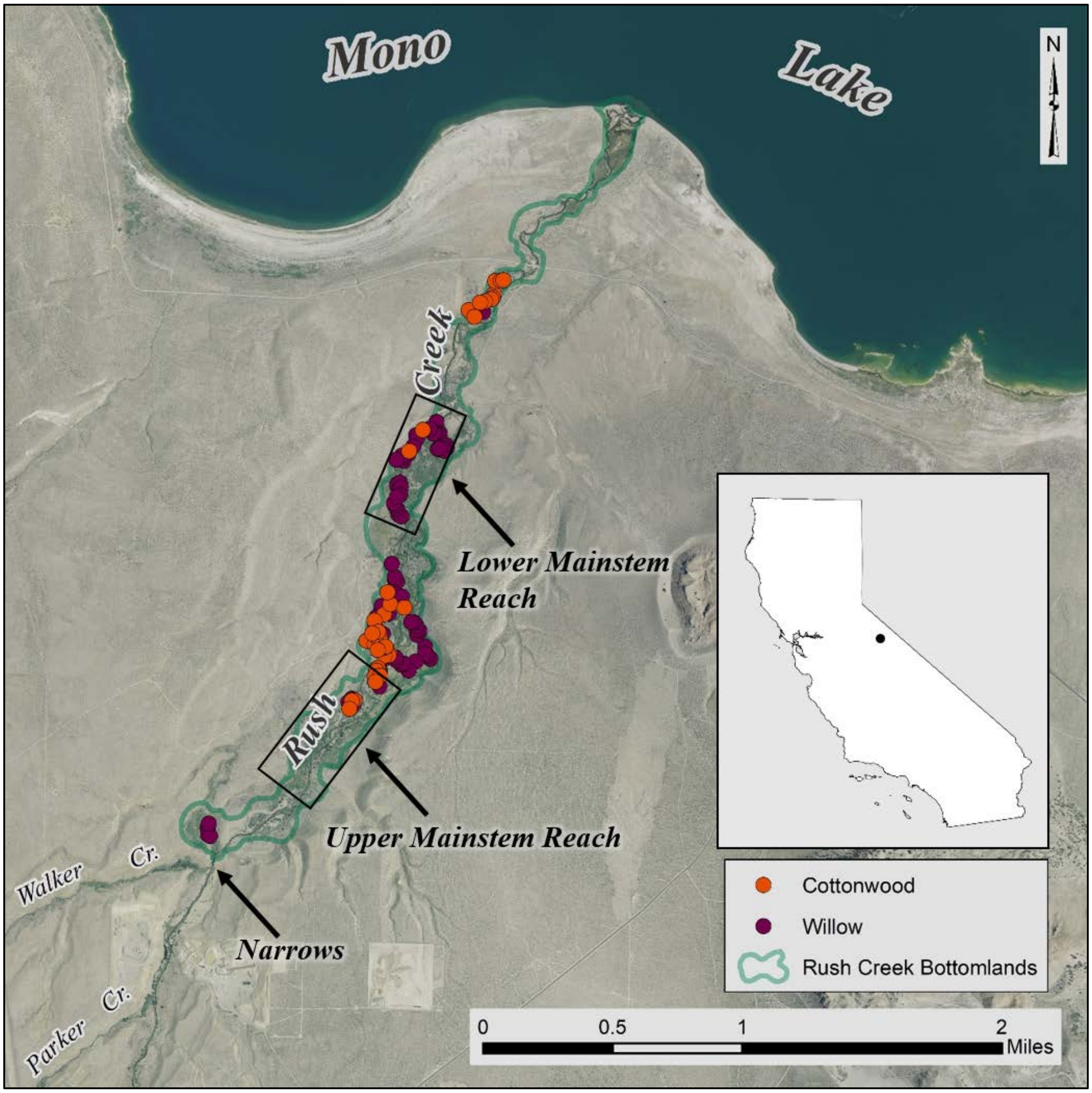

Figure 1. Located in the Mono Lake Basin of the Eastern Sierra of California, the study site includes selected Cottonwood (light orange) and willow (dark purple) trees and stream monitoring reaches of the Rush Creek Bottomlands downstream of the Narrows (ESRI, 2018; NAIP imagery, 2012).

Jordan Adair, a graduate student of Co-Principal Investigator Jim Graham, did the UAV fieldwork and data processing of the UAV's high spatial resolution. Special thanks go to Robbie Di Paolo of the Mono Lake Committee for his much-appreciated fieldwork, lively field discussions, and project coordination. Additional thanks go to Greg Reis, the Mono Lake Committee Hydrologist, for his continued support and development of annual hydrographs from LADWP's stream gaging.

Methods and Findings

The past three years of monitoring provided data on channel morphology and riparian tree vigor in response to three runoff year types, including a 'Dry-Normal I' runoff year (RY2016), an 'Extreme-Wet' runoff year (RY2017), and a 'Normal' runoff year (RY2018). This enabled us to analyze whether our findings from last year accurately depicted tree vigor in response to runoff year type with respect to released flows. We assessed if these conclusions regarding tree vigor and water availability were supported in RY2018 and whether they strengthened our overall monitoring strategy.

High runoff during RY2017 limited geomorphic monitoring. Streamflows in RY2017 were too high to reliably measure all desired morphological stream channel features previously recorded in RY2016. The RY2018 annual hydrograph was a 'Normal I' runoff year allowing us to reliably measure morphological stream channel features we could not in RY2017.

This year's monitoring season also marked the second year of high resolution imagery data collection with remote sensing. Having remotely sensed data from two types of runoff years (RY2017 and RY2018) provided an opportunity for comparing changes in channel morphology. These data also allowed us to assess their utility in assessing riparian tree vigor. Remotely sensed data were ultimately evaluated as an alternative to capturing riparian tree vigor compared to on-the-ground, manual ABI methodologies.

Streamflow in Rush Creek Bottomlands

Estimated flows for Rush Creek describe runoff year types. For reference, we displayed annual hydrographs within our study area 'below the Narrows' (Figure 2). The runoff year types (i.e. 'Dry-Normal I', 'Extreme-Wet', 'Normal', etc.) classified in the Mono Basin Synthesis Report (2010) were utilized for assessing annual water availability in the Rush Creek Bottomlands.

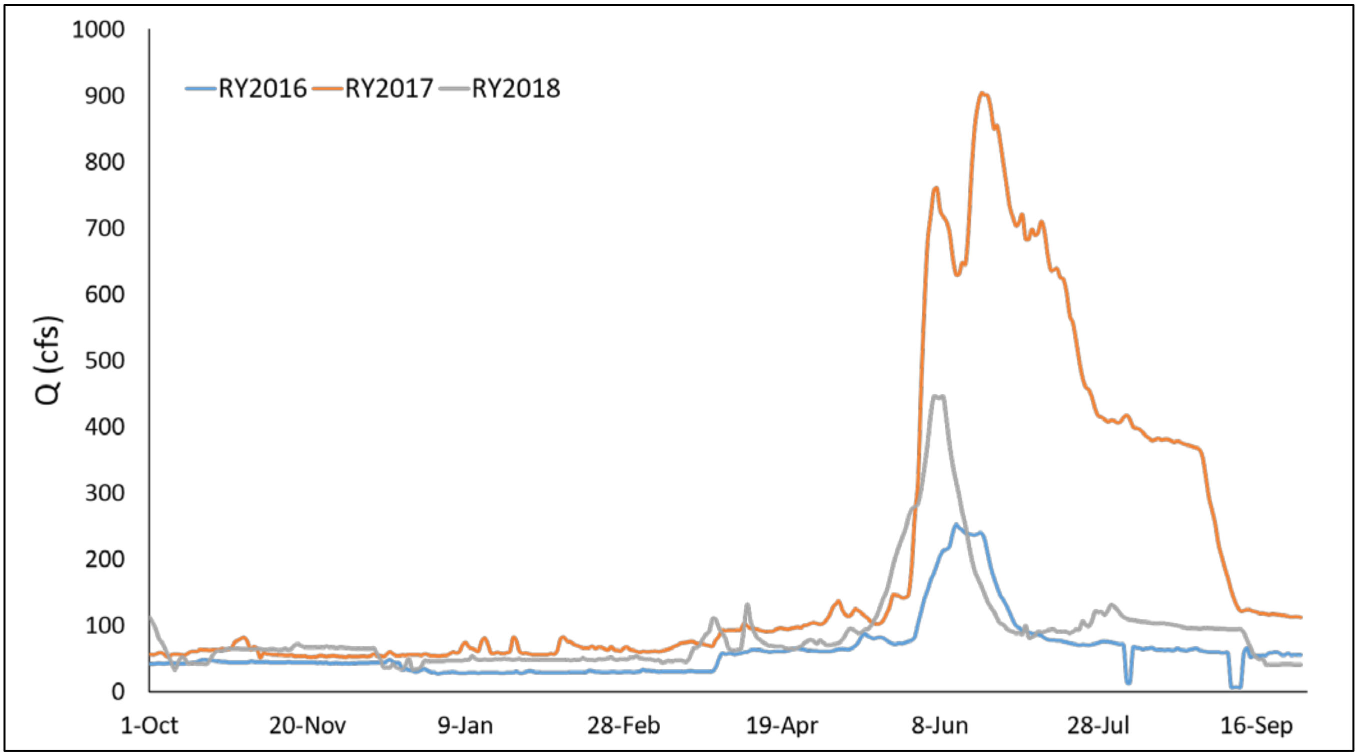

Figure 2. Hydrographs for Runoff Years (RY) 2016 ('Dry-Normal I'), 2017 ('Extreme-Wet'), and 2018 ('Dry-Normal I') in the Rush Creek Bottomlands, CA (Mono Lake Committee, 2018).

Preceded by several drought years, RY2016 continued drought conditions and was considered a 'Dry-Normal I' runoff year, totaling 44,571 acre-feet (af). Conversely, RY2017 was a record-setting 'Extreme-Wet' year, with snowmelt flood peaks reaching 760 and 900 cfs accompanied by an extended high flood base flow exceeding approximately 450 cfs; total annual volume was 147,608 af. RY2018 was considered a 'Normal' year, with a snowmelt flood peak near 450 cfs and an annual volume of 64,079 af.

Monitoring Stream Channel Morphology

Two mainstem channel reaches in the Rush Creek Bottomlands, assigned as 'lower mainstem reach' and 'upper mainstem reach', were selected for stream channel morphological monitoring. The lower mainstem reach extends from the Ford upstream to the 10-Channel confluence, and the upper mainstem reach extends from the 8-Channel entrance upstream to approximately 1100 ft below The Narrows (Figure 3). These two reaches were selected because both were primarily single-thread, yet distinctive, thus providing a good test as to whether the physical variables measured could cleanly differentiate the two. Riffle crest thalweg (RCT) depths (to the water's surface), active channel width (W_{ACT}) at each RCT, and deepest pool depth (to the water's surface) upstream of each respective RCT were collected where each feature occurred in 2016. RCT depths and pool depths were measured to the nearest 0.01 ft using a stadia rod, and the W_{ACT} was measured with a surveyor tape to the nearest 0.01 ft. These measurements were repeated in 2018 to document changes occurring after RY2017 and RY2018 (note: in 2018, due to in-field sampling restrictions, only approximately two thirds of the upper mainstem reach was surveyed). The upstream end is symbolized by the dashed yellow line in Figure 3. Stream channel cross sections (XS) were surveyed within the upper and lower reaches in past runoff years and again in RY2018.

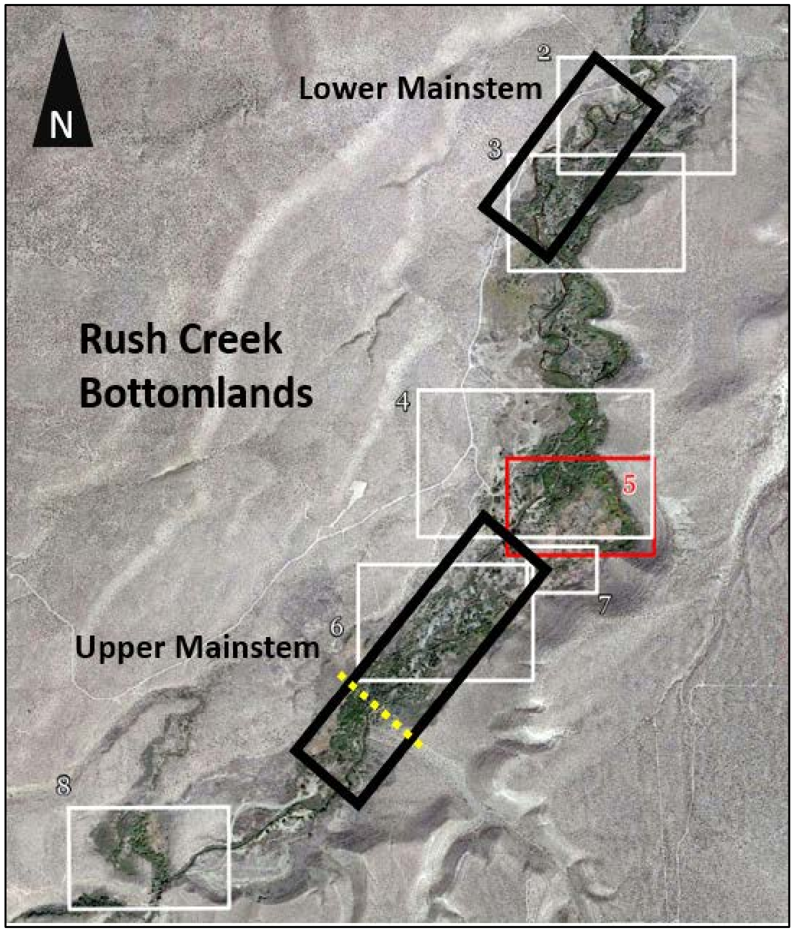

Figure 3. Channel morphology monitoring reaches in the Rush Creek Bottomlands from years 2016 and 2018 are outlined in black. The dotted yellow line in the upper mainstem reach signifies the upstream extent of 2018 surveying efforts. The white, and one red, numbered boxes are locations of riparian zone maps included in ABI fieldwork.

<u>Active Channel Width</u>

The 'active channel' is that portion within the bankfull channel experiencing frequent scour (Figure 4). Active channel streamflow is considered to occur at the onset of channel-controlled flow as the active channel flow transitions from bedform hydraulic control. The active channel streamflow is closely related to RCT depth because the riffle crest also acts as a bedform hydraulic control. By measuring the active channel width (W_{ACT}) at each riffle crest (RC), a distinctive and frequently occurring geomorphic location, changes in channel morphology can be quantified without surveying cross sections requiring fixed locations. This avoids fieldwork complications and allows the practitioner the opportunity to measure how bedforms change, rather than how the active stream has impacted a fixed, surveyed

XS. False negatives might occur when extrapolating data from a single survey at a fixed location following a prominent flood.

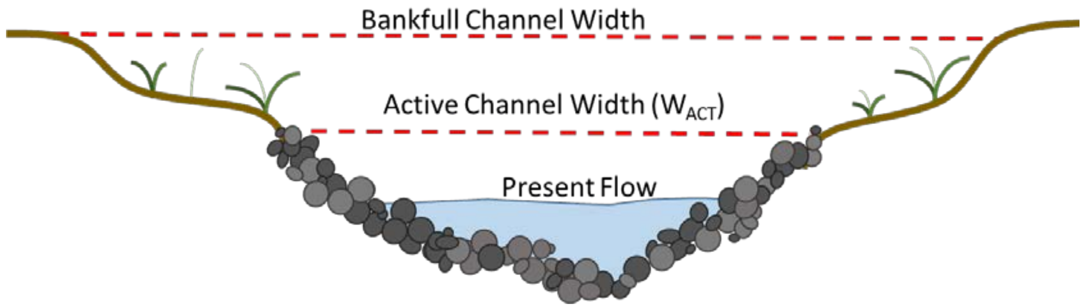

Figure 4. A stream cross-section depicting where to locate active channel width and bankfull channel width.

W_{ACT} at each measured riffle crest was ranked to compute the exceedence value (P-value, %) for each width measured. Ranking generates a separate cumulative distribution of W_{ACT} for each runoff year surveyed. The exceedence value is a cumulative probability estimate; therefore the W_{ACT} measurement associated with the ranked P-value is the approximate percent a given width will occur at a RC. Two sample t-tests were computed to assess significant differences in median annual W_{ACT} among upper and lower reaches across runoff years 2016 and 2018.

<u>Residual Pool/Run Depths</u>

Residual depth, of either a pool or a run, is a measurement of the deepest location in a hydraulic unit (e.g., pool-riffle) independent of ambient streamflow. Residual depth is the difference in depth between a pool and the downstream RCT, functioning as the hydraulic control for the upstream pool (Figure 5) (Lisle, 1987). Pools can be highly sensitive to disturbance of watersheds and riparian areas, experiencing fill and scour. Monitoring inter-annual changes in residual pool depths is important for tracking and assessing fish habitat. Exceedence probabilities of residual depths measured were plotted for evaluating differences spatially (i.e., in the upper reach versus lower reach) between RY2016 and RY2018.

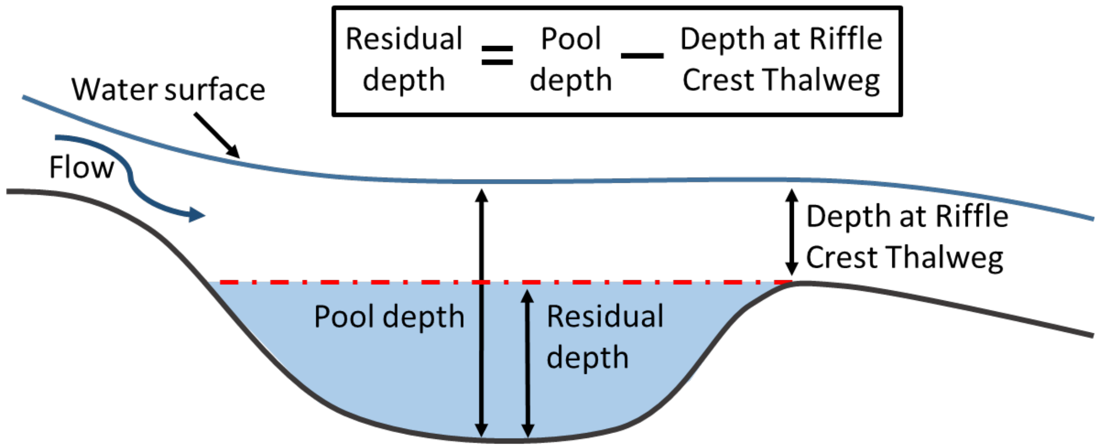

Figure 5. A longitudinal profile of a stream reach detailing measurements for calculating residual depth.

Stream Channel Cross Sections

In 2018, four stream channel cross sections (XS), which had been initially surveyed in August 2015, were resurveyed to assess channelbed changes. Two cross sections were near the old gage station site in Riparian 3 stream section, located in the upper section of the lower mainstem reach (Figure 6). The downstream XS in this reach was named *Jeff Pine XS*, due to it being located between the last two trees of a line of Jeffery Pines (*Pinus jeffreyi*) along the left bank. The upstream XS in this reach was named *Transducer XS* because there is an existing pressure transducer near the left bank survey pin. The other two cross sections were located in the upper portion of the upper mainstem reach. *Beaver Nibble XS* is located in the Riparian 6 section of Rush Creek, approximately 300 to 400 yds upstream of the Channel-4 entrance where beaver activity was observed. The *4-Channel Entrance XS* is located just upstream from the 4-Channel inlet (Figure 7). The *Beaver Nibble XS* was surveyed every year between 2015 and 2018, and the *4-Channel Entrance XS* was surveyed all those years except 2016.

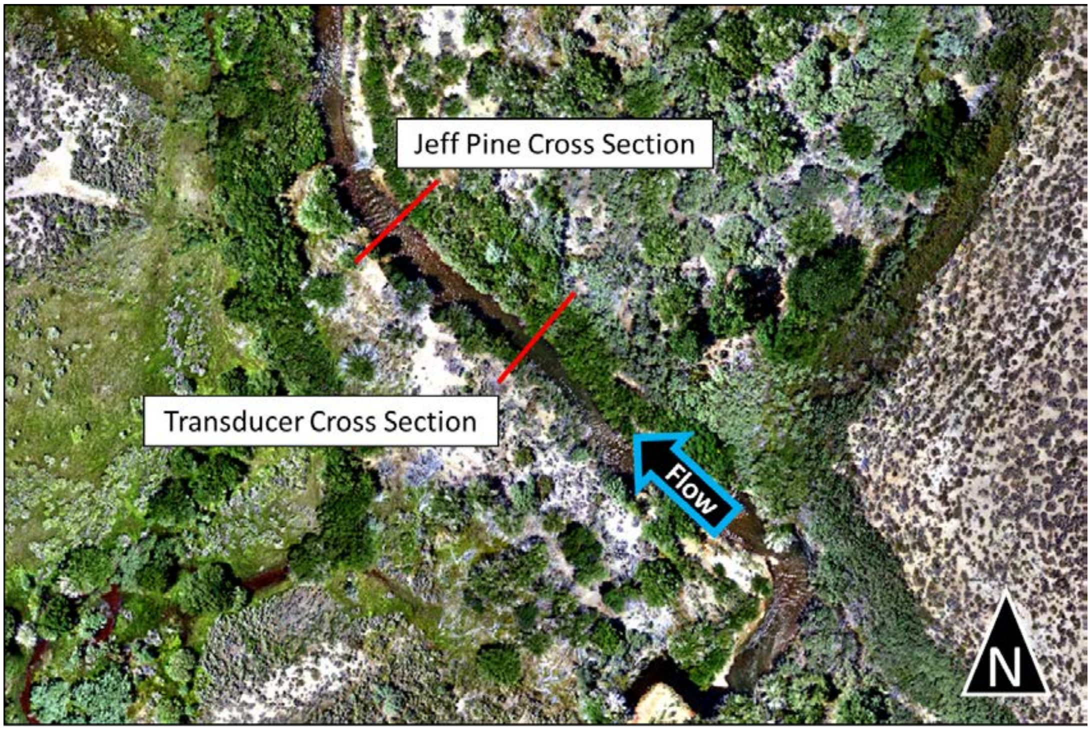

Figure 6. Approximate location of the Jeff Pine XS and the Transducer XS, displayed as red lines, in the Riparian 3 section of Rush Creek (Adair, 2018).

Figure 7. Approximate location of the Beaver Nibble XS and the 4-Channel Entrance XS, displayed as red lines, in the Riparian 6 section of Rush Creek (Adair, 2018).

Results and Discussion

Active Channel Width

In RY2016 two methods of measuring W_{ACT} were used to evaluate monitoring protocols for documenting annual W_{ACT} change. The 'RC method' measured W_{ACT} at each RC (Figure 8). The second approach measured W_{ACT} at random channel features. This 'random method' included W_{ACT} at RCs as well as points of maximum point bar curvature, mid- and lower-riffle locations, pool/run features, and split channels. The 'random method' measured W_{ACT} every 100 feet. These two methods were compared by ranking the W_{ACT} measurements and then calculating the exceedence value (P-value) for each measured width. The exceedence value is a cumulative probability estimate, therefore the W_{ACT} measurement associated with the ranked P-value is the approximate percent that will occur at the corresponding width at a RC or every 100 feet.

Figure 8. Emily Cooper (left) measured the riffle crest thalweg depth to the water's surface along a surveyor tape stretched out the length of the active channel width in Rush Creek. Bill Trush (right) recorded these measurements (Photo: 1 Oct., 2019, 11:24 AM).

Both sets of W_{ACT} measurements (at RC and every 100 feet) were plotted on the same graph for each reach to compare how individual W_{ACT} values varied between methods (Figures 9 & 10). A histogram of the upper and lower reach W_{ACT} values illustrates the variability in applying both methods (Figure 11).

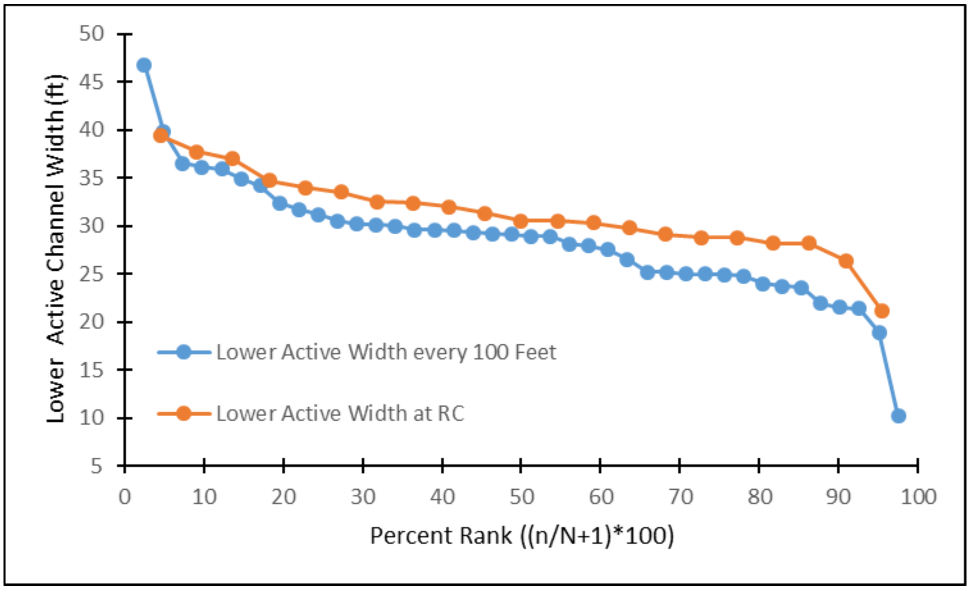

Figure 9. Cumulative distributions for RY2016 active channel widths recorded at riffle crests (orange) and every 100 feet (blue) on the lower mainstem reach in the Rush Creek Bottomlands, CA.

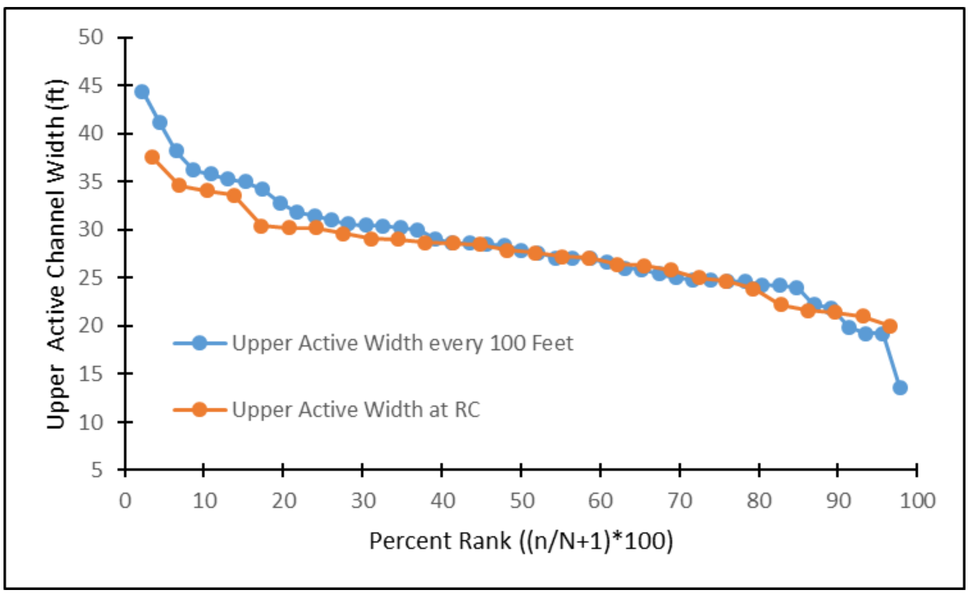

Figure 10. Cumulative distributions for RY2016 active channel widths recorded at riffle crests (orange) and every 100 feet (blue) on the lower mainstem reach in the Rush Creek Bottomlands, CA.

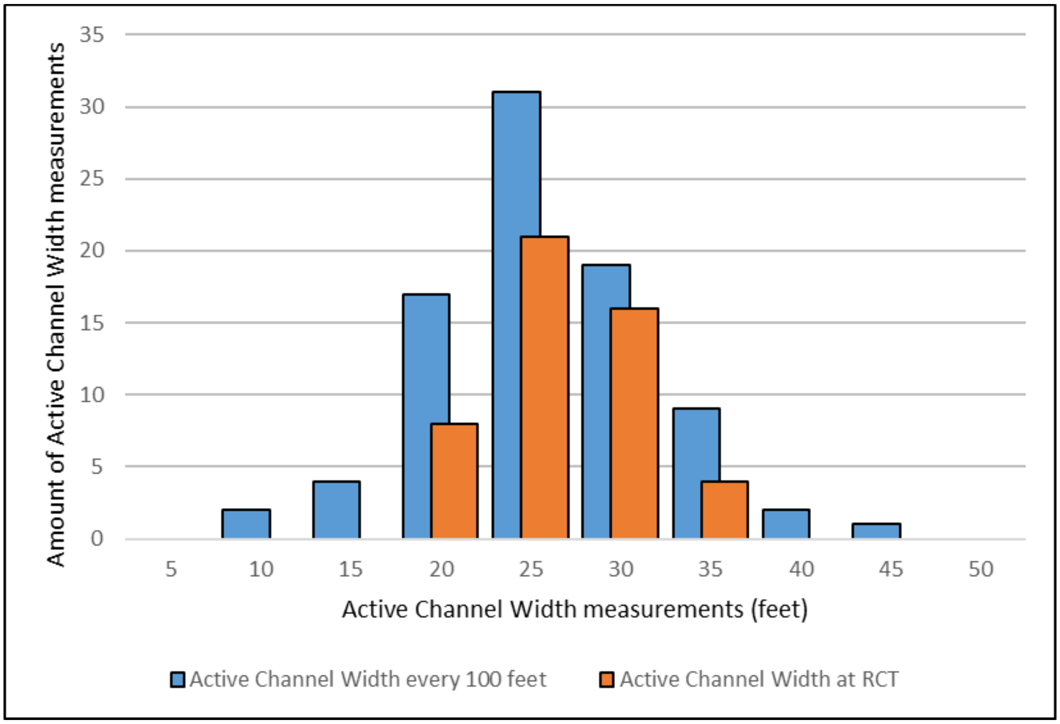

Figure 11. Distribution of active channel widths measured at riffle crests (orange) and every 100 feet (blue) for the upper and lower reach. Each measurement was grouped in increments of five feet (e.g., the '20' on the X axis includes recorded values ranging from 20.00 feet to 24.99 feet).

For upper and lower reaches, the 'random method' of measuring W_{ACT} had a wider distribution compared to the W_{ACT} measured at RCs. For the lower reach, the 'random' W_{ACT} had a range of 46.8 ft (2.4% exceedence) to 10.2 ft (97.5% exceedence). The W_{ACT} measured within the same reach only at RC locations, however, had a narrower range with a greatest width of 39.4 ft (4.5% exceedence) and narrowest width of 21.2 ft (95.4% exceedence). In the upper reach, a similar trend occurred where the 'random' approach had a greater range in values compared to the range in values with the 'RC method'. The difference in range of values, or variance, among the upper and lower reaches and the two methods for measuring W_{ACT} was also revealed in the shape of the exceedence curves (Figures 9 and 10). The smaller slope of the exceedence curves for the W_{ACT} widths measured at RCs shows that there is more consistency with these values. The mean value of W_{ACT} for the 'random' approach among the combined upper and lower reach values was 28.41 ft (SD 5.97 ft). The mean of the W_{ACT} measurements at the RC among the combined upper and lower reach values was 29.15 ft (SD 4.55 ft). An F-test of equality of variance for the 'random' W_{ACT} method resulted in a significantly higher level of variance in both the lower (p = 0.02) and upper (p = 0.04) reaches. Greater variance in 'random' W_{ACT} measurements implied a wider range of values would be expected using this method, decreasing the random methodology's ability to detect future width change.

We decided to use W_{ACT} measurements at the RC to compare/contrast year-to-year width change. Therefore in 2018, W_{ACT} at riffle crests only were evaluated between RY2016 and RY2018. Exceedence curves were plotted on the same graph for RY2016 and RY2018 to compare reaches (Figures 12 & 13).

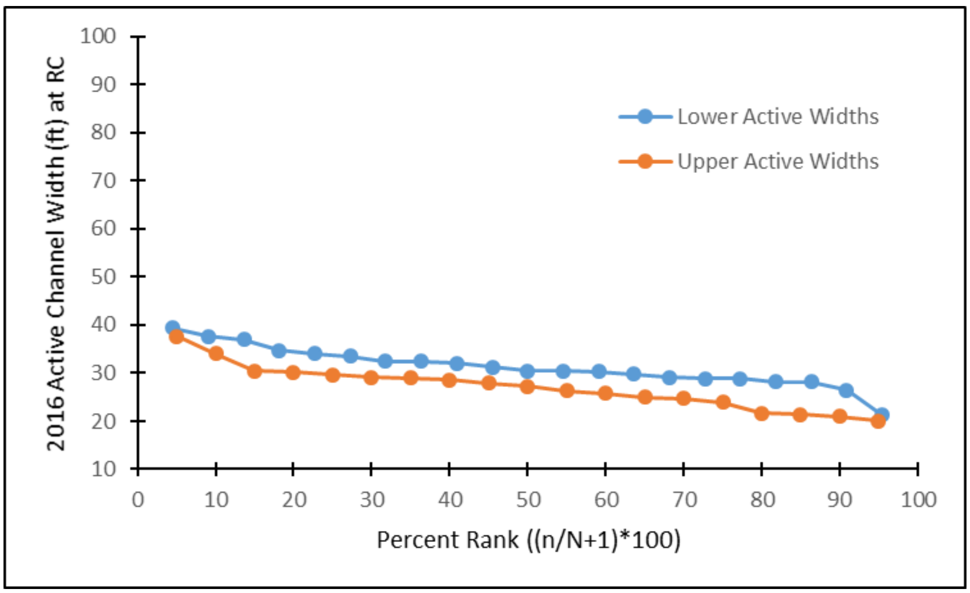

Figure 12. Cumulative distributions for RY2016 active channel widths measured at each riffle crest in the Rush Creek lower reach (blue) and upper reach (orange).

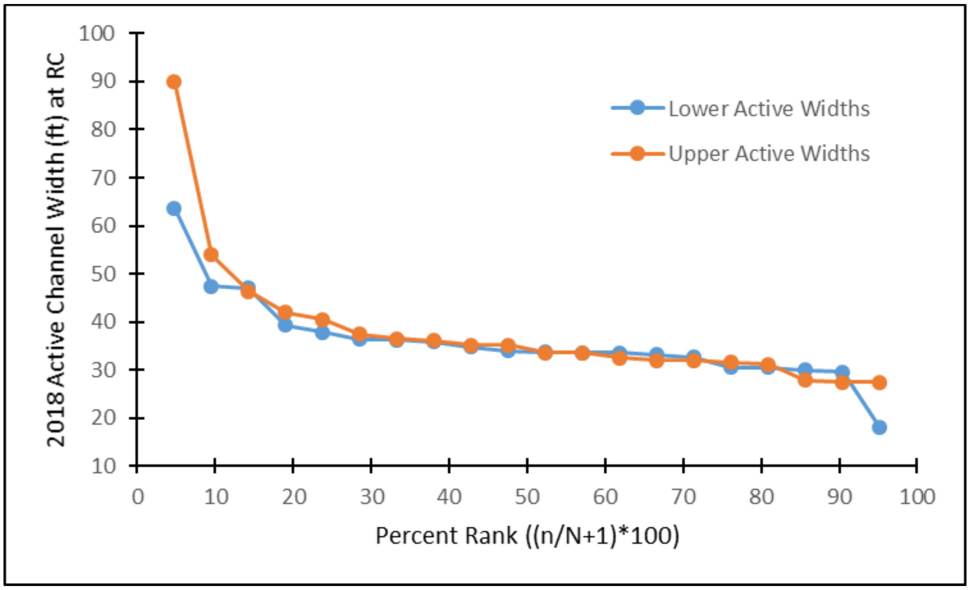

Figure 13. Cumulative distributions for RY2018 active channel widths measured at each riffle crest in the Rush Creek lower reach (blue) and the upper reach (orange).

In RY2016 the lower reach had a median W_{ACT} of 30.5 ft and the upper reach had a median W_{ACT} of 27.2 ft (Figure 12). As predicted, the lower reach had a larger median W_{ACT} because the lower reach is less confined than the upper reach. In RY2018, the median W_{ACT} for the lower reach was 33.85 feet and 34.35 feet for the upper reach (Figure 13). Median widths from RY2018 between the two reaches were more similar to one another than median widths in RY2016. When a two sample t-test (assuming unequal variance) was computed for the active channel width in RY2016, there was a significant difference between the two reaches (p = 0.004). Conversely, the same t-test computed for the RY2018 reaches revealed a significant W_{ACT} difference between upper and lower reaches (p = 0.55). It is likely that the large flooding event in RY2017 mobilized enough of the channel to scour the banks of the more confined upper reach, resulting in more uniformity among upper and lower reaches observed in RY2018.

Next, each year's W_{ACT} exceedence curves were compared to evaluate change from RY2016 to RY2018 (Figures 14 & 15). In RY2018, wider W_{ACT} measurements were recorded in both the upper and lower reaches than previously recorded in RY2016. The median (50% rank) W_{ACT} in the lower reach in RY2016 (30.5 ft) was 3.35 ft narrower than the median W_{ACT} in RY2018 (33.85 ft) (Figure 14). The median

W_{ACT} in the upper reach in RY2016 (27.2 ft) was 7.15 ft narrower than the median W_{ACT} in RY2018 (34.35 ft) (Figure 15). This widening of W_{ACT} in the lower and upper reaches was an outcome of the RY2017 flood hydrograph. The RY2017 flood not only widened the median W_{ACT}, but resulted in wider W_{ACT} across the entire distribution of measurements in RY2018 compared to measurements in RY2016. In RY2018 the upper reach had five W_{ACT} measurements exceeding the widest W_{ACT} in RY2016, with the widest W_{ACT} recorded in RY2016 at 37.6 ft and in RY2018 recorded at 90 ft (Figure 16). The lower reach also experienced greater W_{ACT} in RY2018 than in RY2016. Three RY2018 W_{ACT} measurements exceeded the lowest percentile rank of W_{ACT} for RY2016, whereas the widest W_{ACT} was 39.4 ft in RY2016 and 63.3 ft in RY2018 (Figure 17).

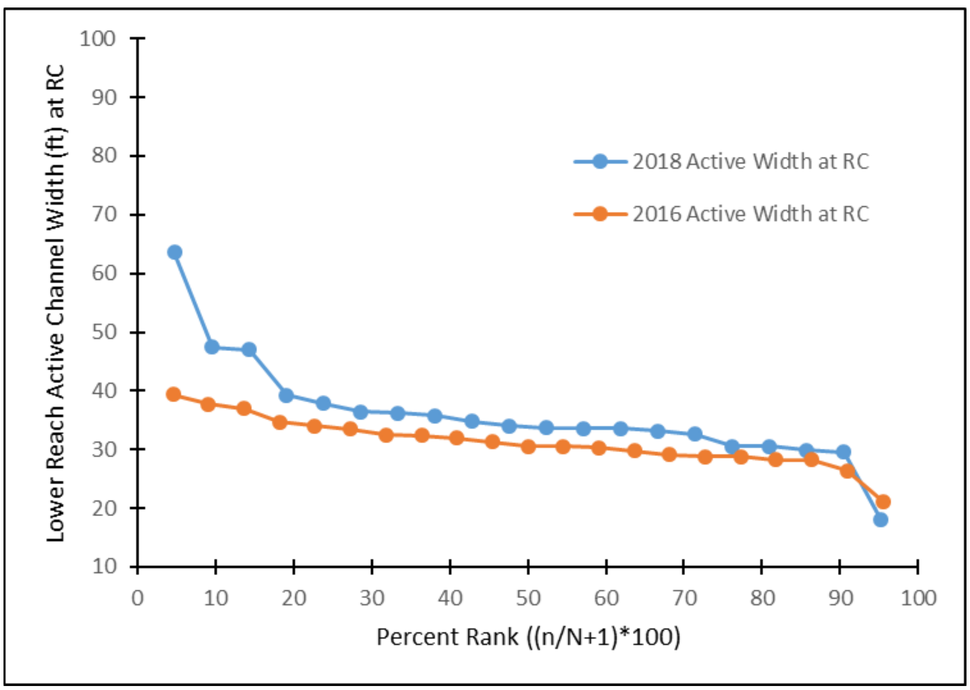

Figure 14. Active channel widths at each riffle crest thalweg for RY2016 (orange line) and RY2018 (blue line) lower reach.

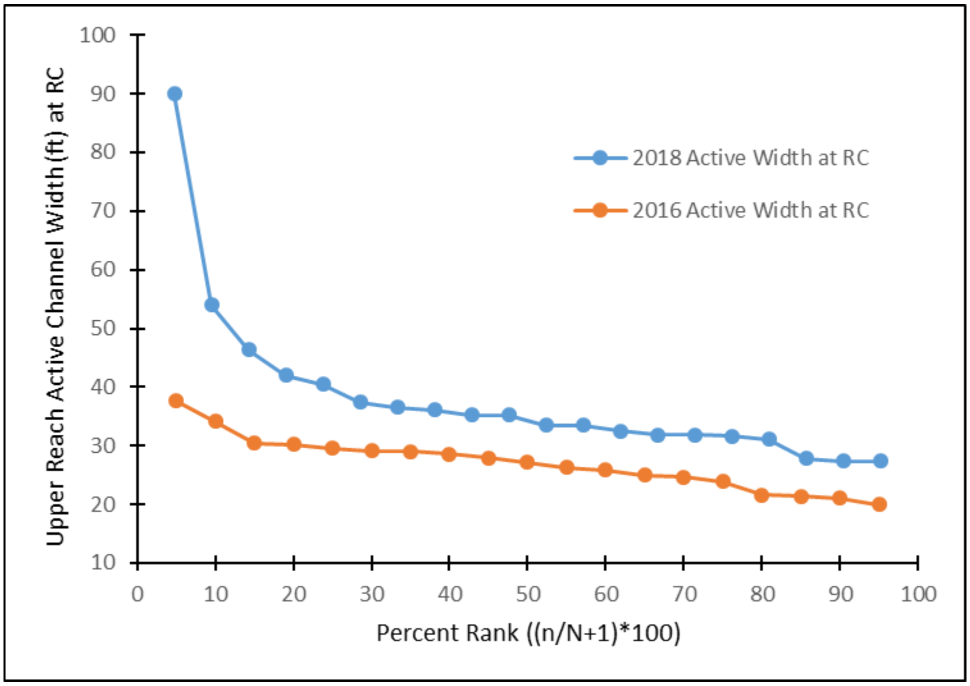

Figure 15. Cumulative distributions for active channel widths at riffle crests for both RY2016 (orange) and RY2018 (blue) upper reach.

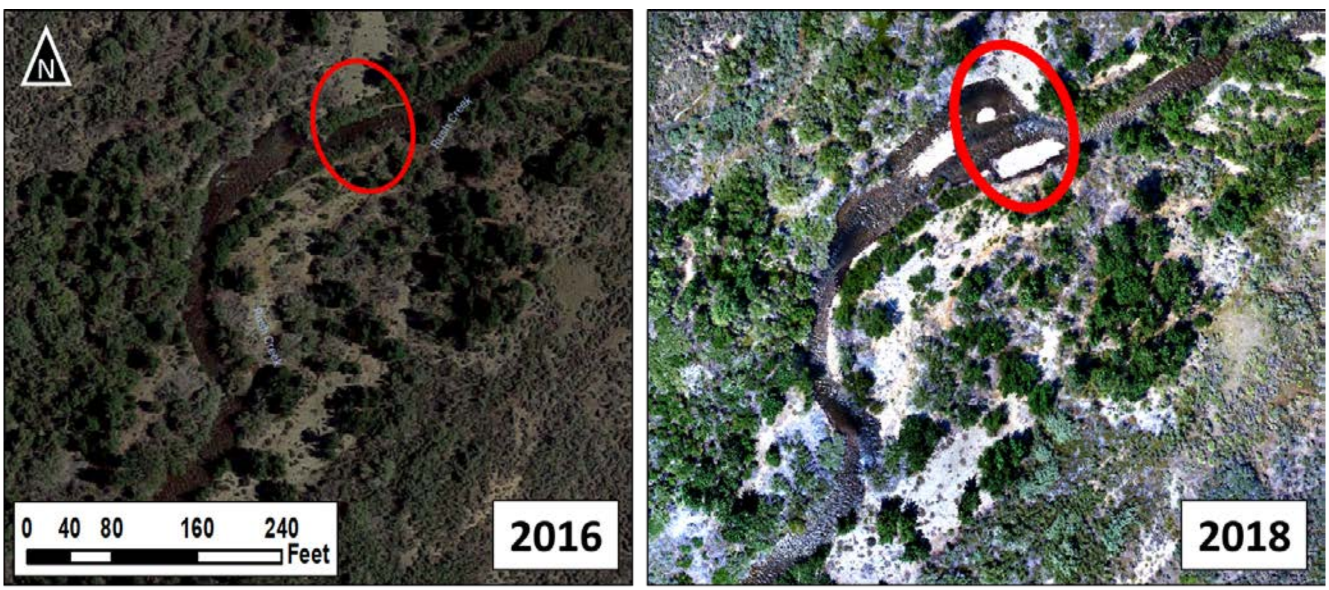

Figure 16. Aerial imagery of the same stream section in Rush Creek showing change in channel morphology from 2016 (Google Maps, 2019) to 2018 (Adair, 2018). The red circle marks the location of a measured active channel width at a riffle crest, showing how the channel widened into the left bank following the RY2017 flood hydrograph.

Figure 17. Aerial imagery of the same reach in Rush Creek showing change in channel morphology from 2016 (Google Maps, 2019) to 2018 (Adair, 2018). The red circle marks the location of a measured active channel width at a riffle crest, showing how the channel widened after 2017 peak flows. Note that the riffle crest in 2018 (with an active channel width at its location of 63.3 ft) was newly formed and therefore there was not a corresponding measurement recorded in 2016. This channel bend is located in the lower portion of the lower mainstem reach within Riparian 2.

Even if these greater widths (lowest percent ranks) in RY2018 are considered outliers, both reaches in RY2018 still experienced overall wider active channels (Figures 12 and 13). Exceedence distributions for the upper and lower reaches changed between RY2016 and RY2018. The lower (p = 0.02) and upper reach (p = 0.001) were wider in RY2018. Wider active channel widths in RY2018 were likely a result of the RY2017 peak flood hydrograph. Greater W_{ACT} widening in the upper reach (Figure 15), than compared to the lower reach (Figure 14), may be explained by the upper reach's greater confinement and reduced sinuosity.

Residual Pool/Run Depths

Residual pool depths were calculated by subtracting the RCT from the deepest upstream pool/run depth (Figure 18).

Figure 18. Mason London measuring the deepest location in a pool upstream of an already measured riffle crest thalweg in the lower mainstem reach of Rush Creek. (Photo taken 29 Sept, 2018, 10:46 am).

Next, exceedence values for each residual pool depth were calculated for the lower and upper reaches for RY2016 and RY2018. Both year's data were plotted onto the same graph for comparison (Figures 19 and 20). There was minor change in the overall residual depths of pools between RY2016 to RY2018 in the lower reach (p = 0.81). Median (50% exceedence) residual pool depths in both years for the lower reach were 2.20 feet. In the upper reach, there were slightly deeper residual pool depths in RY2018 than in RY2016 that were significantly different (p = 0.06). Median residual pool depth in RY2016 in the upper reach was 1.17 feet and 1.52 feet in RY2018, a median difference of 0.35 feet.

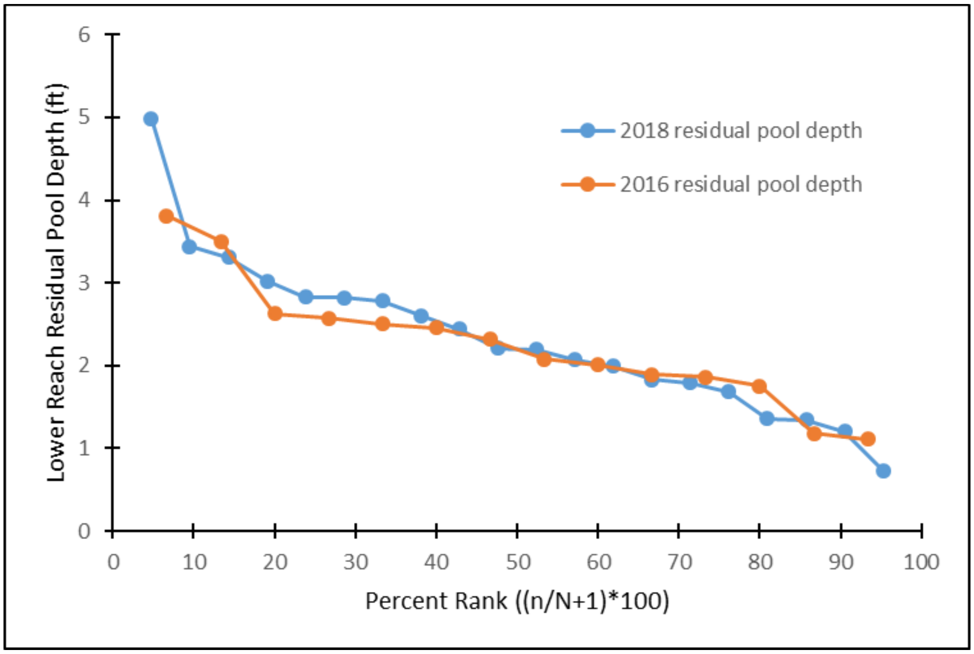

Figure 19. Cumulative distributions for residual pool depths measured in the lower mainstem Rush Creek reach in RY2016 (orange) and RY2018 (blue).

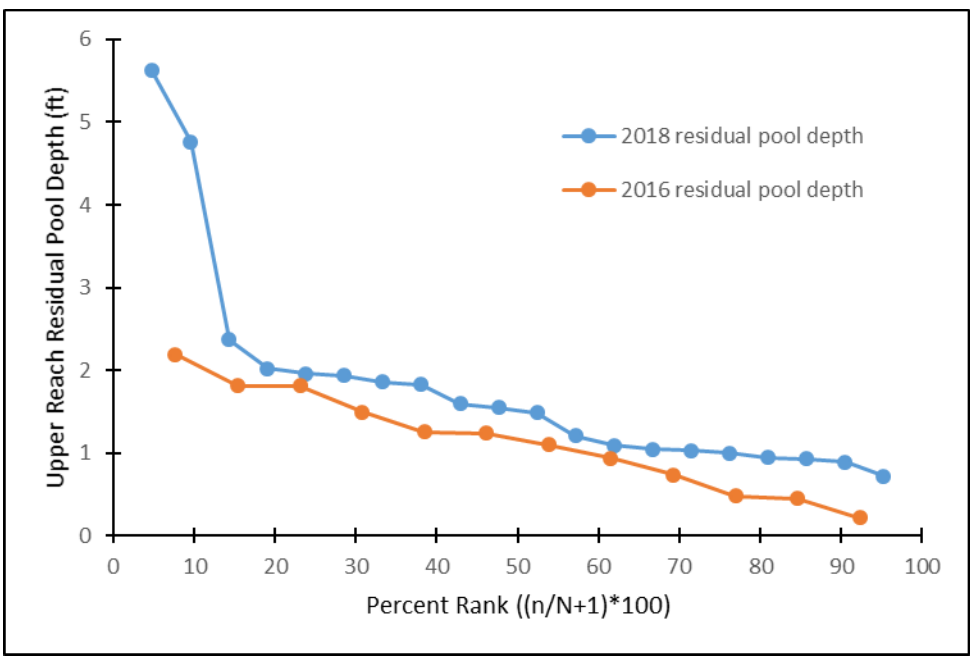

Figure 20. Cumulative distributions for residual pool depths measured in the upper mainstem Rush Creek reach in RY2016 (orange) and RY2018 (blue).

Different geomorphic settings might explain differences in residual depths. The upper reach is more confined, and greater confinement oftentimes generates deeper pools and runs. The lower reach is wider and more sinuous, encouraging lateral scour and deposition during flood peaks. There were two outlier RY2018 pools in the upper reach with considerably deeper residual pool depths. These may have resulted from entire riparian trees collapsing into the channel, stacking into log jams, and scouring pools. These physical features ultimately improve fish habitat by promoting complex bedform development.

Stream Channel Cross Sections

The two cross sections (XS) surveyed in the lower mainstem reach, *Jeff Pine XS* and *Transducer XS*, experienced similar channelbed change between RY2015 and RY2018 (Figures 21 and 22). The top of the left bank benchmark pin was given an elevation of 100 ft.

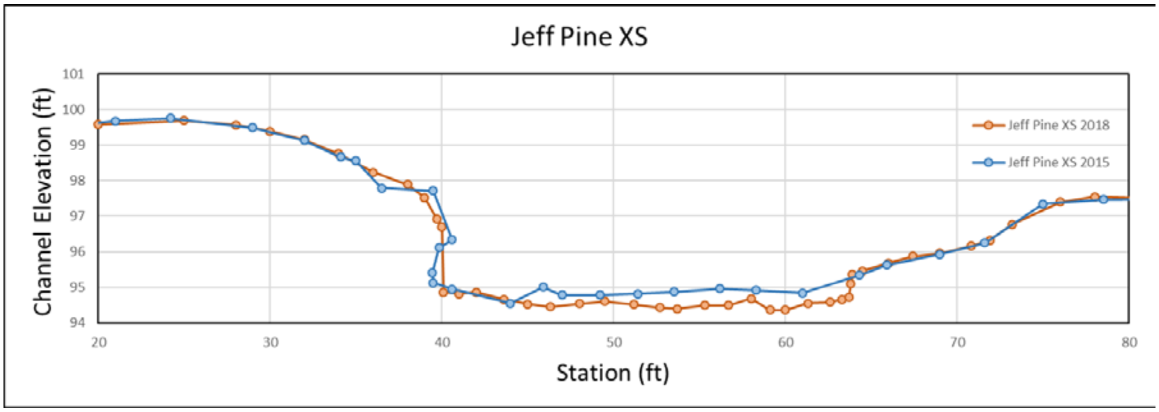

Figure 21. Channelbed elevation (ft) of Jeff Pine cross section in 2015 blue line, and 2018 orange line.

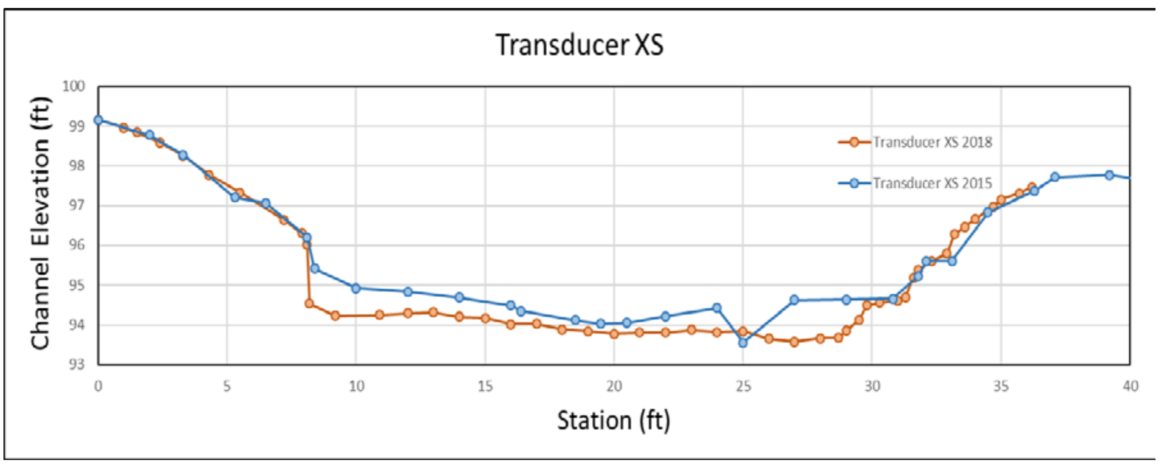

Figure 22. Channelbed elevation (ft) of Transducer cross section in 2015 blue line, and 2018 orange line..

Although *Jeff Pine XS* and *Transducer XS* experienced channelbed erosion between RY2015 to RY2018, the RCT located between the two cross sections (Figure 23) experienced only slightly dropped in channelbed elevation. In RY2015 the RCT was at 94.70 ft elevation; in RY2018 it was 94.65 ft for an elevational drop of 0.05 ft. A nominal difference of 0.05 ft may simply be measurement error in stadia rod placement (e.g., ±0.05 ft from streambed cobbles).

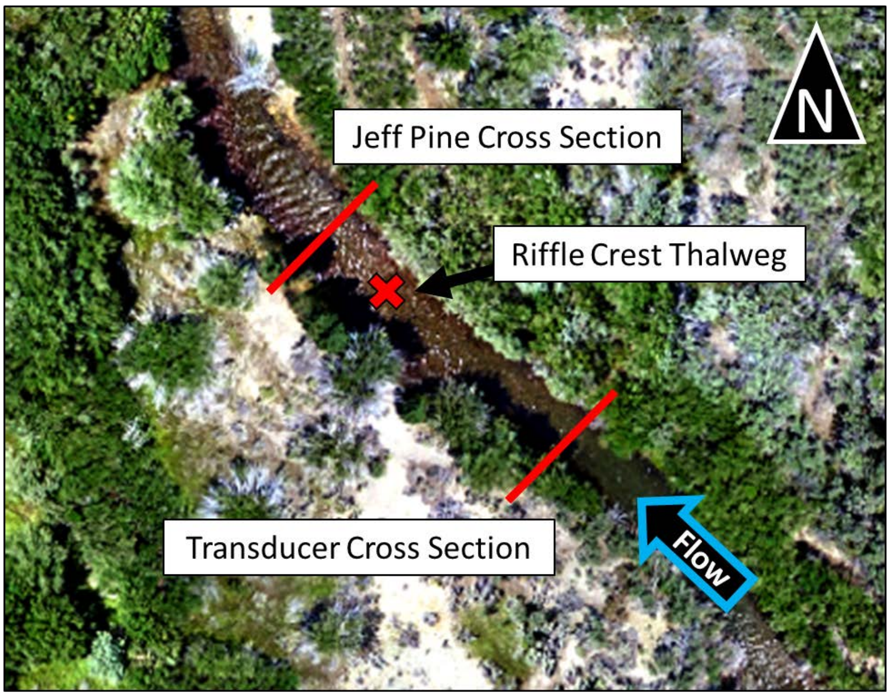

Figure 23. Location of the riffle crest thalweg between the Jeff Pine Cross Section and the Transducer Cross Section in Rush Creek (image collected with UAV in August 2018) (Adair, 2018).

The RY2017 flood hydrograph had greater impact on *Beaver Nibble XS* located in the upper mainstem reach (Figure 24). Closer inspection revealed higher channelbed elevations on river left (Figure 25A) and lower elevations on river right (Figure 25B).

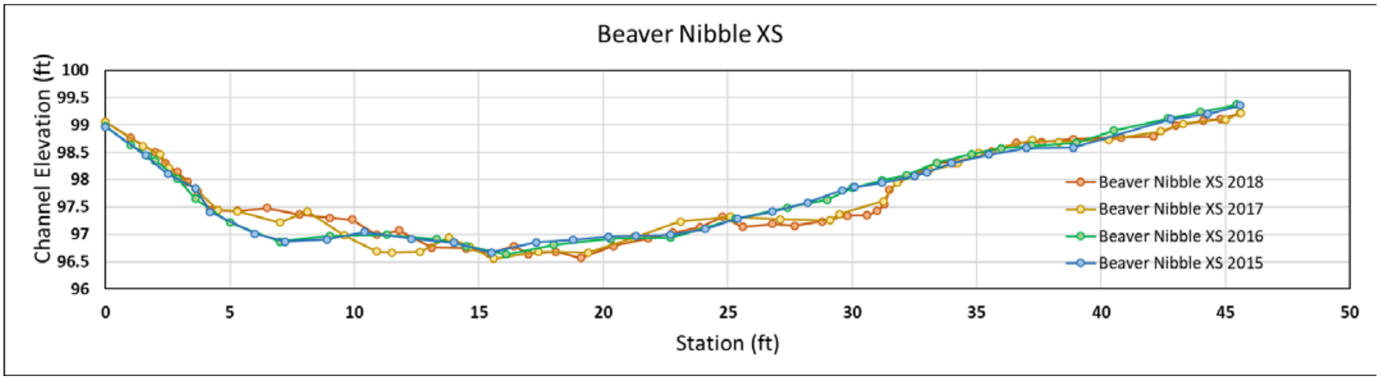

Figure 24. Four years of Beaver Nibble XS survey data (2015 blue, 2016 green, 2017 yellow, 2018 orange) overlaid to observe change in channel morphology of Rush Creek.

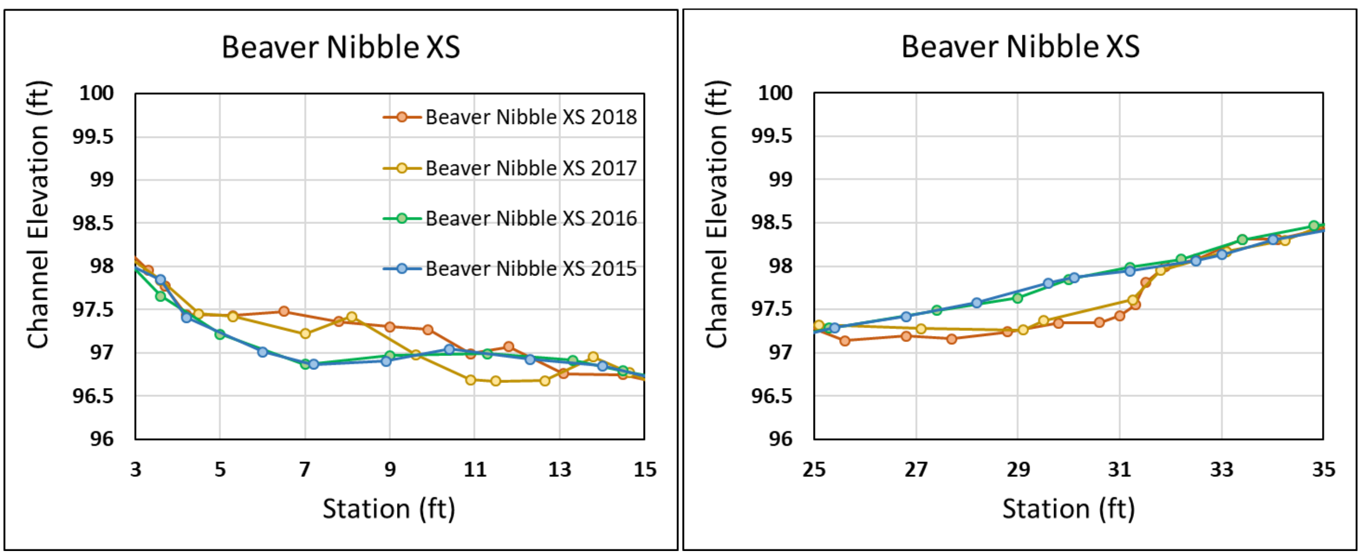

Figure 25. Beaver Nibble XS survey data collected over four years detecting changes in XS channelbed elevation. Note shift in cross section depth from river left to river right after RY2017.

Estimated right bank active channel for *Beaver Nibble XS* was at station 33.4 ft. In 2017 and 2018, a depression formed in the river right streambed near station 32.0 ft (Figure 25B), located within the active channel where frequent scour occurred. Because this cross section is downstream of a right turning bend, scour and bed depression often occurred on the outer bend (river left). Illustrated in Figure 7, the thalweg meanders toward the outer bend (Figure 26).

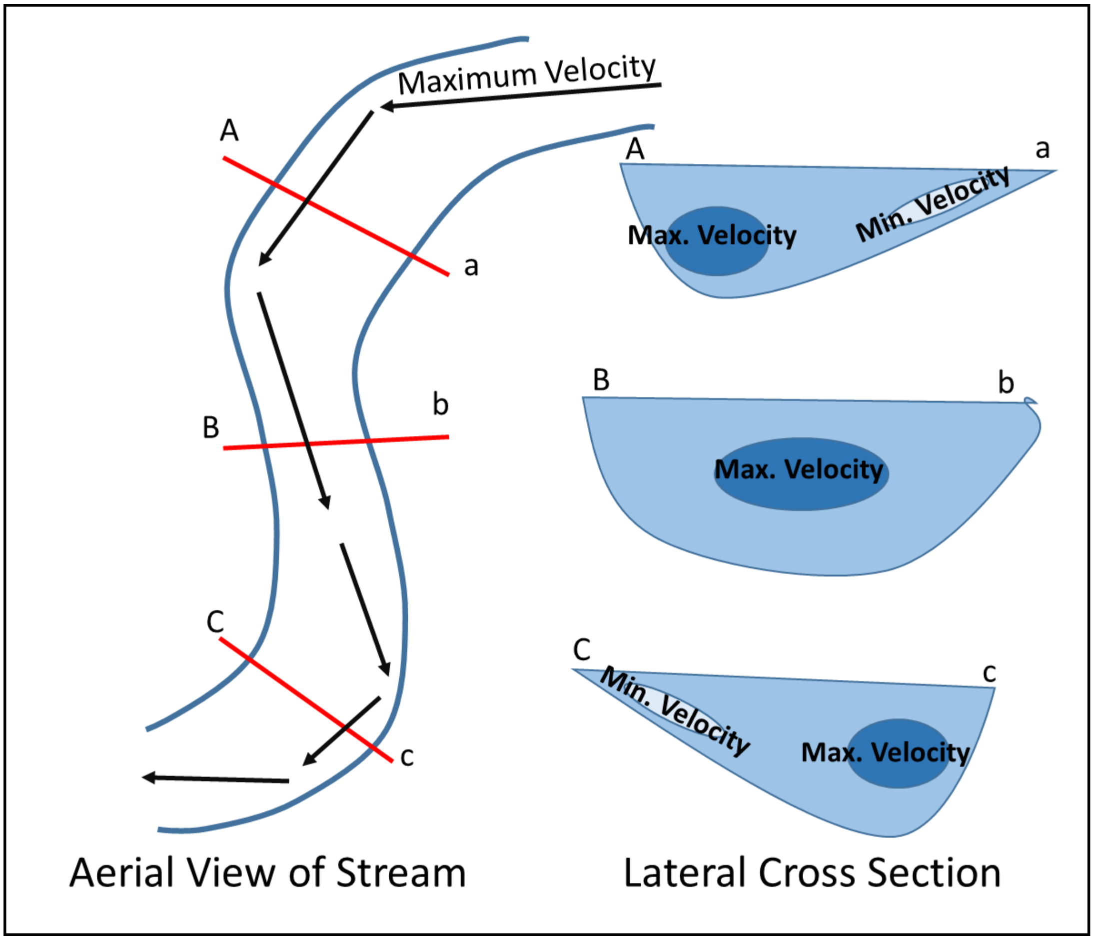

Figure 26. A laterally meandering thalweg affects velocity distribution, scouring, deposition, and ultimately cross sectional shape in a stream.

Erosion along the outer bank of a channel bend causes bend migration. This creates a meandering pattern as the thalweg migrates laterally across the channel from one bend to the next. Typically, sediment is deposited on the inside of a bend where velocity slows, and the channelbed scours on the outside of a bend. Because such a meandering pattern occurs near the *Beaver Nibble XS*, bed scouring was expected along the outer bend at river left in addition to a point bar deposition along the inner bend (river right). Such features were observed from 2015 and 2016 cross section data. However, survey data from 2017 and 2018 show the opposite.

Channelbed elevation on river right lowered more than channel bed elevation on river left (Figures 25A, B).

Changes in the *Beaver Nibble XS* warrants further investigation in RY2019 to hypothesize why this bend appeared to form a bar in a straighter channel as illustrated in Figure 26, cross section “Bb”. The RY2017 flood hydrograph likely had sufficient power to create a more obtuse bend including the *Beaver Nibble XS*. This is supported by the shift from a river right–skewed u-shape (Figure 26, cross section Aa) consistent in 2015 and 2016 into a more uniform u-shape (Figure 26, cross section Bb) in 2017. The trend in bedform shifting toward a uniform u-shape cross section typical of a straighter channel continued in 2018, a ‘Normal I’ water year.

One observation that may explain why this channelbend was trending more obtuse characteristics relates to the nearest RCT hydraulic control. The RCT nearest to the *Beaver Nibble XS* lowered 0.35 ft elevation between RY2015 (97.29 ft) and 2017 (96.94 ft) and then again, albeit slightly, lowered by 0.07 ft from RY2017 to RY2018 (96.87 ft). From 2015 to 2018, channelbed elevation loss in the river right area of the XS was about 0.50 ft and bed elevation loss in the downstream RCT was about 0.42 ft. Channelbed erosion has continued downstream to the RCT, possibly eroding it as well.

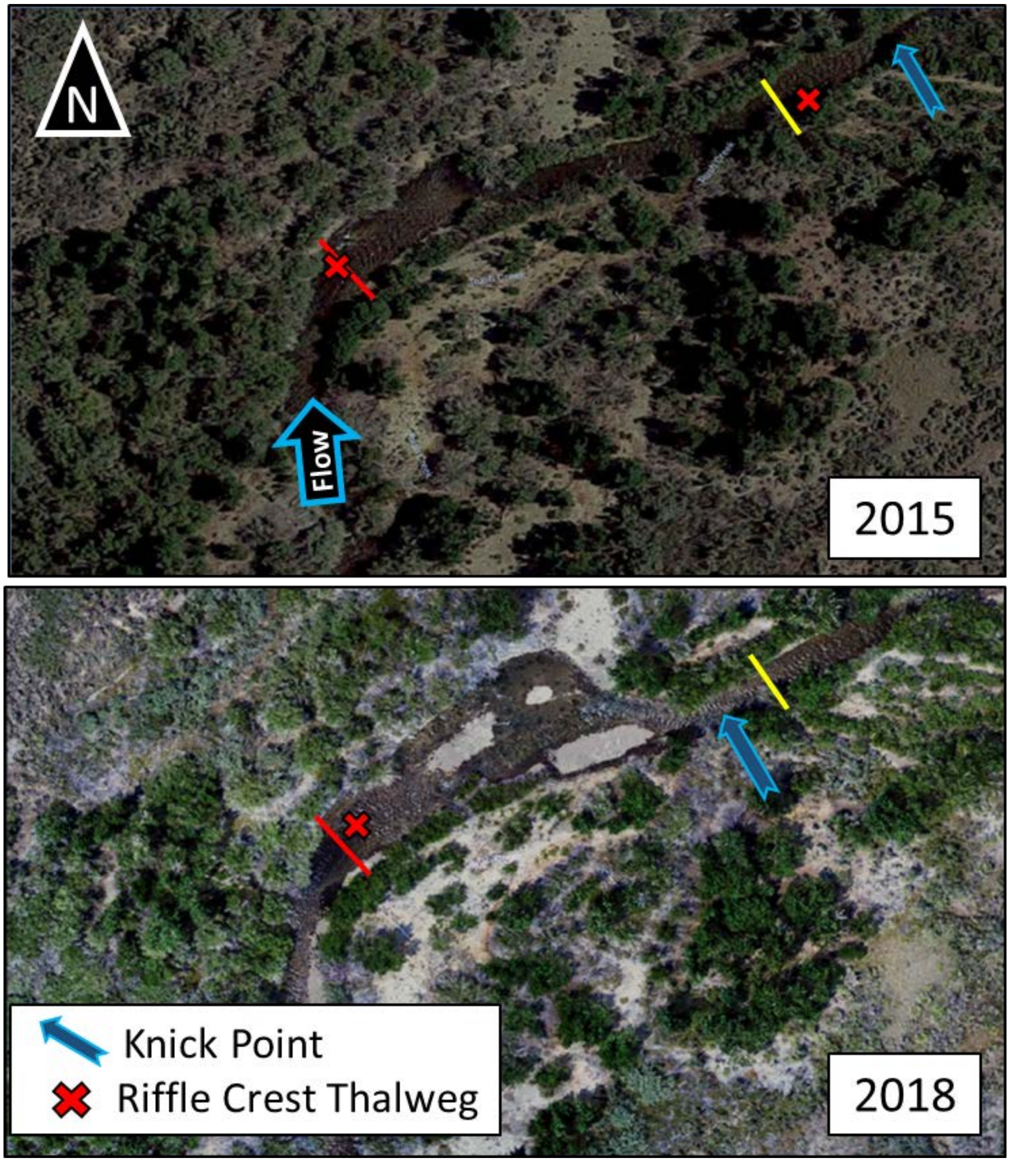

Figure 27. Aerial imagery of the same reach in Rush Creek showing movement of the riffle crest thalweg from 2015 (Google Maps, 2019) to 2018 (Adair, 2018) near the Beaver Nibble XS (red line) and knickpoint migration near the 4-Channel Entrance XS (yellow line). Note the more tightly formed left turn bend upstream of the 4-Channel Entrance XS near the 2018 knickpoint.

From RY2015, RY2017, and RY2018 cross section surveys, the *4-Channel Entrance XS* also underwent significant change in RY2017. The red dotted lines in Figure 28, showing the elevation of peak flow events on the *4-Channel Entrance* XS, offer a visual reference for relative magnitude (depth) of the RY2017 peak flood. Rather than approaching a straighter channel type, however, a tighter, left-curving meander bend (Figure 26, cross section “Cc”) formed slightly upstream of the cross section (Figure 27). This bend deposited coarse bed material, creating more bar area and scouring young willows upstream of the cross section, leading to scour river right of the *4-Channel Entrance XS* (Figure 28). This change in channelbed elevation, throughout the entire streambed cross section, may not have been caused by meander bend evolution, but as a result of knickpoint migration.

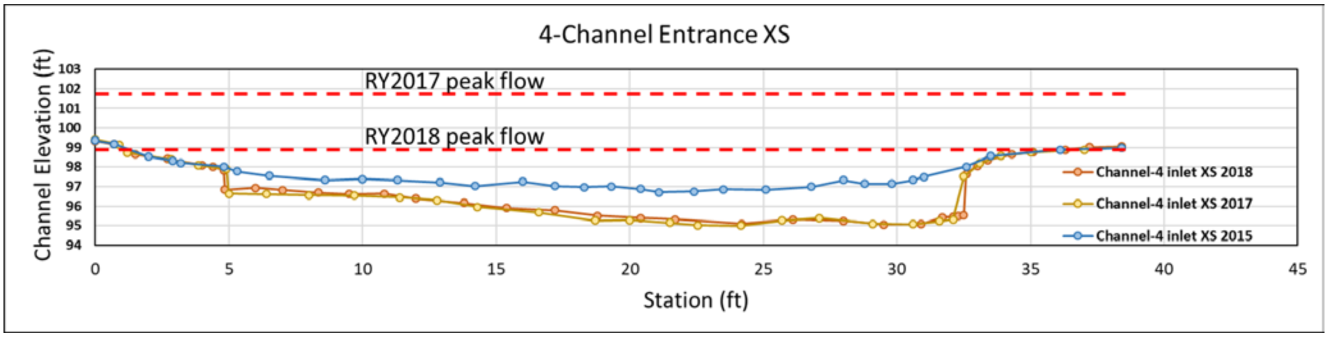

Figure 28. Surveyed 4-Channel Entrance cross sections in 2015 blue, 2017 yellow, 2018 orange.

Successive cross section surveys at fixed location show that interpreting the findings can have several possible causes. This ambiguity will be a topic in RY2019 summer's channel morphology monitoring.

Monitoring Riparian Tree Vigor

Riparian tree vigor was monitored along riparian corridors and floodplain terraces of the Rush Creek Bottomlands during fall of RY2016, RY2017, and RY2018. The goal has been to monitor and assess tree vigor in diverse environmental settings with respect to RY type for black cottonwood (*Populus trichocarpa*), yellow willow (*Salix lutea*), and red willow (*Salix laevigata*). Willms et al. (1998) found annual branch increments (ABI) a useful measurement for assessing tree vigor in response to water availability. We measured annual branch increment (ABI) for selected willow and cottonwood trees and related those measurements to water year type and spatial data throughout the Rush Creek Bottomlands. Predicted relationships among tree vigor, RY type, and geomorphic setting included:

1) Greater cumulative ABI lengths among cottonwood and willow trees in wetter water years;

2) Greater variation in tree response to increased variation in channel morphology (i.e., inundation of side-channels) and more variable water availability;
3) Less ABI variation in ABI among trees in geomorphic settings with more available water from inundation and springs;
4) Shorter willow ABI in response to greater proximity from stream channel;
5) Cottonwood and willow ABI will be longer among younger trees;
6) Greater ABI variation among older and/or stressed trees.

An alternative to monitoring riparian tree vigor was explored in August 2017 and 2018 using Unmanned Aerial Vehicles (UAVs) to collect annual, sub-meter resolution aerial imagery and spectral data in the Rush Creek Bottomlands. These data generated another unit of measure for vegetative vigor known as Normalized Difference Vegetation Index (NDVI). This measure is a ratio of red to near-infrared light (Equation 1). NDVI uses red (VIS) and near-infrared (NIR) wavelengths in satellite-derived or aerial imagery pixels. Plants absorb light in the visible spectrum and reflect light in near-infrared wavelengths; more reflected light in the near-infrared wavelengths indicates dense vegetation (NASA, 2000). NDVI calculations for a given pixel result in values ranging from -1 to +1, with a value close to -1 indicating no vegetation; values close to +1 represent the highest density of photosynthesizing vegetation (NASA, 2000).

Equation 1. $$NDVI = \frac{(NIR-VIS)}{(NIR+VIS)}$$

NDVI has been used in other studies to monitor drought severity (Peters et al. 2002), to determine rates of green-up and senescence (Reed et al. 1994; Pettorelli et al. 2005), and to monitor long-term productivity in agricultural lands (Lenney et al. 1996). We expected measured trends in NDVI to mirror those of the ABI measurements in response to RY type and diverse geomorphic settings.

Tree Selection

Groupings of trees were selected with respect to various floodplain and terrace surfaces throughout the Rush Creek Bottomlands to evaluate whether tree vigor responds to RY type in different geomorphic settings. Initial selection of individual trees was based on: (i) accessibility to the tree (i.e., limited access from dense wood rose (*Rosa woodsia*) in the understory), (ii) occurrence of several branches within arm's reach, and (iii) a visual assessment of general tree health with respect to leaf color, early leaf abscission, or dead branches. Selected trees were assessed for vigor in units of annual branch increment (ABI) (mm) among various spatial settings including grove location, inundation conditions, and proximity to stream channels.

Annual Branch Increment (ABI)

For cottonwood and willow tree species, annual growth occurs terminally along branches. Tree growth responds to the snowmelt hydrograph (Trush et al., 2017, 2018). Annual branch increments (ABI) are separated by terminal bud scars that form completely around a tree branch or stem. After the summertime growing season in Rush Creek, annual growth was measured by sampling each year's ABI. Our technique measured branch length from the bottom of the terminal bud to the nearest terminal bud scar with a metric ruler (Figure 26). A minimum of 20 branches was sampled by randomly selecting branches completely around each tree for ABI measurement. These measurements were repeated over three years in the fall of RY2016 ('Dry-Normal I' water year), RY2017 ('Extreme-Wet' water year), and RY2018 ('Normal' water year), providing a quantitative measure of annual tree vigor to be analyzed with water availability data. While these ABI data provided an annual growth metric for cottonwoods and willows, they were limited to branches within arm's reach and did not include sacrificed (dead) branches.

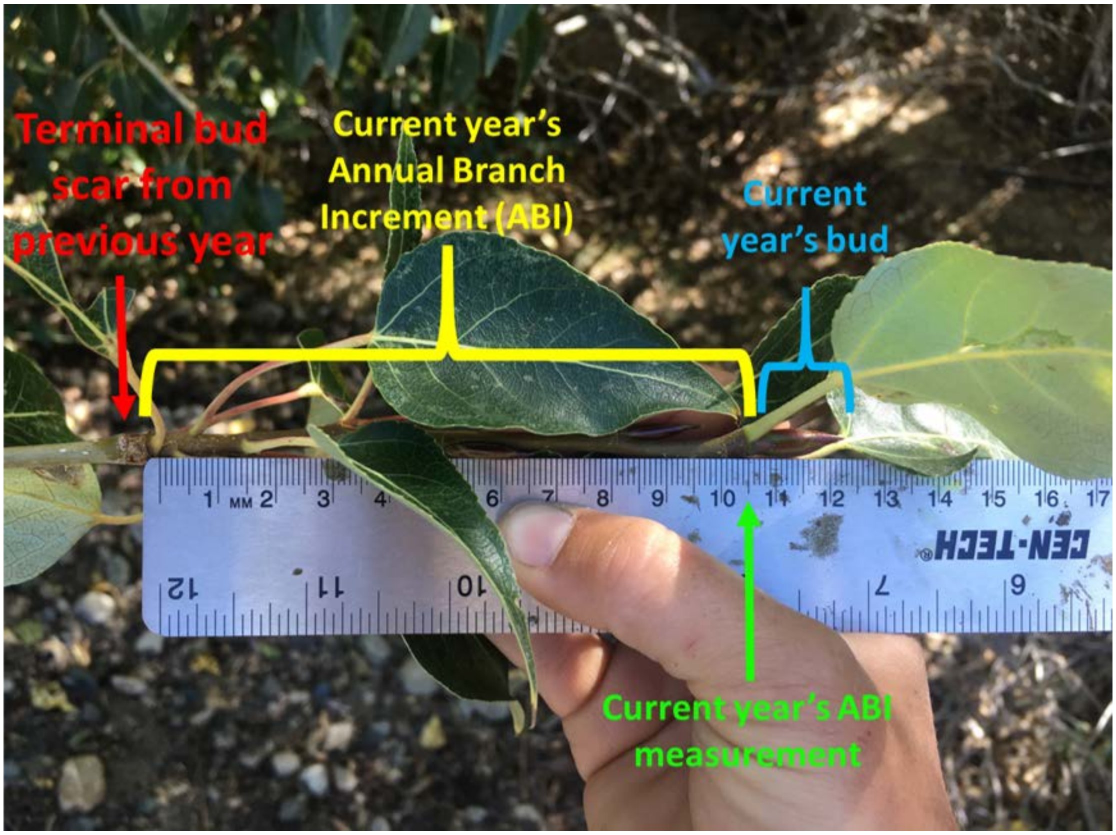

*Figure 26. The current year's Annual Branch Increment (ABI) of a single Cottonwood (*P. trichocarpa*) branch in the Rush Creek riparian corridor was measured from the nearest terminal bud scar (at 0.00 cm on the ruler) to the bottom of the terminal bud (at 10.50 cm on the ruler), excluding the bud in the measurement. For this particular ABI, the growth was 105 mm.*

Approximating Tree Age Related to Vigor

While water uptake among cottonwood and willow trees may be affected by proximity to groundwater in the Rush Creek Bottomlands, tree age also likely affects rate of water uptake and growth. We measured tree base circumference and primary stem diameters as surrogate variables representing tree age. We evaluated differences in ABI among surrogate tree age variables to measure any effects on tree vigor with young versus old trees. This analysis was limited to assuming those surrogate variables represented tree age, an attempt to coarsely distinguish 'old' from 'young' trees, as well as physical limitations to field measurements (Figure 27).

Figure 27. Typical yellow willow tree base in the Rush Creek Bottomlands with many primary stems, potential build-up of sediment around the base, and difficult accessibility (Photo: 26 Sept. 2018, 10:49AM).

Analysis

Cottonwood and yellow willow vigor was evaluated with ABI measurements of individual trees and groups of trees by geomorphic setting. Ranking ABI's with exceedence curves and box plots generated a cumulative distribution of ABI lengths for each sampled tree and within groups of trees more amendable to analysis and interpretation than assigning discrete size classes of ABI lengths plotted in a histogram. We also evaluated cottonwood and willow median ABI and range in ABI lengths for individual trees between runoff years as well as pooled ABI within groups. Beyond comparing median ABI values for cottonwood and yellow willow trees from each monitoring year, we evaluated how median ABI and NDVI trends compared to each other and changed year-to-year.

Results and Discussion

The difference in runoff between RY2016 and RY2017 in Rush Creek was great, given the record-setting 'Extreme-Wet' year in RY2017 compared to a series of drought years leading up to and including RY2016. However, when RY2018 was compared to RY2016 and to RY2017, snowmelt streamflow was relatively similar between RY2016 and RY2018. Riparian tree vigor responded to those year-to-year differences throughout Rush Creek Bottomlands, as demonstrated by our monitoring results of annual branch increment (ABI) measurements.

Annual Branch Increment as Tree Vigor

Throughout all three monitoring runoff years, median ABIs for cottonwood and willow individuals were concentrated around relatively shorter lengths, with the greatest ABI lengths as outliers becoming more frequent each year (Figures 10 and 11). The skewed distribution of median ABI among all individuals per year resulted in average values usually being greater than median values. Similar skewness occurred in the cumulative distributions of ABI measurements in individual trees. Therefore, we used median ABIs opposed to mean whenever one value per tree was analyzed. There was clearly greater ABI among cottonwoods and willows in 2017 compared to 2016, which was expected given the difference in runoff years (Figures 28 & 29). There was minor difference in cottonwood or willow ABI between 2017 and 2018, however there was a greater range having more frequent high-value outliers in the median ABI among cottonwoods in 2018. For both cottonwoods and willows, there was not only greater ABI, but also greater range in median ABI during 2017 and 2018 compared to 2016. Less dramatic differences in ABI from 2017 to 2018 were expected given the change from an 'Extreme-Wet' runoff year to a 'Normal' runoff year, and possibly due to residual effects in water availability from RY2017.

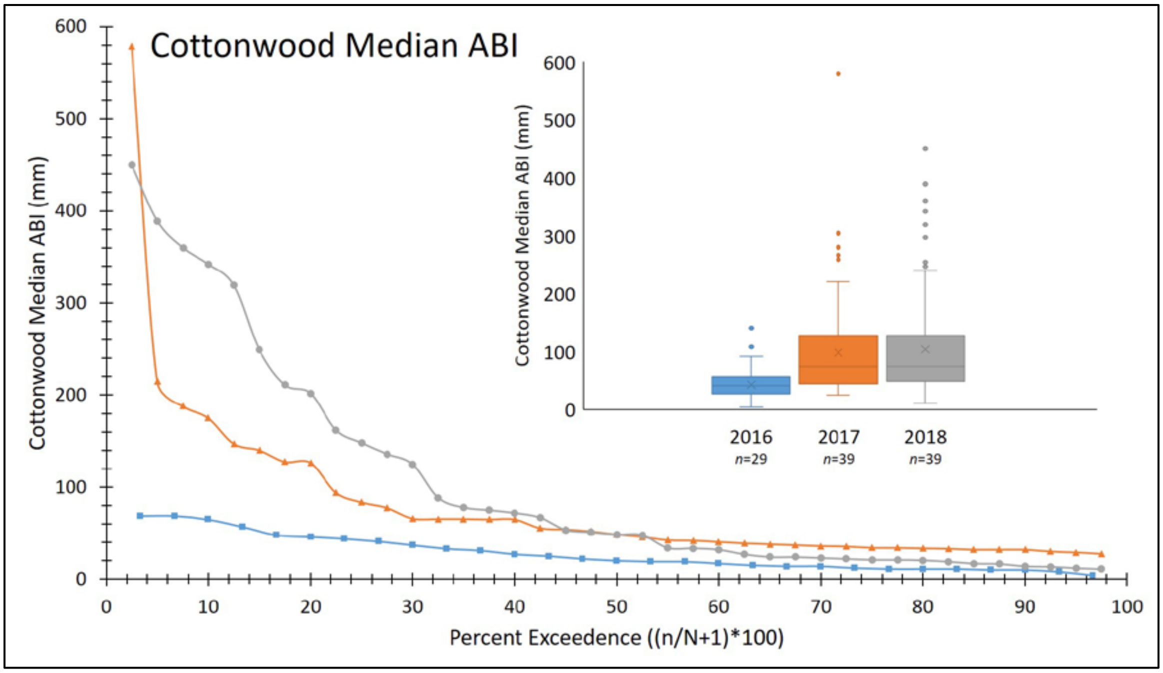

Figure 28. Exceedence curves and boxplot distributions of median annual branch increment values of individual cottonwood trees measured in the fall of 2016, 2017, and 2018 in the Rush Creek Bottomlands, CA. Boxplot whiskers represent min and max with outliers represented as dots; 25th, 50th, and 75th percentiles are represented by box top, midline, and bottom, respectively. Mean values are represented by dark "x"—note skewed distribution's effect on the mean.

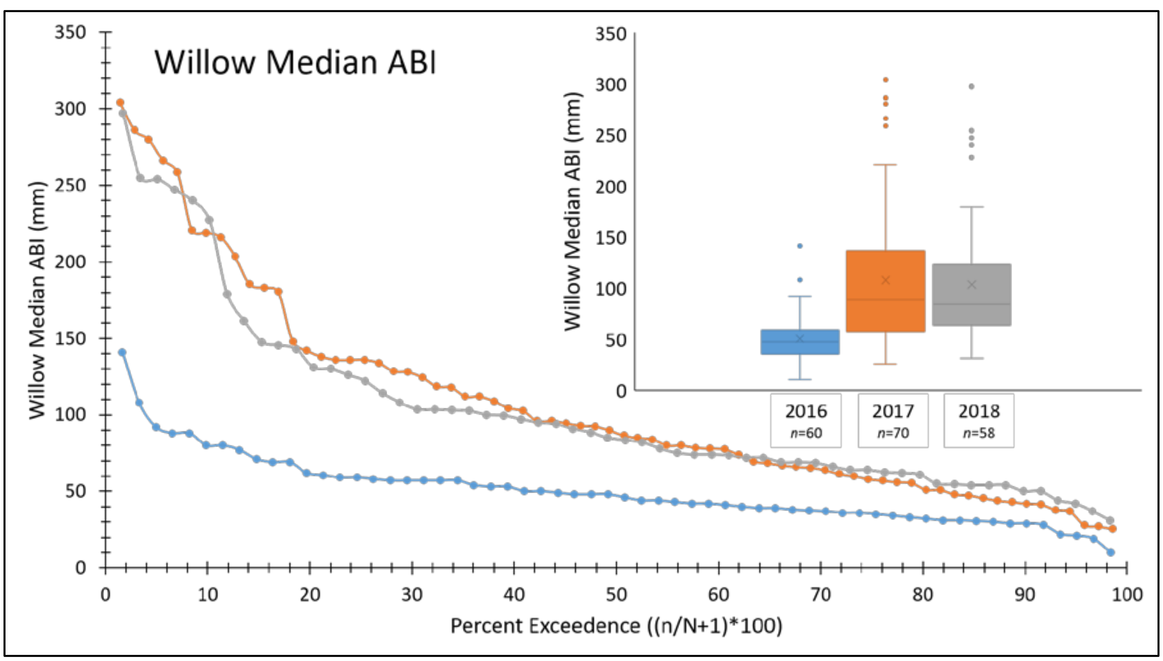

Figure 29. Exceedence curves and boxplot distributions of median annual branch increment values of individual willow trees measured in the fall of 2016, 2017, and 2018 in the Rush Creek Bottomlands, CA. Boxplot whiskers represent min and max with outliers represented as dots; 25th, 50th, and 75th percentiles are represented by box top, midline, and bottom, respectively. Mean values are represented by dark "x"—note skewed distribution's effect on the mean.

We compared median ABI values of cottonwood and willow individuals between 2016 to 2017; 2017 to 2018; and 2016 to 2018, where the dashed line represents a trend with no change in ABI from year to year (Figures 30, 31, 32). Data points scattered below and to the right of the dashed line represent a greater ABI value in the year along the x-axis compared to the year along the y-axis, and vice versa. Figure 30 shows how much greater overall ABI values were for cottonwood and willow individuals in RY2017 compared to RY2016, as many points representing individuals deviate from the dashed line towards greater ABI in RY2017. Most cottonwood and willow individuals sampled substantially changed ABI lengths from RY 2016 to RY 2017. Greater ABI was expected due to the major increase in runoff from RY 2016 to RY 2017. When comparing ABI among individuals from RY 2017 to RY 2018, however, cottonwoods and willows deviate from no change in both directions, meaning some had greater ABI in RY 2017 versus RY 2018 but others had greater ABI in RY 2018 than RY 2017 (Figure 31). Over the past three monitoring years, the general trend among cottonwoods and willows resulted in higher median ABI comparing RY2016 to RY2018 (Figure 32). As expected, more variable conditions in water availability and stream morphology in RY 2017 and RY 2018 likely explain more variation in tree growth response observed in the ABI measurements in RY 2017 to RY 2018. These results are cause for further investigation into smaller-scale geomorphic settings affecting water availability and tree response.

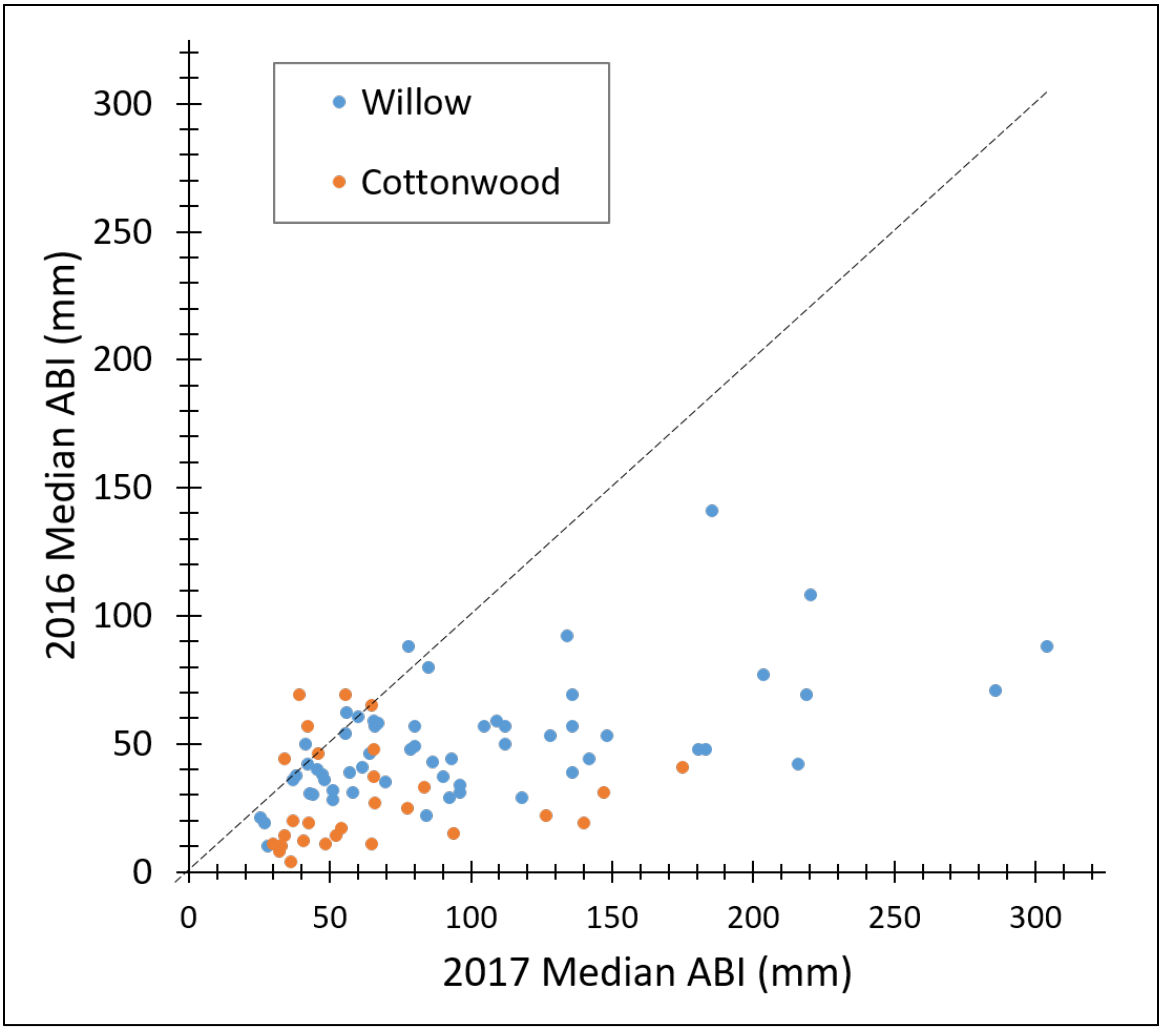

Figure 30. Year-to-year comparison of median ABI for cottonwood (orange) and willow (blue) individuals measured in the Rush Creek Bottomlands, CA between runoff years 2016 and 2017.

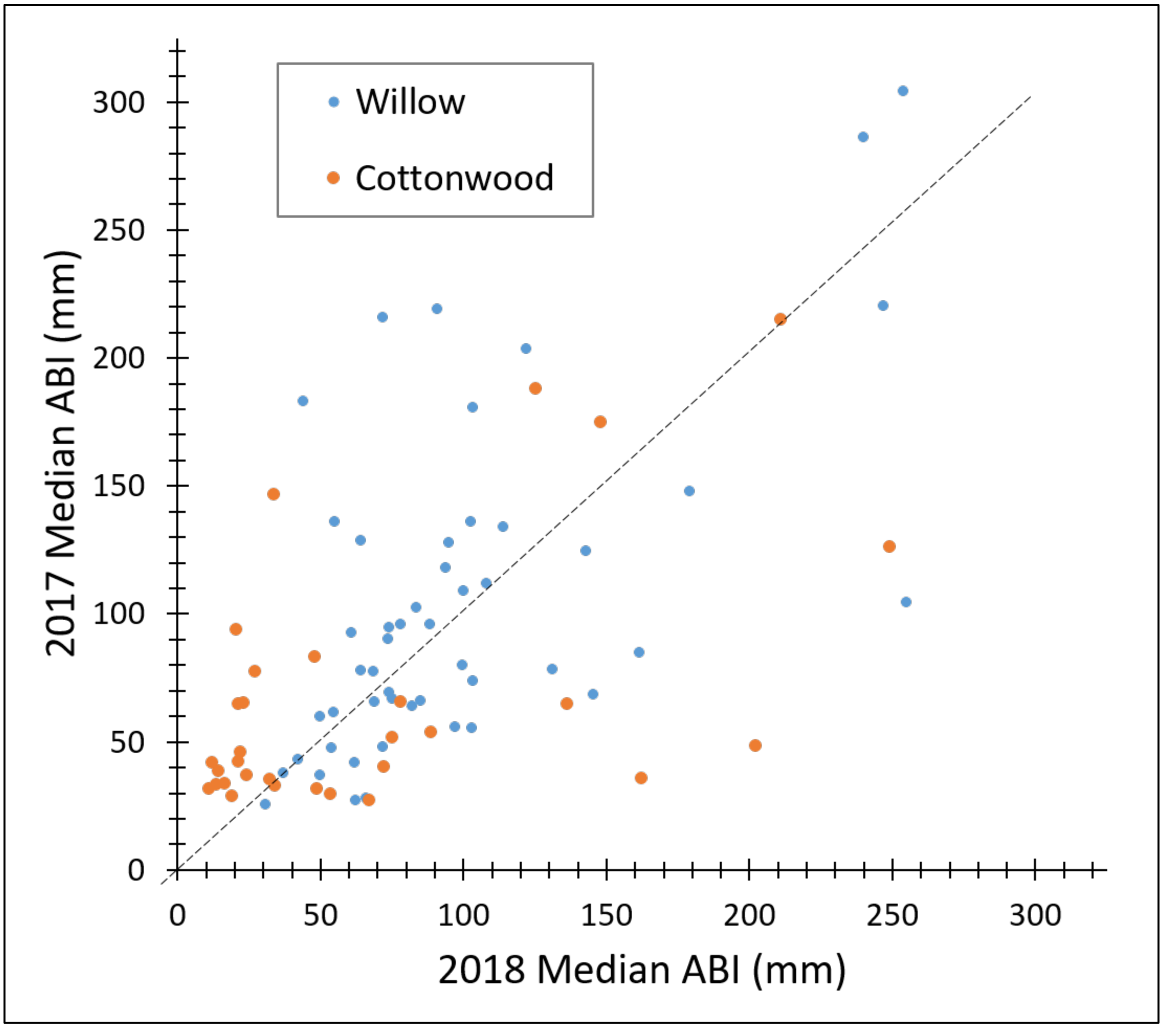

Figure 31. Year-to-year comparison of median ABI for cottonwood (orange) and willow (blue) individuals measured in the Rush Creek Bottomlands between runoff years 2017 and 2018.

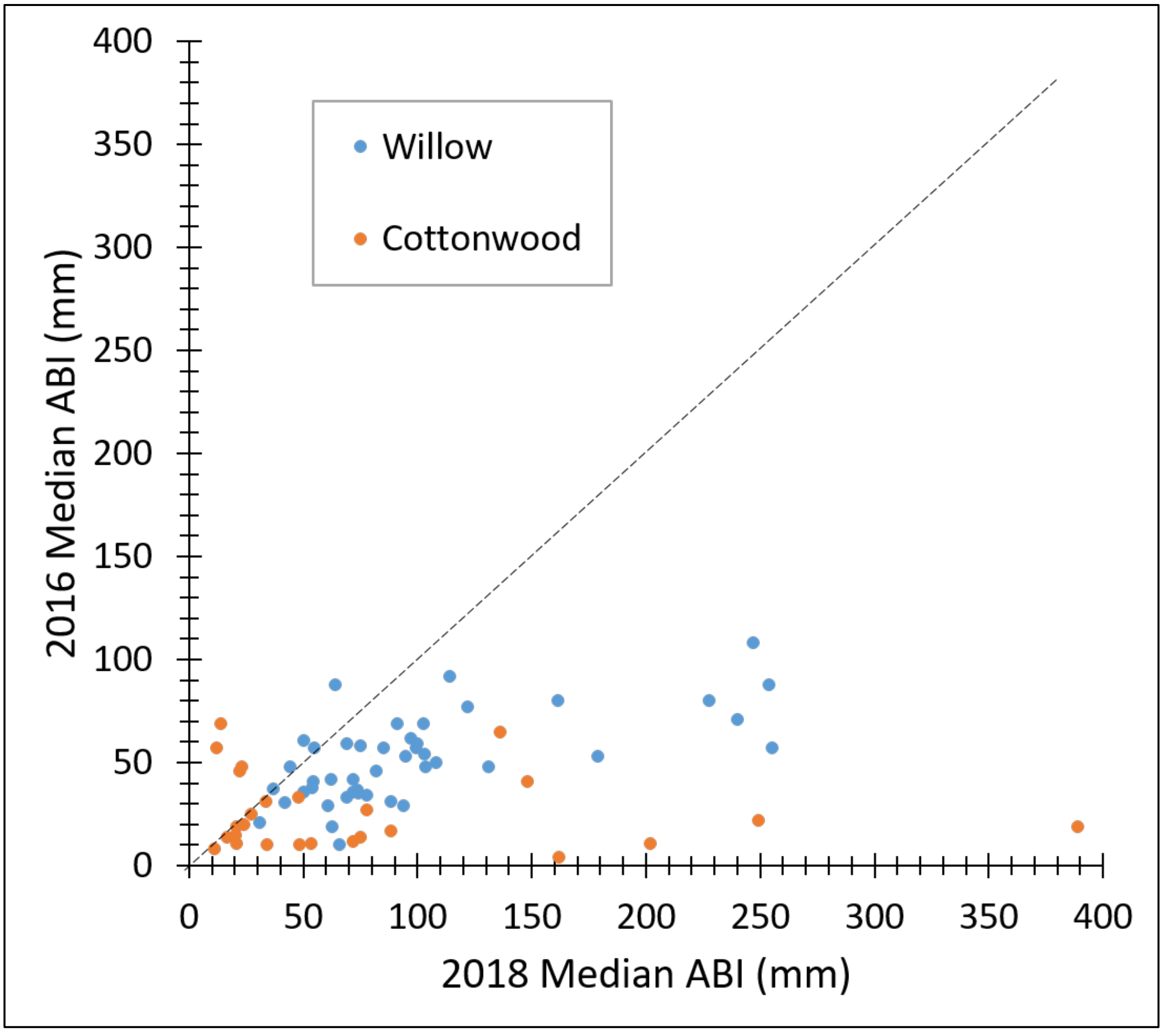

Figure 32. Year-to-year comparison of median ABI for cottonwood (orange) and willow (blue) individuals measured in the Rush Creek Bottomlands between runoff years 2016 and 2018.

NDVI as Tree Vigor

In addition to assessing tree vigor with ABI measurements, we compared changes in median NDVI for individual willows from year to year and found overall similar range but lesser median NDVI values in 2017 than in 2018 (Figure 33). This suggests that there were greater amounts of photosynthesis occurring among willow individuals during the growing season in 2018 compared to 2017, despite a much wetter RY2017. NDVI overall trends in willow tree vigor were opposite, albeit slight, to those year-to-year trends in median ABI from 2017 to 2018. We also evaluated and found no significant correlations between ABI and NDVI measurements, which raised questions regarding the comparability of the two methods for measuring tree vigor in Rush Creek.

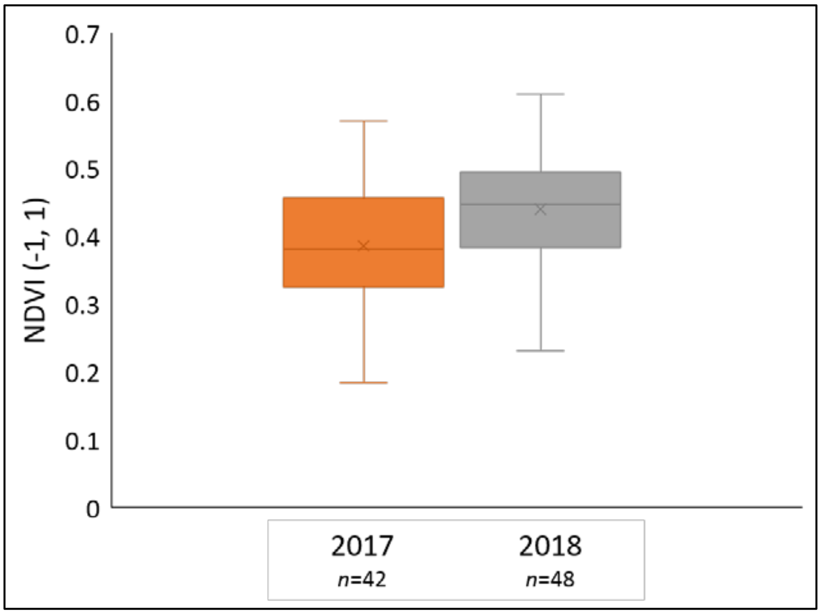

Figure 33. Distribution of median NDVI values of individual willow trees measured in the summer of 2017 and 2018 in the Rush Creek Bottomlands, CA.

There were substantial differences in ABI and NDVI for assessing tree vigor. Annual branch increments captured stem growth representing an entire year's growth, whereas our NDVI data captured an index of photosynthesis relative to leaf density from one moment during the growing season in August. Many mature yellow willows had significant portions of their canopy as dead branches while other portions were bright green. With the high spatial resolution of the UAV NDVI data, dead branches were distinguished from those alive; ABI data, however, didn't include dead branches (i.e., ABI=0). NDVI also only captured the top part of a tree's canopy, whereas ABI data captured branches around a tree's laterally growing branches within arm's reach. Despite their differences, overall trends in assessing tree vigor were expected to reveal similar trends. However, RY2018 findings from ABI and NDVI evaluation of tree vigor were different. For future monitoring, we expect higher temporal resolution of NDVI will reveal a more complete story than ABI findings.

Trees Grouped in Geomorphic Settings

To assess tree vigor in response to different geomorphic settings with potentially differing annual water availabilities, individual trees were grouped according to groves located in various geomorphic settings (Figure 34). We delineated a total of 11 groups, all in which willows occurred and seven in which cottonwoods occurred. The groups were identified based on upstream-downstream location along Rush Creek, riparian or floodplain attributes, and water sources other than surface runoff from precipitation (i.e., springs or inundation from beaver dams) (Figures 35 to 38).

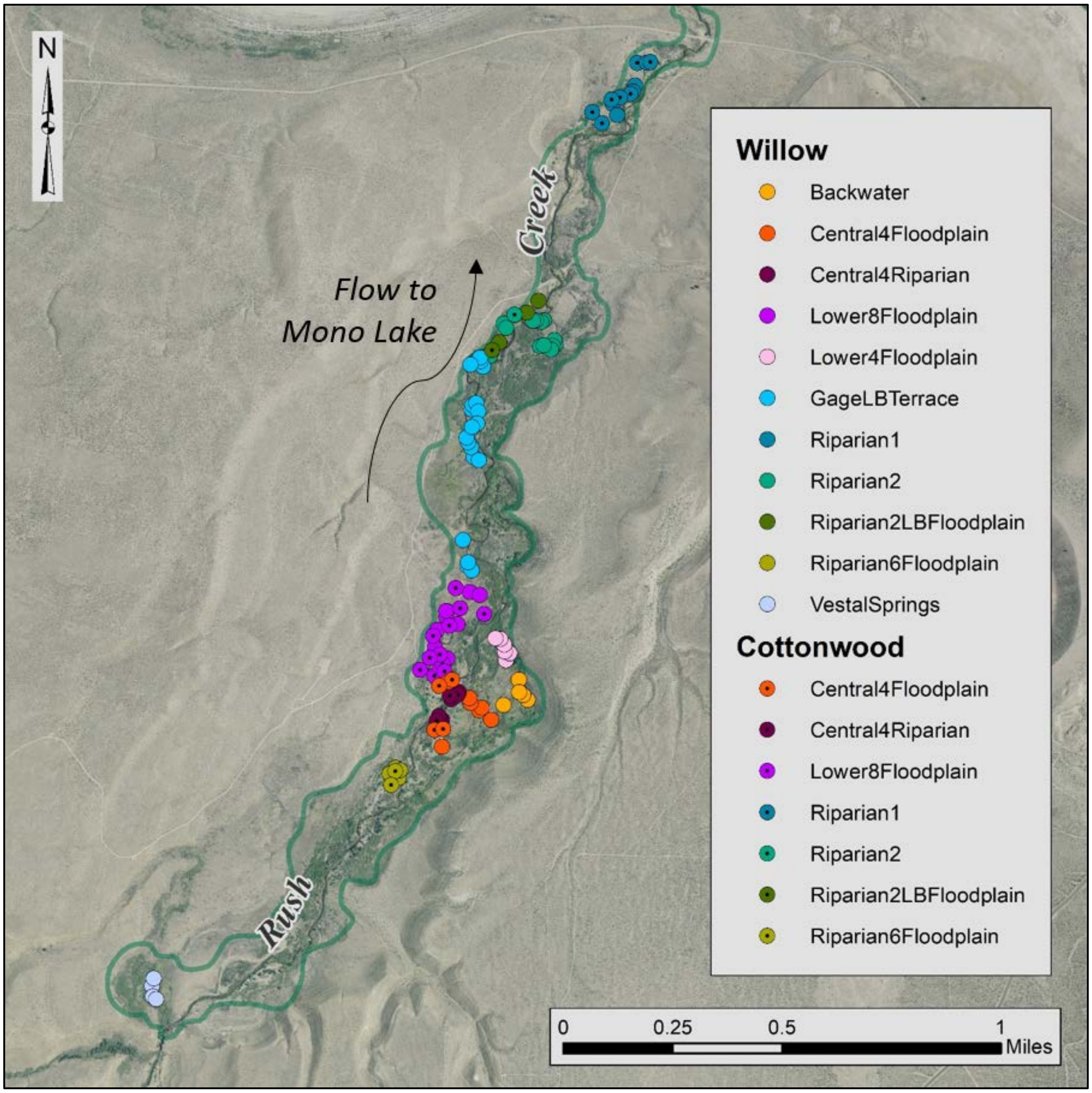

Figure 34. Map of cottonwood and willow trees grouped by geomorphic setting and grove location in the Rush Creek Bottomlands, Mono Basin, CA (ESRI 2019, NAIP imagery 2012).

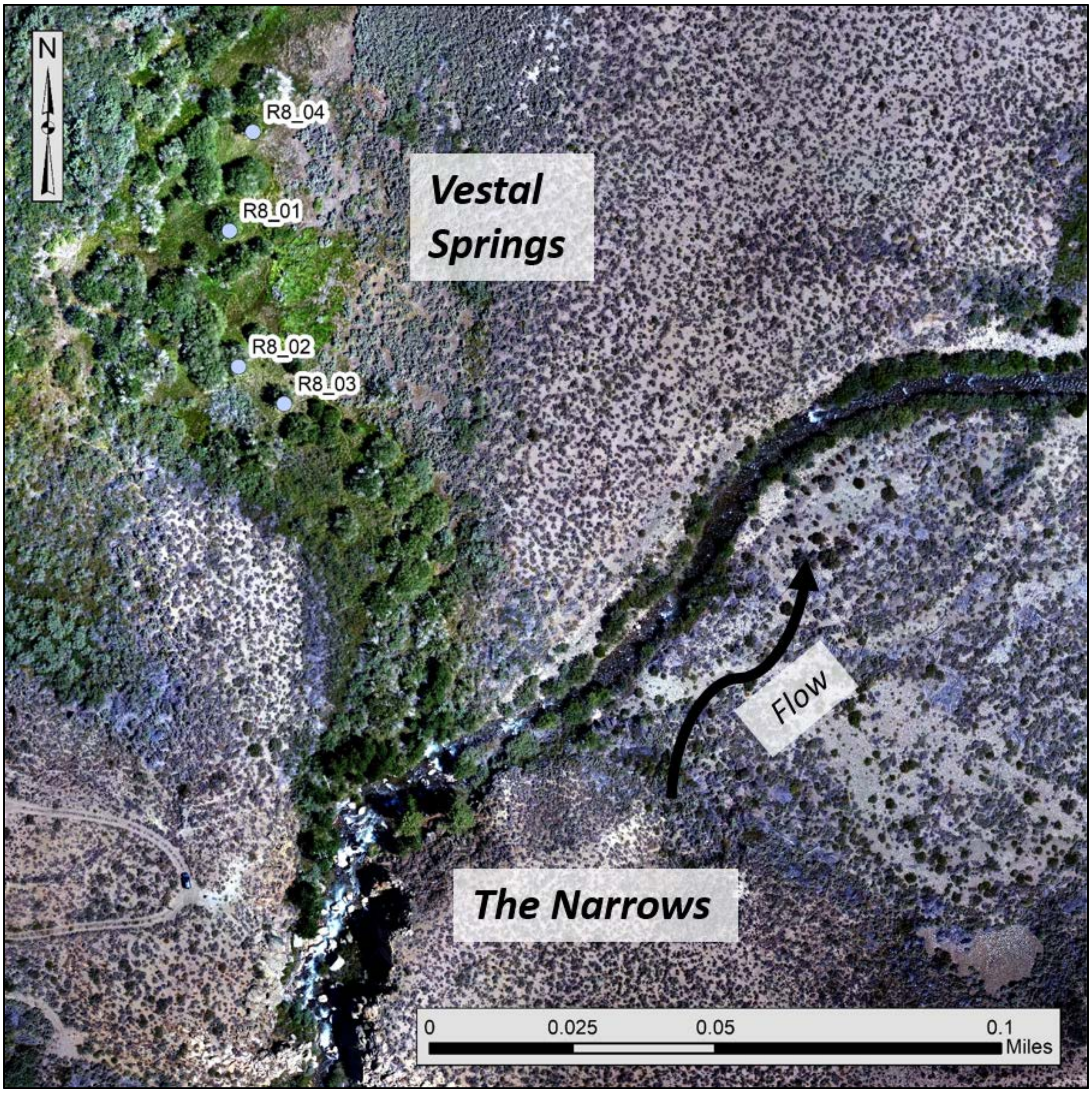

Figure 35. A close-up of willow trees measured among one grouped setting, located in the upper reaches of the Rush Creek Bottomlands, CA (ESRI, 2019, Adair, 2018).

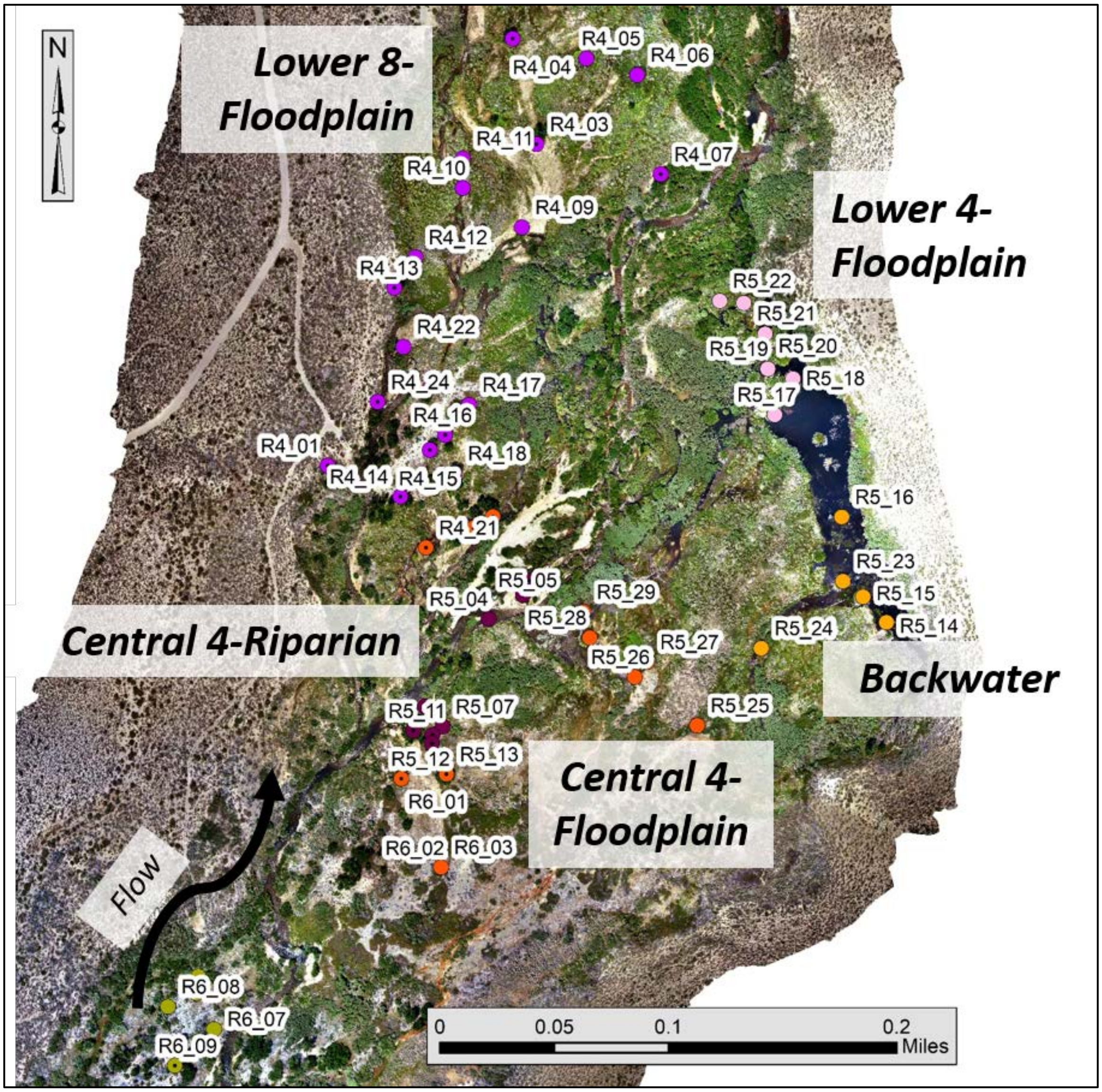

Figure 36. A close-up of cottonwood and willow trees measured among four grouped settings, located in the middle reaches of the Rush Creek Bottomlands, CA (ESRI, 2019, Adair, 2018).

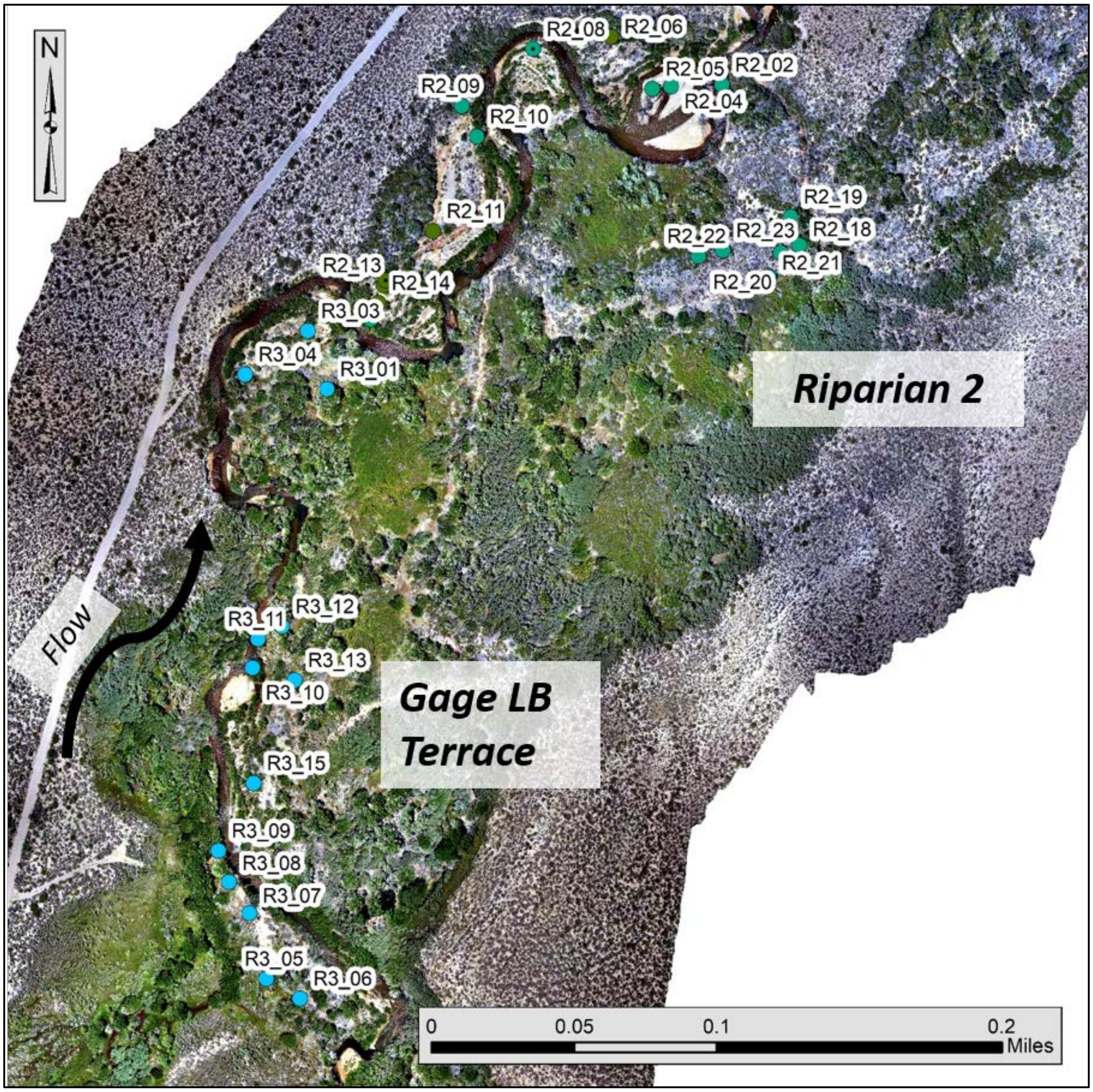

Figure 37. A close-up of cottonwood and willow trees measured among two grouped settings, located in the middle reaches of the Rush Creek Bottomlands, CA (ESRI, 2019, Adair, 2018).

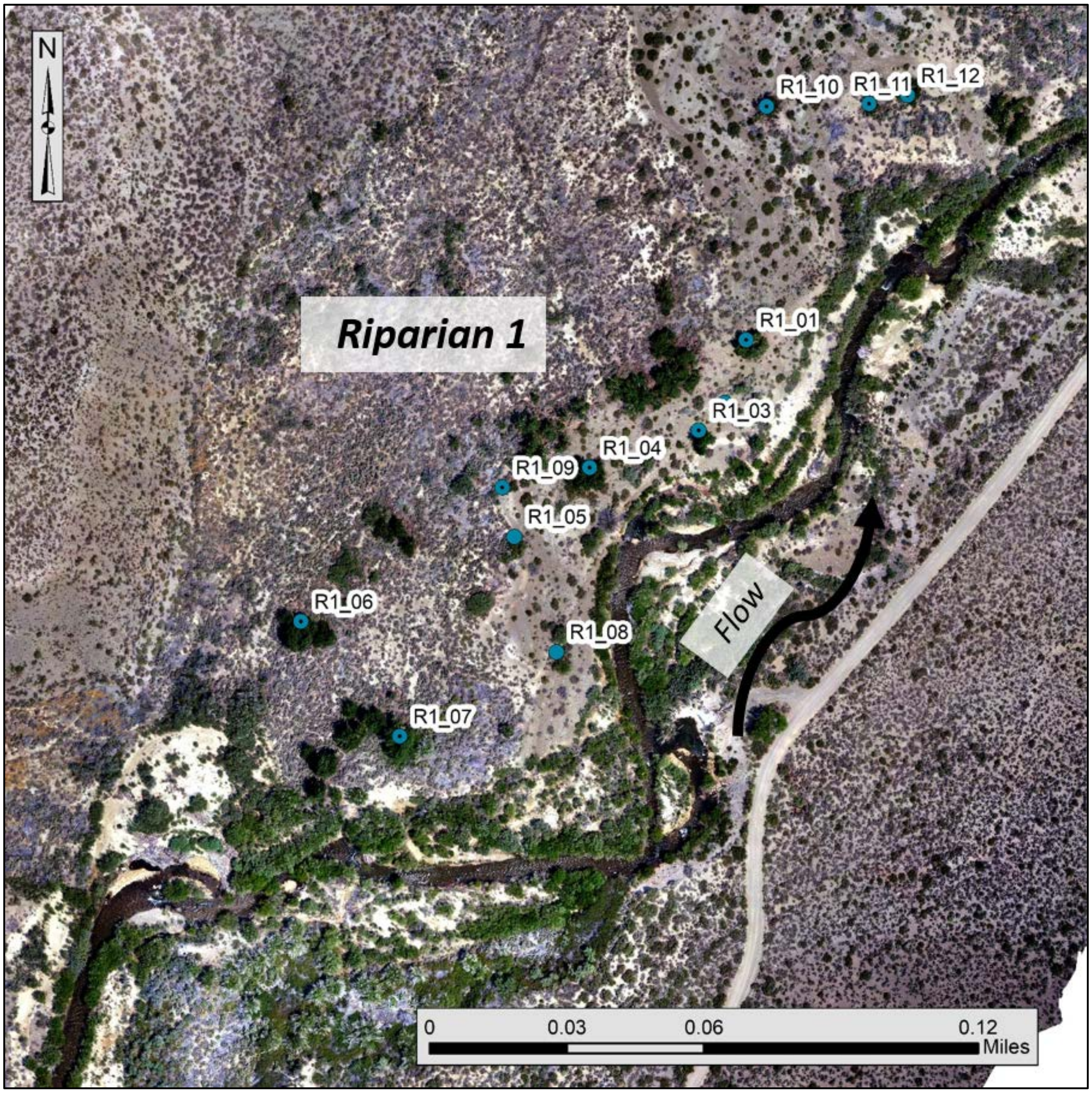

Figure 38. A close-up of cottonwood and willow trees measured among one grouped setting, located in the lower reaches of the Rush Creek Bottomlands, CA (ESRI, 2019, Adair, 2018).

In addition to comparing ABI measurements among individual trees, ABI measurements were pooled among all trees within a group. Pooled ABI within each tree group was compared with RY2018 data (Figure 39). Cumulative ABI among all trees within a group revealed skewed distributions due to outliers with greater ABI lengths, as well as variance among group medians and ranges. The Riparian 2 group had the highest ABI compared to all other groups for cottonwood in RY2018. Lower 8 Floodplain and Central 4 Floodplain cottonwood groups had comparable ABIs with medians approximately 100 mm ABI. Riparian 1, Riparian 2 LB Floodplain, and Riparian 6 Floodplain groups had the lowest ABI among all cottonwood groves, with median ABIs of 22 mm, 23 mm, and 38 mm, respectively. Riparian 2 cottonwoods and willows experienced the highest range in ABI values, with outliers occurring at greater ABI lengths. Willow tree groupings had the greatest median ABI (~120 mm) in the Lower-8 Floodplain and in Riparian 2. These groups also had some of the greatest willow ABI (2300-800 mm), but those greater ABI values were outliers, where both groups' median ABI occurred at ~160 mm. The RY2018 median ABI for willows were closer in range among groups (range 74 – 203 mm) compared to that for cottonwoods (range 23 – 350 mm).

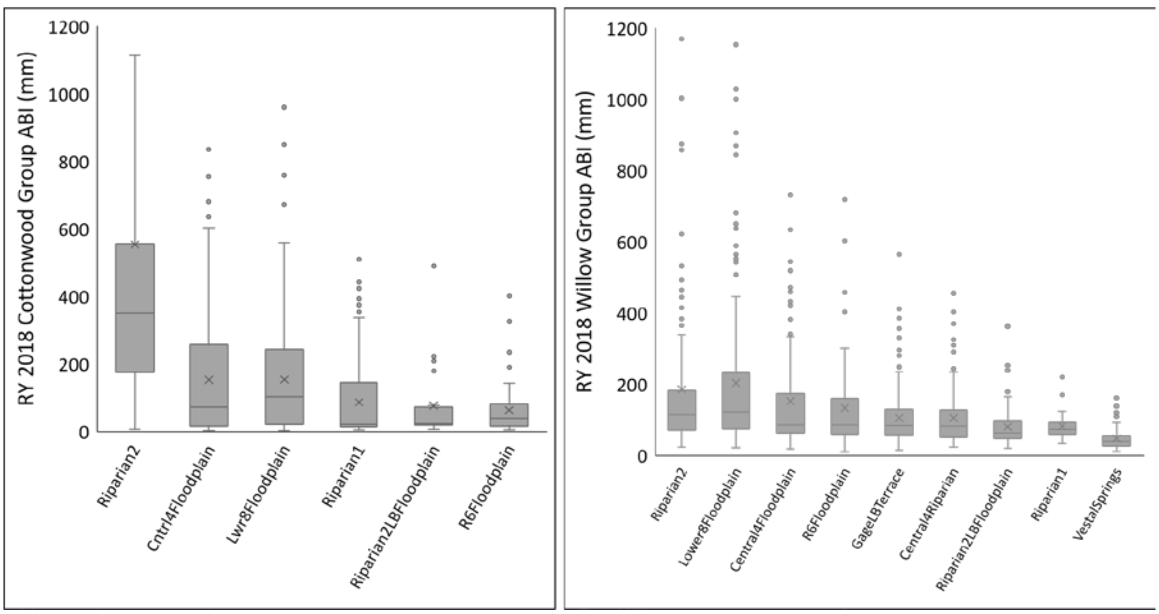

Figure 39. Distribution of ABI among cottonwood and willow groves grouped in different settings in the Rush Creek Bottomlands, CA. Note: Cottonwood ABI in Riparian 2 have outliers extending up to 3,000 mm and willow ABI in Riparian 2 and Lower-8 Floodplain have outliers extending upwards of 2,300 mm.

Overall, we used our grouped results to assess status of tree vigor in different geomorphic settings of the Rush Creek Bottomlands. Trees occurring in Riparian 2, Lower-8 Floodplain, and the Central-4 Floodplain groups experienced the most growth in ABI including the greatest range in ABI compared to other groups. Some individual trees may have been dealing with water stress by responding with a greater range in ABI, which can reflect branch dieback. However, when evaluating

ABI grouped from many trees in one surface-type or geomorphic location, greater variability in ABI may have resulted from more varied conditions such as reactivation of side-channels or prolonged duration of water supply in certain areas. Major channelbed aggradation at the 8 Side-Channel entrance from RY2017 peak flood diverted streamflow from the mainstem and into the 8 Side-Channel, making water highly accessible through the summer to cottonwoods and willows in Lower 8-Floodplain with possible residual effects in RY2018 growing season. The increase not only in water availability from RY2017 and RY2018 but also in the spatial and temporal variability of water availability likely has provided opportunity for growth among certain trees when given the chance. This was likely occurring in those groups of trees where more vigorous growth was observed.

We then compared year-to-year ABI among groups of cottonwoods to detect responses between runoff years (Figure 40). Across all monitoring years for cottonwoods, Riparian 2 had the greatest median ABI, where it decreased by 20% from RY2016 to RY2017 and increased by 2.2 times from RY2017 to RY2018. The Riparian 2 group of cottonwoods also had the greatest range in ABI from 2018 measurements. The Central-4 Floodplain exhibited greatest changes not from RY2016 to RY2017, but from RY2017 to RY2018, where median ABI was highest and range in ABI was much greater. ABI among cottonwoods grouped in Riparian 1 experienced greater range with greater ABI values becoming more frequent with each year. Median ABI among cottonwoods in Riparian 1 increased more than 5 times from 2016 to 2017 and almost halved from 2017 to 2018. The Lower-8 Floodplain had a gradual increase in median ABI over the years as well as an increase in range in ABI. This observation in the Lower-8 Floodplain was likely due to activation of the Lower-8 side-channels during the 2017 flood hydrograph. Converse to increasing trends over time in other groups, cottonwoods grouped in the Riparian 2 LB Floodplain decreased in median ABI with each year.

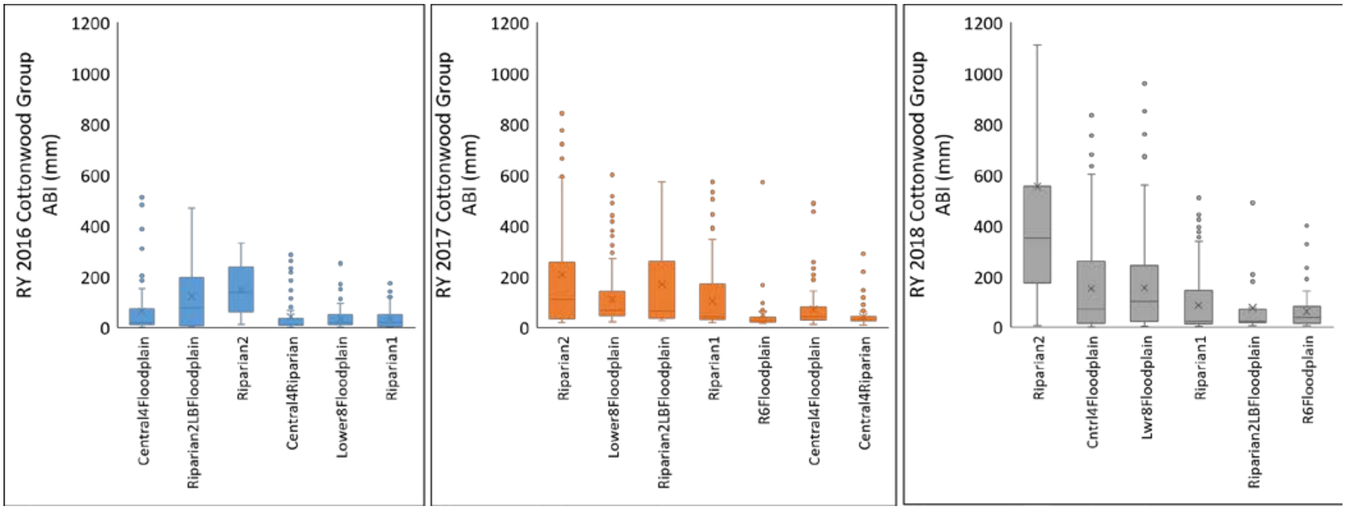

Figure 40. Year to year distribution of greatest to least ABI among cottonwood groves grouped in different settings in the Rush Creek Bottomlands, CA. Note: Scales were made the same for comparison between years. Cottonwood ABI in Riparian 2 have outliers extending up to 3,000 mm in RY 2018.

We then conducted the same analysis on willows, comparing year to year changes in ABI across groups (Figure 41). All willow groups in RY2016 experienced relatively low median ABI with little range or variability in ABI across groups, a clear reflection of the dry runoff year with preceding dry years. In response to the 'Extreme-Wet' runoff year in 2017 and 'Normal' runoff year in 2018, greatest willow growth expressed in median ABI occurred in the Lower-8 Floodplain, followed by willows in the Riparian 2 group. Again, the ABI response observed in the Lower-8 Floodplain was likely from activation of Lower-8 side-channels during RY2017 peak flows. Greater median and ranges in ABI within the Riparian 2 willows in 2017 and 2018 monitoring may be a response to nearby changes in channel morphologies. During the RY2017 flood hydrograph, an overtightened left bend was cut-off, creating an expansive point bar (Figure 17). This shifted streamflows farther over onto the river right side of the mainstem channel, making several willow trees in this group closer to flowing water. Overall, willow median ABI values and ABI variability increased among groups except for the Vestal Springs group, whose trees were expected to respond differently than the rest due to their alternative water source from springs.

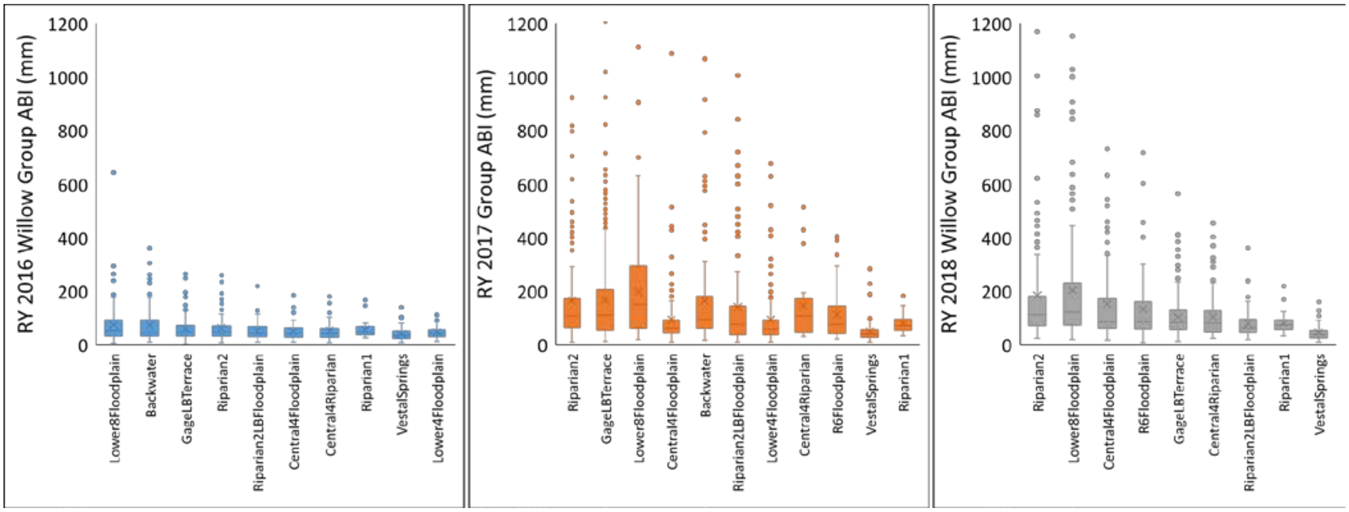

Figure 41. Year to year distribution of greatest to least ABI among willow groves grouped in different settings in the Rush Creek Bottomlands, CA. Note: Scales were made the same for comparison between years. Willow ABI in Riparian 2 and Lower-8 Floodplain in RY 2018 have outliers extending upwards of 2,300 mm.

Overall impressions on evaluating riparian tree vigor in the Rush Creek Bottomlands were that willows engender less variability than cottonwoods. Our data showed willows were more closely related to water availability, whereas cottonwoods showed less predictable responses to greater water availability. Some studies show that cottonwoods have a threshold for water availability, where they can negatively respond when the shallow groundwater table is too high. Cottonwoods in certain areas of our study site displayed more variable responses to water availability than willows, which may be explained by unfavorable high water conditions. The results of our study show the resiliency of cottonwoods in response to water stress with variable ABI in RY2016. That is not to say there was harm from higher water years in RY2017 and RY2018, however, because overall those wetter water years resulted in more vigorous cottonwood groups and individuals.

As anticipated, the Vestal Springs group acted somewhat of a "baseline" condition for sampled willow trees throughout the Bottomlands due to the continuous freshwater source feeding-in from Vestal Springs. Year-to-year ABI measurements among four trees that comprised the Vestal Springs group were compared (Figure 42). The range of ABI values was similar in R8_01 and R8_02 across RY2017 and 2018. Tree R8_04 had similar ABI values in RY2016 and RY2017, but those were greater than in RY2018. Conversely, tree R8_03 had higher ABI values in RY2018 compared to 2017, but ABI in RY2017 were similar to those in RY2016. Trees in the Vestal Springs grove responded without detectable correlations to different water years. Overall, median values among individual willows in Vestal Springs maintained very close median ABI values over the years. These tree responses were likely due to the small changes in spring groundwater inputs that provided more consistent water for trees in Vestal Springs compared to trees in other areas of the Rush Creek Bottomlands relying solely on either surface runoff, or a mixture of surface runoff and groundwater storage dictated by Rush Creek's stream flow. Unfortunately, Parshall flumes installed among the Vestal Springs were no longer functional. If those flumes were still functional, we would have a better quantification of annually variable springflows.

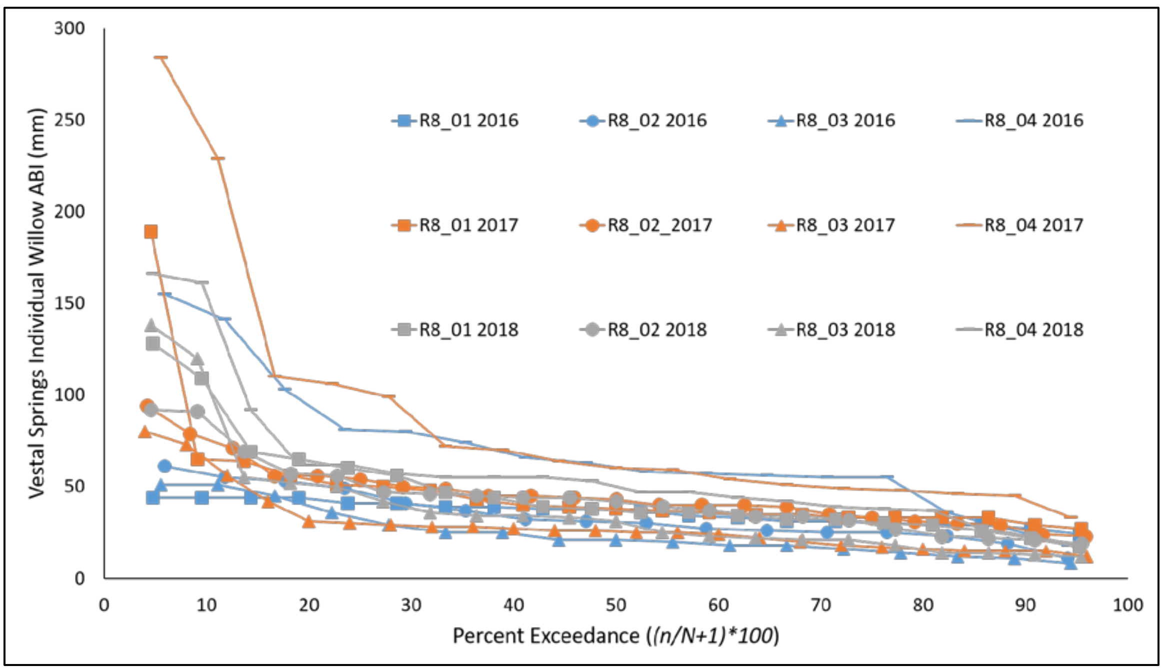

Figure 42. Comparison of 2016 (blue), 2017 (orange), and 2018 (grey) distribution of ABI among individual trees in Vestal Springs, Rush Creek Bottomlands.

Some individual trees became inundated after RY2017 peak flows due to beaver colonization. We hypothesized that such conditions were unsuitable for tree vigor and expected less vigorous response in ABI measurements in RY2018. However, we were unable to survey this area during 2018 fieldwork due to increased wasp presence. The trees inundated from beaver dams appeared to have more yellowing in the leaves, compared to greener, healthier looking trees in other areas.

Approximated Tree Age Related to Vigor

We expected younger riparian trees to exhibit more vigorous growth. As a surrogate measure of tree age, tree base circumference and primary stem diameter were related to tree vigor measurements (ABI) among cottonwood and willow trees. There was no significant relationship between willow ABI and base circumference nor with primary stem diameters. For cottonwoods, there were some younger trees with smaller bases associated with greater median ABI, but this relationship had no analytical significance.

Willows and cottonwoods in Rush Creek may have different rates of growth and water uptake throughout their life cycles, but annual monitoring was not designed to measure either rate. Measuring willow tree bases was difficult due to often being buried with sediment and due to limited access. Many older cottonwoods in the Rush Creek Bottomlands had significant branch sacrifice, where branches of a tree die back and stem growth was directed to younger branches of a tree. The age surrogate variables were expected to highlight 'young' versus 'old' trees and differences in ABI between the two age classes. Refinements to this portion of the analysis in the future may reveal a closer relationship with ABI.

Conclusions and Recommendations for Future Monitoring

Summary of Findings

Monitoring in the Rush Creek Bottomlands during RY2016, RY2017, and RY2018 provided a unique opportunity for assessing spatial and temporal changes in channel morphology and riparian tree vigor. Changes in channel morphologies were documented by monitoring active channel widths, residual pool/run depths, stream channel cross sections, and high-resolution UAV imagery. Monitoring changes in active channel width proved most informative by measuring those widths at each riffle crest opposed to measuring at equidistant, random features. Active channel widths in RY2018 were significantly wider than in RY2016, a result of the flood hydrograph in RY2017. From RY2016 to RY2018, residual pool depths were greater in the upper monitoring reach but did not change in the lower monitoring reach. Cross sections in lower reaches of Rush Creek revealed slight lowering of channelbed elevation, but there was no significant change in elevation at the bed hydraulic control feature nearest those cross sections. Cross sections in the upper reaches of Rush Creek experienced signs of channel straightening and greater change in channelbed elevation compared to the lower reach cross sections. The upper reach was likely more susceptible to scour and erosion from RY2017 peak flows due to the confined nature of the channel compared to the wider, more sinuous channel shape of the lower reach.

Riparian cottonwoods and willows in the Rush Creek Bottomlands were monitored for vigor in units of annual branch increments (ABI) and in units of NDVI from remotely sensed data. Annual tree responses were assessed among individual trees and among trees grouped into geomorphic settings. Both cottonwoods and willows experienced greater median and range in ABI during RY2017 and RY2018 compared to RY2016. Less dramatic differences in ABI observed from RY2017 to RY2018 were expected given the change from an 'Extreme-Wet' runoff year to a 'Normal' runoff year, and possibly due to residual effects in water availability from RY2017. NDVI overall trends in willow tree vigor are opposite, albeit slight, to those year to year trends in median ABI from 2017 to 2018. There were no significant correlations between ABI and NDVI measurements, likely because the two methods capture tree vigor at spatial and temporal scales too dissimilar for comparison. While all groups of cottonwoods and willows responded more positively to RY2017 and RY2018 compared to RY2016, some groups were more vigorous than others. The more vigorous groups of trees were likely responding to changes in channel morphologies in the mainstem of Rush Creek as well as to side-channel activation. Geomorphic changes and beaver activity in the now inundated Lower-4 Floodplain complicated trends in tree vigor. Those inundated trees appeared to respond negatively to such conditions but were physically inaccessible in RY2018. Three years of monitoring willows near Vestal Springs have provided a 'baseline' condition for referencing other willows' response to water availability in the Bottomlands. Overall, using ABI to assess annual changes in riparian tree vigor has proven a useful monitoring tool in lower Rush Creek.

Future Monitoring Recommendations

The methodology for measuring the ABI's in the Rush Creek Bottomlands was adapted from the methods used by Willms et al. (1998), which analyzed the response of different cottonwood species to water availability. Willms et al. (1998) found that there was a significant correlation between ABI growth and streamflow. Because our monitoring approach comes from a goal of simplicity, the approach adopted by Willms et al. (1998) to measuring ABI was slightly different than ours. After Willms et al. selected individual trees for their study, they selected specific branches where they could measure a minimum of 10 years of ABI growth using terminal bud scars along a stem. This enabled collection of ABI measurements for the previous 10 years in one field season. We took these findings and attempted to utilize this approach more simply as a monitoring tool for present water years, thus the need for annual visits.

While previous years' ABIs were easily identified among cottonwood trees, we have found this less obvious on willow branches. Our approach to measuring cottonwood and willow tree vigor with ABI measurements has thus far included return visits each season to specified trees but on randomly selected branches of those trees. Our approach does show correlation to water year, namely observed among overall greater ABI values in response to the 'Extreme-Wet' RY2017. However, our simplified approach has resulted in a wide range of values that might not sufficiently detect the finer nuances of a water year's impact on riparian tree vigor. One recommendation for mediating this issue would be to repeat our methods for RY2019 monitoring in addition to selecting specific branches from trees and measuring the ABI growth from past years as far back as possible, as done in the Wilms et al. (1998) study. This would give us two sets of ABI data from each specific tree, whose variance could then be compared. We hypothesize that the method with less variance in ABI values would serve to be a better, more representative dataset for measuring a given tree's true ABI and vigor.

Another alternate approach for better detecting ABI correlation with stream flow runoff would be to survey these riparian trees earlier in the season. Willms et al. (1998) analyzed their ABI data by examining monthly stream flow averages and determined that the ABIs had a much stronger correlation to the streamflow between January and May. They note that late-spring and especially summer flows were expected to more closely correlate to ABI growth, but stronger correlation observed with earlier flows "suggest that: (i) the recharge of the riparian water table during the high flow spring period is important for subsequent cottonwood growth and (ii) cottonwood branch growth may especially occur early in the growing season." If cottonwoods respond more sensitively to snowmelt runoff, perhaps our monitoring efforts should include ABI measurements during late-spring and summer in addition to fall. We would anticipate observing initial growth of the season being more closely related to water year type than observed later in the season when less vigorous branches or stems of trees have time to 'catch-up' throughout the summer with remaining duration of water supply. Data collection during late-spring, soon after the peak flow event, as well as during fall at the end of the growing season (as done for this study in previous years) would provide comparable datasets for tree response to water availability on a finer time scale.

Literature Cited

Adair, J. 2018. 5-7 August. Rush Creek Unmanned Aerial Vehicle high resolution imagery. Retrieved from http://gis.humboldt.edu/ISAMM/Projects/RushCreek/Map.html.

Lenney, M. P., Woodcock, C. E., Collins, J. B., & Hamdi, H. 1996. The status of agricultural lands in Egypt: the use of multi-temporal NDVI features derived from Landsat TM. *Remote Sensing of Environment*, *56*(1), 8-20.

Lisle, T.E. 1987. Using "residual depths" to monitor pool depths independently of discharge Res. Note PSW-RN-394. Berkeley, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest and Range Experiment Station. 4 p.

Los Angeles Department of Water and Power. 2016. Mono Basin Operations, Fisheries Monitoring, Stream Monitoring, and Waterfowl Habitat and Population Monitoring. Section 4. HSU River Institute RY2015 Monitoring Report. May 2016. 17 p.

NASA. 2000. NASA Earth Observatory Measuring Vegetation (NDVI & EVI). Accessed 2019. https://earthobservatory.nasa.gov/features/MeasuringVegetation/measuring_vegetation_2.php

Peters, A. J., Walter-Shea, E. A., Ji, L., Vina, A., Hayes, M., & Svoboda, M. D. 2002. Drought monitoring with NDVI-based standardized vegetation index. *Photogrammetric Engineering and Remote Sensing*, *68*(1), 71-75

Pettorelli, N., Vik, J. O., Mysterud, A., Gaillard, J. M., Tucker, C. J., & Stenseth, N. C. (2005). Using the satellite-derived NDVI to assess ecological responses to environmental change. *Trends in Ecology & Evolution*, *20*(9), 503-510.

Reed, B. C., Brown, J. F., VanderZee, D., Loveland, T. R., Merchant, J. W., & Ohlen, D. O. 1994. Measuring phenological variability from satellite imagery. *Journal of Vegetation Science*, *5*(5), 703-714.

Trush, W. J., Graham, J., Adair, J., & Cooper, E. J. 2018. RY2017 Mono Basin Stream Monitoring Report (Rep.). Los Angeles, CA: Los Angeles Department of Water and Power.

Trush, W. J. 2017. RY2016 Geomorphic and Riparian Monitoring Report (Rep.). Los Angeles, CA: Los Angeles Department of Water and Power.

McBain and Trush, Inc. and Ross Taylor and Associates. 2010. Synthesis of instream flow recommendations to the State Water Resources Control Board and the Los Angeles Department of Water and Power. April 30, 2010. 159 p.

Williams, C. A., & Cooper, D. J. 2005, June 28. Mechanisms of Riparian Cottonwood Decline along Regulated Rivers. Retrieved from https://link.springer.com/article/10.1007/s10021-003-0072-9.

Willms, J., Rood, S. B., Willms, W., & Tyree, M. 1998. Branch growth of riparian cottonwoods: A hydrologically sensitive dendrochronological tool. *Trees,12*(4), 215. doi:10.1007/s004680050143.

Section 5

Mono Basin Waterfowl Habitat Restoration Program 2018 Monitoring Report with Recommendations by Ms. Debbie House, Interim Mono Basin Waterfowl Monitoring Program Director

Mono Basin Waterfowl Habitat Restoration Program

2018 Monitoring Report

Prepared for the State Water Resources Control Board

The Los Angeles Department of Water and Power (LADWP) is conducting monitoring in compliance with the 1996 Mono Basin Waterfowl Habitat Restoration Plan (Plan) (LADWP 1996) and State Water Resources Control Board Order WR 98-05 (SWRCB 1998). Monitoring conducted in 2018 by LADWP included:

- Monthly Mono Lake elevation readings
- Daily stream flows in Rush, Lee Vining, Parker and Walker Creeks
- Lake limnology including meteorological, physical/chemical, phytoplankton, and brine shrimp population monitoring
- Summer waterfowl ground surveys and documentation of habitat use
- Fall aerial waterfowl surveys at Mono Lake, Bridgeport Reservoir and Crowley Reservoir
- Still-image photography of waterfowl habitats at Mono Lake, Bridgeport Reservoir and Crowley Reservoir
- Surveillance for saltcedar (*Tamarix* spp.)

The *Mono Basin Waterfowl Habitat Restoration Program 2018 Monitoring Report* summarizes the results of this monitoring. This report also includes a reevaluation of recommendations presented in the *Mono Basin Waterfowl Habitat Restoration – 2017 Compliance and Periodic Overview Report* (*Periodic Overview Report*) regarding modifications to the limnology and waterfowl population monitoring programs.

Hydrological Summary

The 2018 runoff year in the Mono Basin (April 1, 2018 - March 31, 2019) was a “Normal” year type at 108,164 acre-feet or 89% of average runoff. Since implementation of the Plan in 1998, fluctuations in lake level have occurred primarily due to variation in water years, and Mono Lake has experienced four periods of increasing elevation, and three subsequent decreases. 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 Decision 1631 placed limitations on water exports from the Mono Basin by the City of Los Angeles. Since October 2016, the elevation of Mono Lake increased in response to the extreme wet year of 2016-2017 to its most recent maximum in June 2018 of 6,381.8 feet.

Limnology Summary

The 2018 monitoring year marked the 2nd year of the 5th meromictic event since the beginning of the program. The existing chemocline persisted at depths between 8 m and 15 m with the salinity gradient peaking at 16.8 g/L in July. The gradient decreased to 6.8 g/L at the end of the year. The hypolimnion remained mostly anoxic, and ammonium was deprived from the epilimnion and accumulated in the hypolimnion. The *Artemia* population decreased from 15,158 m^{-2} in 2017 to 12,120 m^{-2} in 2018. The value was higher than the record low of 7,676 m^{-2} in 2015, but still remained far below the long term average of 18,951 m^{-2}. Due to an average influx of freshwater and decreased *Artemia* population, clarity of the lake was not as good as in 2017. For the third year in row, the centroid (the calculated center of abundance of adults) remained above 220 days reversing the long term declining trend.

Future limnological condition of Mono Lake will largely depend on future runoff conditions. A lack of sustained high freshwater input or extremely large freshwater input will result in higher salinities. A prolonged wet period is necessary to lower the hypolimnetic salinity. Prolonged drought could have an opposite effect as seen between 2012 and 2016 during which salinity increased from 78.7 g/L to 97.5 g/L. The adverse effect of drought on salinity could become more severe as a drier and warmer climate is predicted for much of California in the future. The *Artemia* population in Mono Lake appears to survive and thrive in the salinity levels during monitoring years. However, further decline in the lake level could result in much higher salinity, which could approach the tolerance level of *Artemia*.

Vegetation Summary

The most significant change the recent increase in lake elevation has had is to restore the connectivity of existing ponds with the water line and spring outflow areas of Mono Lake. The increased connectivity of shoreline ponds with the shoreline and spring outflow areas results in improved habitat quality for waterfowl. Although the increased lake elevation observed from 2017-2018 resulted in improved habitat conditions, the highest recent elevation of 6382.1 feet in June 2018 did not result in more shoreline ponds, which are also an important waterfowl habitat component.

Waterfowl Summary

The breeding waterfowl community has continued to demonstrate a positive response to the primary restoration objective of increasing the level of Mono Lake. In 2018, the breeding waterfowl population showed signs of recovery following the extended drought from 2012-2016 as total breeding waterfowl and brood numbers were at their highest since 2012. Although runoff year 2016-2017 was an extreme wet year, the lake elevation was still low during the summer of 2017, and breeding waterfowl populations still depressed. There may be a lower threshold of lake elevation below which changes in the breeding habitat become more significant. This lower threshold appears to be around 6,382 feet, as below that elevation, all waterfowl breeding parameters have shown a decline.

Fall waterfowl use at Mono Lake in 2018 was extremely low and well below the long-term average. The total fall count in 2018 was 8,732 as compared to the 2002-2019 average of 25,434 +/-2,892 SE. The reasons for the decrease in numbers at Mono Lake in 2018 are not clear, however similar decreases were not observed at either Bridgeport or Crowley Reservoir. These results suggest waterfowl may have been responding to conditions at Mono Lake.

No direct correlation has been found between total fall waterfowl and fall lake elevation. *Artemia* cyst production in fall provides a partial explanation of the annual variations in waterfowl populations at Mono Lake. This relationship may be direct, as the diet of some waterfowl species using saline lakes has been found to consist largely of *Artemia* cysts, or indirect, reflecting other factors. Multiple factors influence migrating populations, including conditions elsewhere in the flyway, productivity on breeding grounds, habitat conditions enroute, weather, and disease. Lacking a clear understanding of factors influencing fall waterfowl populations at Mono Lake, and population turnover rate, the reasons for declining or low numbers cannot be stated with any certainty.

Recommendations

The time period for restoration of waterfowl habitat in the Mono Basin has been greatly extended due primarily to the protracted time period that has been required for lake elevation recovery. In light of this, recommendations for a less-frequent but more focused approach for the long-term monitoring of waterfowl habitat in the Mono Basin were proposed in the 2017 *Periodic Overview Report*. Recommendations for the limnology and waterfowl population monitoring put forth in the *Periodic Overview Report* were reevaluated during the preparation of this report, and a summary of proposed changes is provided.

I welcome discussion regarding proposed modifications to the monitoring program.

Debbie House

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Debbie House

Interim Mono Basin Waterfowl Monitoring Program Director

Los Angeles Department of Water and Power (LADWP). 1996. *Mono Basin Waterfowl Habitat Restoration Plan.* Prepared for the State Water Resources Control Board. In response to Mono Lake Basin Water Right Decision 1631.

Los Angeles Department of Water and Power (LADWP). 2000. *Mono Basin Implementation Plan*. To comply with State Water Resources Control Board Decision 1631 and Order No. 98-05 and 98-07.

State Water Resources Control Board (SWRCB). 1998. Order WR 98-05. Order requiring stream and waterfowl habitat restoration measures. September 2, 1998.

Mono Basin Waterfowl Habitat Restoration Program 2018 Monitoring Report

Prepared by:
Deborah House, Mono Basin Waterfowl Program Director
Motoshi Honda, Watershed Resources Specialist
Los Angeles Department of Water and Power
Bishop, CA 93514

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

May 2019

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EXECUTIVE SUMMARY

In 1983, National Audubon Society v. Superior Court (1983) 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 basin. Order WR 98-05 (SWRCB 1995) 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 2018 under the Mono Basin Waterfowl Habitat Restoration Plan (Plan) (LADWP 1996) as required by Order 98-05.

Although the restoration of waterfowl habitat is not yet complete, LADWP's compliance with Decision 1631 and Order 98-05 has resulted in ecological benefits for the Mono Basin. In addition, the monitoring programs provide a dataset from which ecological trends in the Mono Basin can be evaluated.

Significant restoration accomplished in the Mono Basin has included the reestablishment of perennial flows in Rush Creek, Lee Vining Creek, Parker Creek and Walker Creek. In Rush Creek, all channel openings required under the Plan have also been completed. Outstanding issues are the continued financial assistance available from LADWP for the purpose of waterfowl habitat improvement at the County Ponds or Black Point, and the recovery of Mono Lake to the average target lake elevation of 6,392 feet.

Climatic factors may be influencing Mono Lake and its recovery. Mono Lake has not yet reached the target lake elevation, even though initial modeling predicted this restoration objective would be met in approximately 20 years (or by 2015). From 1998 to 2018, Mono Lake has experienced five periods of increasing elevation including the recent increase beginning in 2017, and three subsequent decreases, through a total elevation range of 8.0 feet and fluctuations in lake level have occurred primarily due to variation in water years. 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. Based on an assessment of runoff data dating back to the late 1930's, it appears that dry years are becoming drier in recent history, thus inhibiting the recovery to the target level. Weather data indicate that since 1995, the summer minimum temperatures have been above their long-term average. More recently, the winter minimum temperatures have also shown a trend of being above their long-term average.

The *Artemia* population is greatly influenced by the Mono Lake mixing regime. The mixing regime of Mono Lake is driven by the amount of freshwater input, and above-normal runoff years result in a stratification of the lake (meromixis). Since 1995, there have been five meromictic events, the latest commencing following the extreme wet year of 2017. The 2018 monitoring year marked the 2^{nd} year of the 5^{th} meromictic event since the beginning of the program. During years of meromixis, nutrients accumulate in the hypolimnion. During periods of below normal runoff, meromixis breaks down, the lake turns over, and the nutrients become available throughout the water column. *Artemia* populations have demonstrated a response to this breakdown of meromixis with population peaks during the year following the breakdown of meromixis. The magnitude of these peaks has been positively correlated with duration of meromixis. The last two meromictic events, which only lasted 1 to 2 years, have resulted in smaller peaks.

There has been a clear temporal shift in peak abundance of instar and adult *Artemia* as monthly peaks are occurring earlier in the year. This pattern has weakened somewhat in the last two years with respect to adult shrimp abundance.

As is typical of closed basin systems, the salinity of Mono Lake increases with decreases in lake volume or inputs. Salinity has been demonstrated to adversely affect the survival, growth, reproduction, and cyst hatching of *Artemia*. Five years of drought between 2012 and 2016 resulted in the lake level declining from 6,383.6 feet in April 2012 to a low of 6,376.8 feet in October 2016, and an increase in salinity from 75.7 g/L in 2012 to 96.6 g/L in 2017. During this period of increasing salinity, the abundance of *Artemia* also declined. In 2017, with the second largest input on record into Mono Lake, salinity decreased to 80.9 g/L by September, and the *Artemia* population showed some recovery. Thus, despite the observed fluctuation in salinity observed, the *Artemia* population has shown some resiliency.

As compared to conditions at the end of the drought in 2016, the most recent increase in lake level up to a maximum in 2018 of 6381.8 feet improved waterfowl habitat conditions somewhat. The most notable effect the recent increase in lake elevation has had is to restore the connectivity of existing ponds with the water line and spring outflow areas of Mono Lake. The number of open water lake-fringing ponds did not appear to change significantly, however as this elevation may be below an elevation threshold needed for the formation of additional ponds.

Long-term monitoring has demonstrated that breeding waterfowl respond positively to the primary waterfowl habitat restoration objective of increasing the level of Mono Lake. In 2018, the breeding waterfowl population showed signs of recovery following the extended drought from 2012-2016 that resulted in a seven foot drop in lake elevation. Although runoff year 2016-2017 was an extreme wet year resulting in increased runoff and rising lake levels in 2017, the lake elevation was still low during the summer of 2017, and breeding waterfowl populations still depressed. There may be a lower threshold of lake elevation below which changes in the breeding habitat become more significant as shoreline ponds dry and wetland habitats become increasingly disconnected from the shoreline. Current data suggests this lower threshold appears to be around 6,382 feet, as below that elevation, all waterfowl breeding parameters have shown a decline.

Fall waterfowl use at Mono Lake in 2018 was extremely low and well below the long-term average. The total fall waterfowl observed at Mono Lake in 2018 was 8,732 as compared to the long-term total mean of 25,434. Peak numbers have averaged 7,941, ranging from a low of 1,826 in 2018 to a high of 17,844. The population estimator, which is the most conservative estimate of annual fall waterfowl use, indicates that an average of 9,210 waterfowl (range 2,148-18,590) visit Mono Lake each fall. There has been a downward trend in total fall waterfowl use at Mono Lake over the 2002-2018 period.

The two key waterfowl species at Mono Lake are the dabbling duck Northern Shoveler and the diver Ruddy Duck, which generally comprise over 80% of total fall waterfowl numbers. In 2018, significant decreases were observed in the fall populations of Northern Shoveler and Ruddy Duck. The reasons for the decrease in numbers in 2018 are not clear, however similar decreases were not observed at either Bridgeport or Crowley Reservoir. These results suggest waterfowl may have been responding to conditions at Mono Lake. Multiple factors influence migrating populations, including conditions elsewhere in the flyway, productivity on breeding grounds, habitat conditions enroute, weather, and disease. Lacking a clear understanding of factors influencing fall waterfowl populations at Mono Lake, and population turnover rate, the reasons for declining or low numbers cannot be stated with any certainty.

Trend analysis suggests long-term declines in the fall populations of Cinnamon Teal, Green-winged Teal and Ruddy Duck at Mono Lake. Although Northern Shoveler counts were very low at Mono Lake in 2018, due to the long-term variability in annual totals for this species, no significant trend has been detected.

No direct correlation has been found between total fall waterfowl and fall lake elevation.

Artemia cyst production in fall provides a partial explanation of the annual variations in waterfowl populations at Mono Lake. This relationship may be direct, as the diet of some waterfowl species using saline lakes has been found to consist largely of *Artemia* cysts, or indirect, reflecting other factors.

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. 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.

In light of the extended time period required for restoration of Mono Lake, recommendations for a less-frequent but more focused approach for the long-term monitoring of waterfowl habitat in the Mono Basin were presented in *Mono Basin Waterfowl Habitat Restoration – 2017 Compliance and Periodic Overview Report* (*Periodic Overview Report*) (LADWP 2018). Recommendations for the limnology and waterfowl population monitoring put forth in the *Periodic Overview Report* were reevaluated during the preparation of this report, and a summary of proposed changes is provided in this report. It is also recommended that the second year of the waterfowl time budget study be completed by the end of 2020 as required by Order 98-05, and that a short-term hypopycnal area investigation be completed.

1.0 INTRODUCTION

Mono Lake is a large terminal saline lake at the western edge of the Great Basin in Mono County, California. Mono Lake is widely known for its value to migratory waterbirds, as it supports up to 30% percent of the North American Eared Grebe (*Podiceps nigricollis*) population, the largest nesting population of California Gulls (*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 can be influenced by factors such as salinity, water depth, and temperature, water influx and evaporation on a seasonal, annual, and inter-annual basis. Saline lakes often respond rapidly to environmental changes, with one of the most influential being alterations to the hydrological budget (Jehl 1988, Williams 2002). Water demands for agriculture, human development and recreation are also 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 to Grant Lake Reservoir 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 (Los Angeles Department of Water and Power (LADWP) data, from "Mono Basin Monthly"). In 1970, the completion of the second aqueduct in the Owens Valley expanded the capacity of the Los Angeles Aqueduct system, resulting in an increase in diversions and frequent full diversion of flows from Lee Vining, Walker, Parker and Rush Creek (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. Levels dropped to a record low of 6,372.0 feet 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 reached 6,391 feet. Decision 1631 by the SWRCB ordered a reduction in diversions by the City, and for 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 the waterfowl habitat monitoring program for Mono Lake. In 2017, LADWP conducted a comprehensive analysis and summarization of restoration actions taken under Order 98-05 since its inception (LADWP 2018). This Periodic Overview Report 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. This report summarizes the results of monitoring conducted in 2018 and reassesses recommendations put forth in the Periodic Overview Report.

2.0 WATERFOWL HABITAT RESTORATION MEASURES

The SWRCB issued Order 98-05 in 1998, defining the waterfowl restoration habitat restoration measures and associated monitoring to be conducted to comply with Decision 1631. The export criteria of Decision 1631 were developed to result in an eventual long-term average lake water elevation 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 restoring the lake elevation to 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 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, maintaining an average lake elevation of 6,392 feet, and the return of perennial flow to the tributary streams would be the most significant of waterfowl habitat restoration measures to be taken. In addition to raising the lake elevation, and the stream restoration efforts, 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 other 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 3.1-1. 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 the 4bii channel 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 8 channel was widened at the upstream end. In contrast to rewatering for constant flow, the final design called for flows overtopping the bank and flowing into the 8 channel at approximately 250 cfs and above. Woody debris was spread and willows were transplanted along new banks following excavation. Further rewatering of Rush Creek side channel complex 8 was deferred by the Stream Scientists. Final review was conducted by McBain and Trush. After presentation of the final review, LADWP followed the recommendations of the Stream Scientists and SWRCB approved the plan. Side 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. At that time, it was determined that rewatering the 10 channel 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 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**
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 the 11 channel 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 the 13 channel 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 the 13 channel. 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)

<table>
<tr><th>Activity</th><th>Goal</th><th>Description</th><th>Progress to Date</th><th>Status</th></tr>
<tr><td rowspan="2">Financial Assistance to USFS for Waterfowl Habitat Improvement Projects at County Ponds and Black Point areas</td><td>To support repairs and improvement of infrastructure on USFS land in the County Ponds area.</td><td>Upon request of the United States Forest Service (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.</td><td rowspan="2">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. USFS has requested funds for a project estimated at $100,000. MLC has requested that the remainder of the funds be applied toward the total cost of the Mill Creek Return Ditch upgrade which would provide benefits for waterfowl habitat. These funds will continue to be budgeted by LADWP until such a time that they have been utilized. Currently, this money has tentatively been included in the 2013 Settlement Agreement as part of Administrative Monitoring Accounts to be administered by a Monitoring Administration Team (MAT).</td><td rowspan="2">In Progress</td></tr>
<tr><td>To support waterfowl habitat improvement projects on USFS land in the Black Point area.</td><td>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.</td></tr>
</table>

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. Once Mono Lake reaches the target elevation, LADWP will reassess the prescribed burn program. Based on results from the assessment, LADWP will either reinstate the program or request relief from the SWRCB from this requirement.	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 will continue until notice from SWRCB is received that LADWP's obligation for this in the Mono Basin is complete.	Ongoing

3.0 WATERFOWL HABITAT RESTORATION MONITORING PROGRAM

The Plan and SWRCB Order WR 98-05 also 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 3.1-1 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 2018, monitoring conducted under the Program included lake elevation, stream flows, lake limnology and secondary producers, and waterfowl population surveys. The remainder of this report provides a summary and discussion on the 2018 data collected under the Program.

Table 3.1-1. 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; 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

Monitoring Component	Description	Required Frequency	Dates Monitoring Performed
Mono Basin Habitat Restoration Monitoring Program *(as described in SWRCB Order 98-05 and the Waterfowl Habitat Restoration Plan dated February 29, 1996)*			
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 counts per year at Mono Lake, Bridgeport Reservoir and Crowley Reservoir	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

3.1 Hydrology

Background

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. Within the hydrographically closed Mono Basin, all surface and groundwater drains towards Mono Lake. 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 more gently sloping shoreline is more typical of the north and east shores (Vorster 1985, Raumann et al. 2002).

The hydrologic components monitored at Mono Lake in 2018 include lake elevation and stream flows. LADWP hydrographers conduct the hydrologic monitoring.

Lake Elevation

Since Mono Lake lies in a closed basin with no outlet, lake elevation is driven by inflow from surface water, precipitation, ground water, and evaporative losses (Vorster 1985). Climatic variation in the late Pleistocene and Holocene periods resulted in an extreme high stand of 7,200 feet, and an extreme low of an approximately 6,368 foot lake elevation (Scholl et al. 1967 in Vorster). In historic times, lake level and salinity has fluctuated in response to climate variation (SWRCB 1994).

Stream Flow

Perennial creeks tributary to Mono Lake originate on the east slope of the Sierra Nevada. There are five primary creeks in the Mono Basin: Rush, Lee Vining, Mill, Parker, and Walker Creeks - three of which (Rush, Lee Vining and Mill) reach the western shoreline of Mono Lake. Parker and Walker Creeks are tributary to Rush Creek. The creeks tributary to Mono Lake are primarily snow-melt fed systems, with peak flows typically occurring in June or July, especially in normal-to-wet years for the larger creeks, but 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. Rush Creek was permanently re-watered in 1982, but the two tributaries to Rush Creek (Parker Creek and Walker Creek), were not re-watered until 1990. Lee Vining Creek was re-watered in 1986. Based on hydrologic Water Year periods (October through September of each year), prior to 1990, the combined input to Mono Lake from Rush and Lee Vining Creeks was lower due to export activity by the City (Figure 3.1-1). Prior to 1990, flows in Rush and Lee Vining Creeks were also more variable, occurring mainly during wet years.

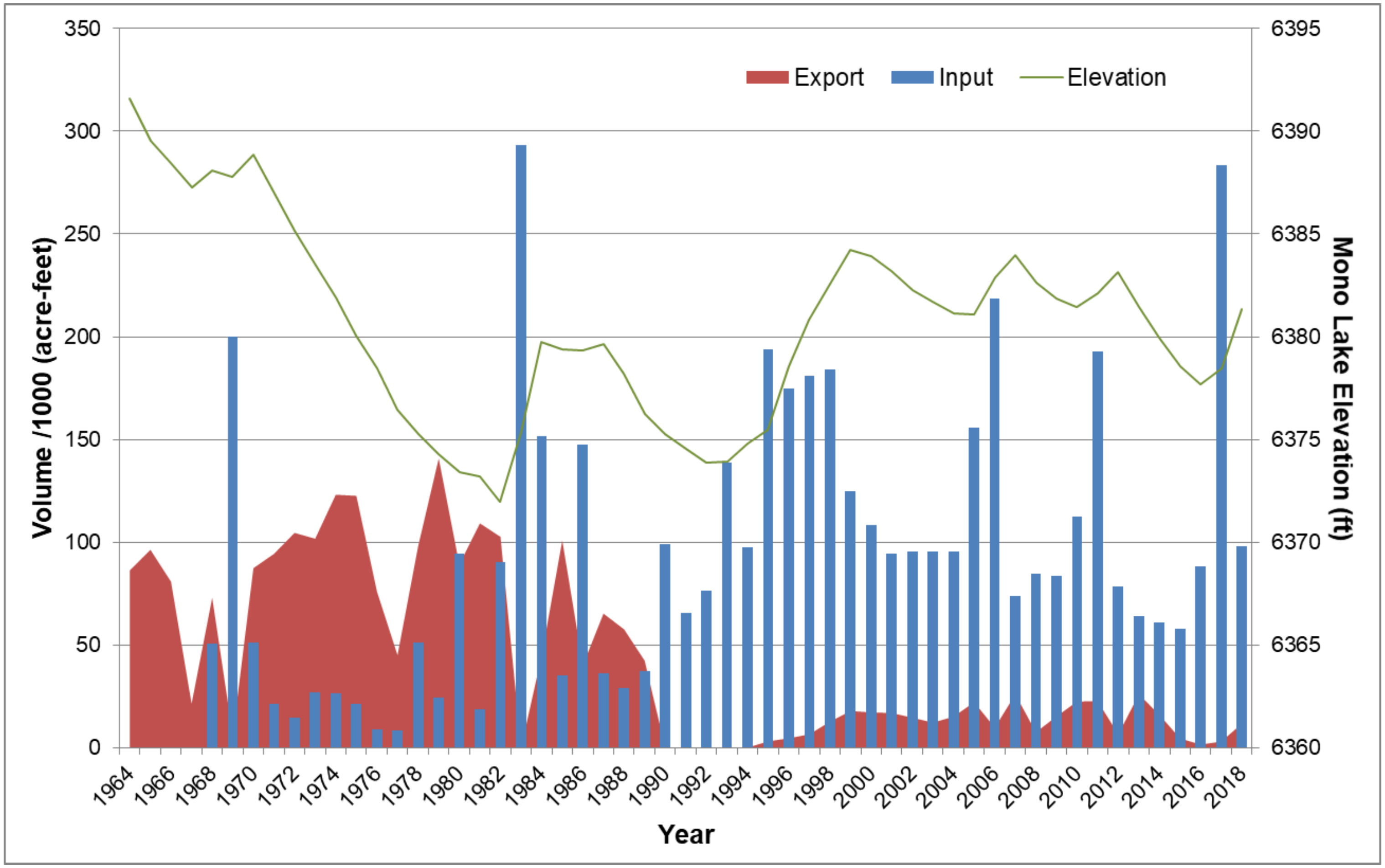

Figure 3.1-1. Annual Input, Export, and Mono Lake Elevation

Input to Mono Lake from Rush Creek and Lee Vining Creek and surface elevation from 1963-2018 reported in acre-feet per water year (October-September).

3.1.1 Hydrologic Monitoring Methodologies

Mono Lake Elevation Monitoring

The elevation of Mono Lake is measured manually on a biweekly basis at a staff gauge located near the Old Marina along the west shore. Lake elevation monitoring data are used for determining progress in meeting the targeted lake level, for determining appropriate export amounts, and for providing environmental data to evaluate the response of biological indicators including secondary producers, vegetation, and waterfowl.

Stream Flow Monitoring

Stream flow monitoring is conducted along the five perennial creeks– Rush, Lee Vining, Mill, Parker, and Walker Creek. There are eight gauging stations tracking Mono Lake inflow along the tributaries: six are operated and maintained by LADWP and two are operated and maintained by Southern California Edison. At each station, flow is measured at 15-minute intervals and converted into daily flow, which is used to calculate monthly and annual inflow into Mono Lake. Stream flow data are used for determining compliance with the Mono Basin Stream and Stream Channel Restoration Plan (LADWP 1996), and to provide environmental data to evaluate the response of biological indicators.

3.1.2 Hydrology Data Summary and Analysis

Lake Elevation

Monthly Mono Lake elevation data were summarized for the time period 1998-2018, or since implementation of Order 98-05. Simple linear regression was used to describe lake elevation changes since implementation of the Order. Patterns of lake elevation change were evaluated on a yearly and monthly basis. To elucidate the differences in patterns observed, water years were further categorized as to runoff year type as described in Order 98-05. The runoff year is April 1 to March 31, and runoff year type is based on the LADWP April 1 Mono Basin runoff forecast, although adjustments may be made on May 1. Runoff year type is based on a comparison of the total acre-feet of predicted runoff to the 1941-1990 average runoff of 122,124 acre-feet (Table 3.1-1).

Table 3.1-1. 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

Streams

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 (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 above gauging stations are operated and maintained by LADWP. The inflow from Mill and Wilson Creeks cannot be precisely determined due to discontinuation of gauging stations; thus, the inflow is estimated by summing outflow of Lundy Lake through Mill Creek and through the power plant into Wilson Creek. Currently both stations are operated and maintained by Southern California Edison. Dechambeau, Mill, and Wilson Creeks lose water through diversion before reaching Mono Lake; however, flow from Log Cabin Creek and a series of springs located in the northwest corner of the lake may make up the loss.

3.1.3 Hydrology Results

Runoff

The 2018-2019 Runoff year was classified as “Normal” type at 108,164 acre-feet, or 89% of the long-term average. The 2018-2019 runoff was well below the 2017-2018 runoff, which ranked 2nd highest with 235,544 acre-feet, or 193% of the long term average. Since Decision 1631, there have been three distinct wet periods even though the magnitude and duration of the wet periods has decreased progressively. The first wet period lasted between 1995 and 1998 with the average of 146% of Normal; the second wet period only lasted two years from 2005 to 2006 averaging 153% of Normal; the third wet period also lasted two years from 2010 to 2011 averaging 130% of Normal. The year 2017 was the second wettest on record with 195% of Normal but followed by below average runoff year and also preceded by the driest 5-year period on record.

Lake Elevation

From 1998 to 2018, Mono Lake has experienced four periods of increasing elevation, and three subsequent decreases, through a total elevation range of 8.0 feet (Figure 3.1-2). The highest elevation the lake has achieved since 1998 has been 6,385.1 feet which occurred in July 1999 and August 2006. 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. The most recent increase began in 2017 in response to the extreme wet year of 2016-2017 in which the lake rose from 6,376.8 feet in January 2017, to a maximum of 6,381.8 feet in June 2018.

The lake elevation tends to be most stable in the winter months of January through the end of March, showing slight declines in early spring, especially in dry to wet/normal years (Figure 3.1-3). Increasing evapotranspiration rates in early spring may lead to a slight decrease in lake levels, however in extreme wet years, the lake level has not shown the same decrease. In dry to normal years, early runoff will cause a slight increase by June, however this bump in elevation is slightly later in wet/normal years. In extreme wet years, the maximum annual lake level is not typically reached until July or August.

Stream Flows

Prior to 1990, the combined input to Mono Lake from Rush and Lee Vining Creeks was lower and more sporadic, mainly occurring during wet years due to export by the City (see Figure 3.1-1). Decision 1631 and Order 98-05 dictated the instream flows (or base flows) and channel maintenance flows (or peak flow) for Lee Vining Creek, Rush Creek, Parker and Walker Creek. Instream and channel maintenance flows for other tributaries to Mono Lake were not specified by the Order.

Since 1990, Rush Creek has averaged 62,171 acre-foot discharge annually while Lee Vining Creek has averaged 39,288 acre-foot (Table 3.1-2). The highest annual input on record is 185,473 acre-foot in 1983 for Rush Creek and 91,133 acre-feet in 2017 for Lee Vining Creek. Dechambeau Creek has averaged 826 acre-feet since 1944 and has contributed less than 1% of total annual input since 1990. The combined flow of Mill and Wilson Creeks has averaged 18,821 acre-feet since 1968 and has contributed approximately 15% each year to annual input since 1990.

In the Mono Basin, runoff year types are cyclical, with wet years followed by dry years. In the late 1930s to early 1940s, the late 1970s to 1980s, and the late 1990s, the wet periods lasted longer than they have as of late (Figure 3.1-4). Dry and dry/normal years have been the most frequent runoff year type occurring in the Mono Basin since 1990, as for the last 13 of 28 years runoff has been less than 82.5% of normal, or more frequently, less than 68% of normal. Furthermore, in 19 of the last 28 years, runoff was less than 95% of the overall average of 95%. Two extreme wet years have been experienced since Order 98-05, including runoff year 2006 and 2017. Between 1990 and 1999, the runoff was 102% of the long-term average. In contrast, between 2000 and 2016, average runoff was 85%, during which only 4 years show runoff above 100% of the long-term mean. The extreme wet year of 2017 had runoff of 176% of normal.

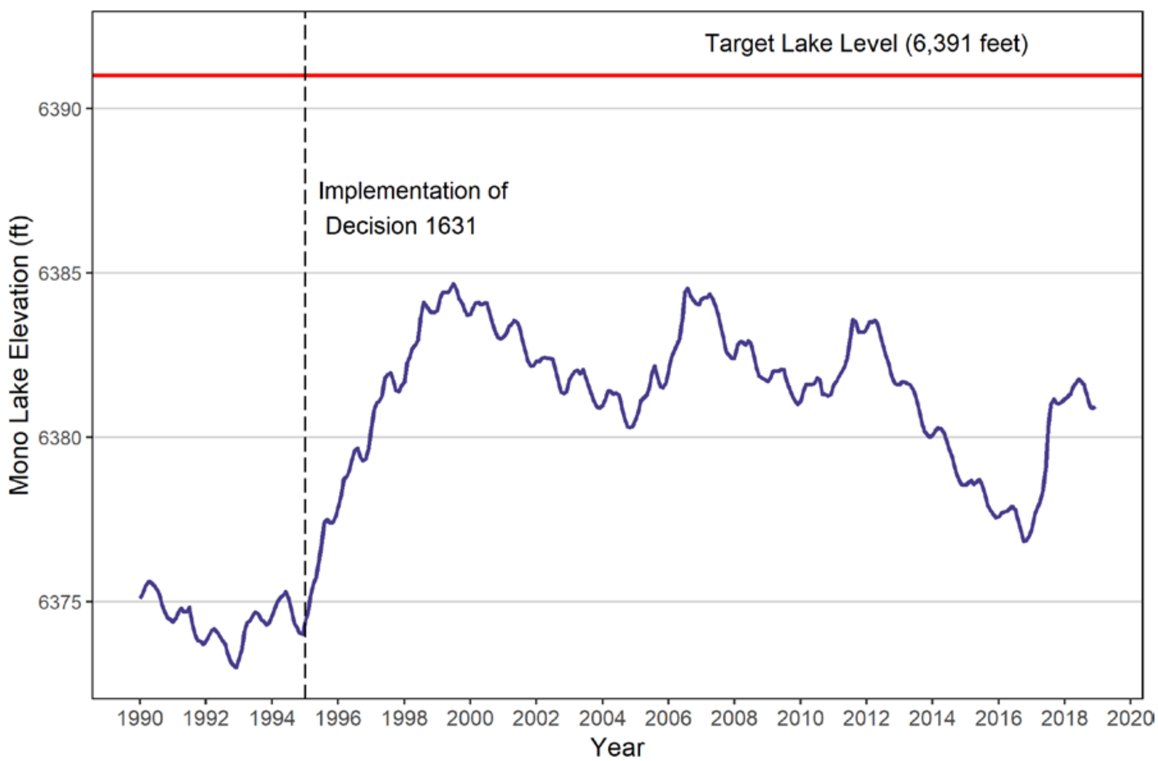

Figure 3.1-2. Mono Lake Elevation between 1998 and 2018

Since Decision 1631, there have been four wet periods of lake level increase.

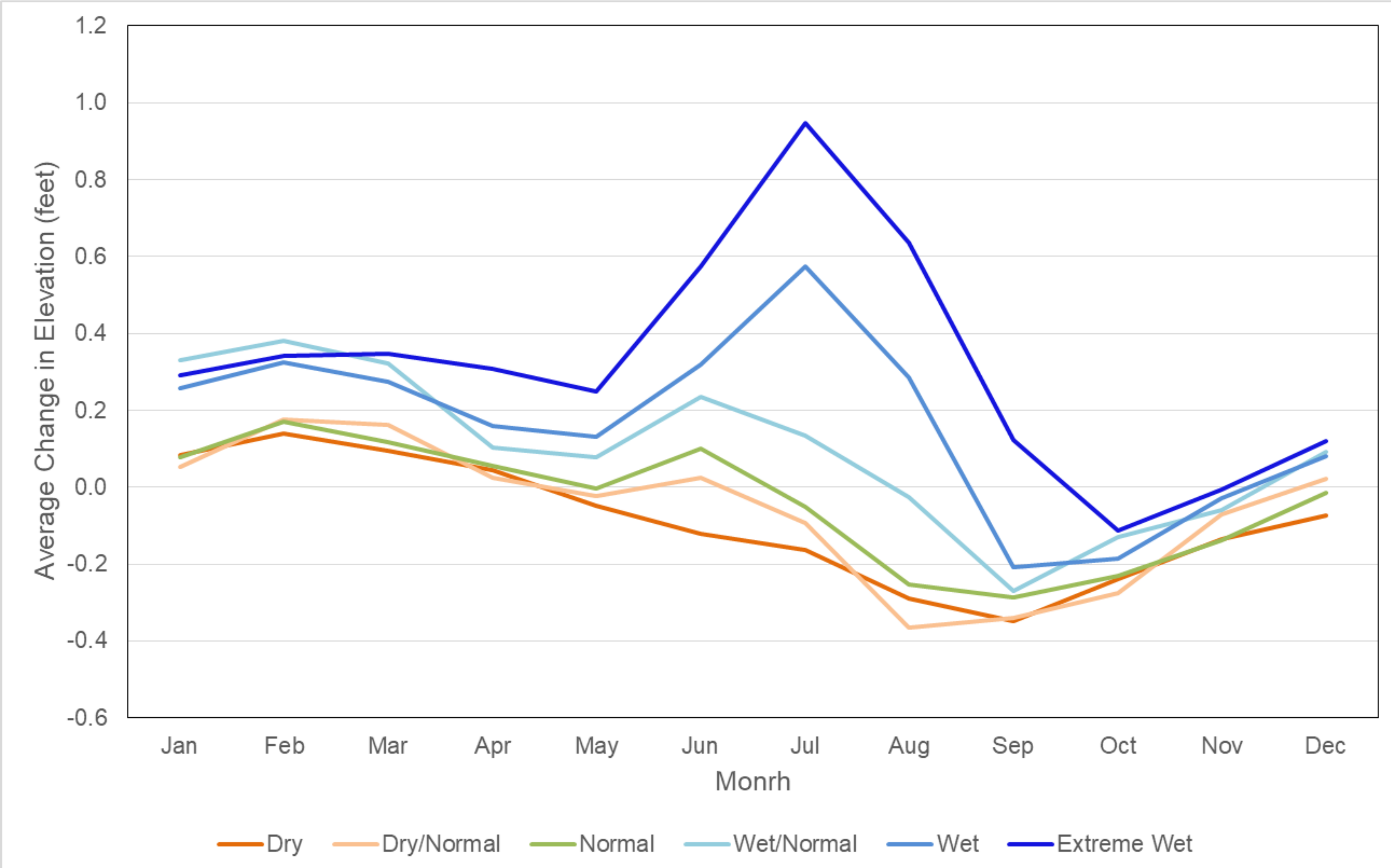

Figure 3.1-3. Monthly Pattern of Lake Elevation Changes by Runoff Year Type

Table 3.1-2. Annual Flow Volume in Acre-Feet of Five Mono Lake Tributaries Based on Water Year

Year	Rush Creek	Lee Vining Creek	Dechambeau Creek	Mill Creek	Wilson Creek
1990	71,047	18,644	326	489	8,626
1991	35,714	20,562	265	513	8,213
1992	44,632	20,799	179	501	10,089
1993	77,461	42,279	440	1,798	16,912
1994	56,776	29,377	451	516	10,603
1995	94,596	66,443	911	10,203	21,697
1996	91,842	56,284	1,244	4,566	20,992
1997	82,424	66,317	1,486	4,623	26,290
1998	93,178	62,335	1,326	6,017	21,097
1999	58,047	46,204	1,151	1,459	18,013
2000	50,497	40,432	750	1,252	15,118
2001	49,357	31,034	576	773	12,500
2002	45,900	36,599	406	788	11,920
2003	49,028	30,778	530	1,108	14,091
2004	47,644	31,872	550	159	14,956
2005	72,766	55,367	995	6,823	19,817
2006	108,899	75,861	1,460	10,085	22,064
2007	38,428	24,091	998	1,267	8,906
2008	45,159	25,632	588	2,557	10,708
2009	36,570	30,654	586	3,658	12,111
2010	57,622	34,776	672	4,314	15,015
2011	96,433	65,454	1,151	7,588	22,409
2012	46,535	19,487	927	2,369	8,904
2013	34,776	18,320	476	2,179	8,237
2014	31,893	20,048	340	1,979	6,560
2015	32,754	16,525	273	1,806	6,679
2016	44,242	28,421	276	2,751	12,481
2017	145,349	91,133	1,433	19,550	25,861
2018	63,397	33,625	1,211	6,649	15,072

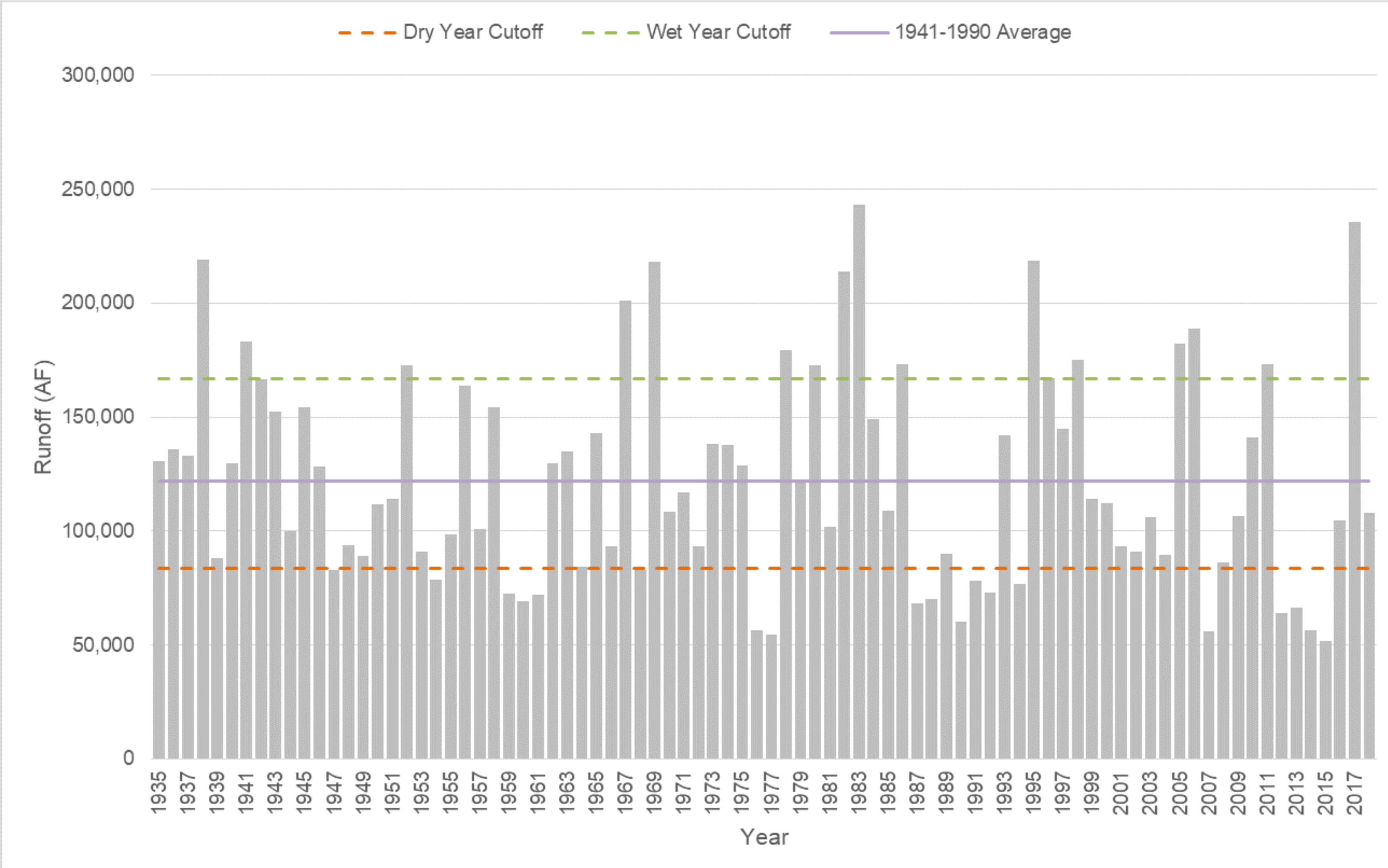

Figure 3.1-4. Mono Basin Runoff Based on Runoff Year for Entire Period of Record 1935-2018

3.1.4 Hydrology Discussion

Lake Elevation

Climatic factors may be influencing Mono Lake and its recovery. Mono Lake has not yet reached the target lake elevation, even though initial modeling predicted this restoration objective would be met in approximately 20 years (or by 2015). Following the preliminary injunction in 1989 and Decision 1631 in 1994, there have been a series of wet and dry periods that have affected the elevation of Mono Lake. Based on an assessment of runoff data dating back to the late 1930's, it appears that dry years are becoming drier in recent history, thus inhibiting the recovery to the target level.

Stream Flows

Runoff in the Mono Basin is 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 found to last longer than the more recent wet periods. Since both Lee Vining and Rush Creeks were permanently re-watered, 31% of years were classified as "Dry" compared to 18% for years prior to 1990. Meanwhile, 24% of years were classified as either "Wet" or "Extreme Wet" after the permanent re-watering as compared to 18% prior to 1990. These patterns suggest more extreme shifts in climate as more years fall into either dry or wet.

3.2 Limnology

Mono Lake supports a relatively simple yet productive aquatic ecosystem. Benthic and planktonic 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. 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 midges (*Cuciloides occidentalis*), and deer fly (*Chrysops* spp.) graze on benthic algae (Jones and Stokes Associates, Inc 1993).

Within the hydrographically closed Mono 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 to 97 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 1996). 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 of 2012.

Laboratory support including the analysis of ammonium and chlorophyll *a* has been provided by Environmental Science Associates (ESA), Davis, California since 2012.

This report summarizes monthly field sampling in 2018, and discusses the results in the context of the entire period of record. In addition to the report summarizing Mono Lake conditions in 2018, past findings are also summarized to demonstrate long term trends in the *Artemia* population and Mono Lake water parameters to gain deeper insight into *Artemia* population dynamics. This report also presents recommendations for the program.

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 are consistent and follow those identified in *Field and Laboratory Protocols* except where noted below.

Meteorology

One meteorological station on Paoha Island provided 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. Sensor readings are made every second and stored as either ten-minute averages or hourly values in a Campbell Scientific CR 1000 datalogger. Data are downloaded to a storage module, which is collected periodically during field sampling visits.

At the Paoha Island station, wind speed and direction (RM Young wind monitor) are measured by sensors at a height of 3 m above the surface of the island and are averaged over a 10-minute interval. During the 10-minute interval, maximum wind speed is also recorded. Using wind speed and direction measurements, the 10-minute wind vector magnitude and wind vector direction are calculated. Hourly measurements of photosynthetically available radiation (PAR, 400 to 700nm, Li-Cor 192-s), 10-minute averages of relative humidity and air temperature (Vaisalia HMP35C), and total rainfall (Campbell Scientific TE525MM-L tipping bucket) are also stored. The minimum detection limit for the tipping bucket gage is 1 mm of water. The tipping bucket is not heated; therefore the instrument is less accurate during periods of freezing due to the sublimation of ice and snow. Due to inconsistent precipitation readings of the Paoha Island weather station, daily precipitation recorded at Cain Ranch is reported.

In addition to the Paoha Island station, monthly total precipitation has been recorded at the LADWP Cain Ranch site since May 1931. The 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.

Field Sampling

Sampling of the physical, chemical, and biological properties of the water including the *Artemia* community is conducted at 12 buoyed stations at Mono Lake (Figure 3.2.1). The water depth at each station at a lake elevation of 6384.5 feet1 (1,946 m) is indicated on Figure 3.2.1. Stations 1-6 are considered western sector stations, and stations 7-12 are eastern sector stations. Starting in November 2018, monitoring was conducted in 2 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 are generally conducted around the 15^{th} of each month and the 2018 sampling dates are listed in Table 3.2.1. In 2018, Stations 1, 2, and 12 were not sampled in May due to high wind. Due to severe weather, no *Artemia* sampling was conducted in September, and sampling of water chemical/physical properties was not conducted at Stations 1, 10, and 12.

Table 3.2-1. Mono Lake Limnology Sampling Dates for 2018

Month	Sampling Dates	
Feb	2/13/2018	
Mar	3/7/2018	
Apr	4/20/2018	
May	5/16/2018	
Jun	6/13/2018	
Jul	7/17/2018	
Aug	8/22/2018	
Sep	9/14/2018	
Oct	10/16/2018	
Nov	11/13/2018	11/14/2018
Dec	12/12/2018	12/13/2018

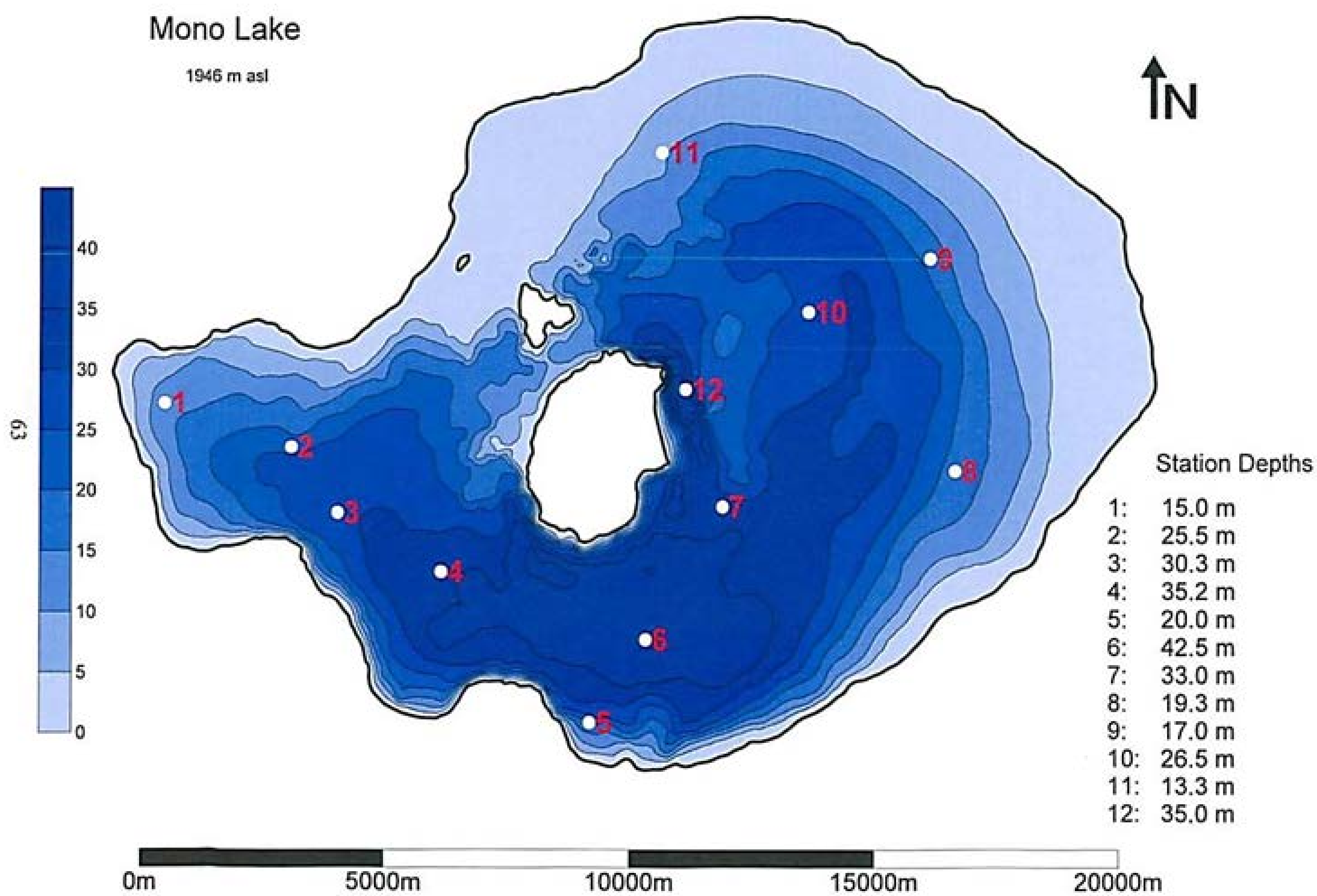

Figure 3.2-1. Sampling Stations at Mono Lake and Associated Station Depths

Physical and Chemical

Sampling of the physical and chemical properties include lake transparency, water temperature, conductivity, dissolved oxygen, and nutrients (ammonium). Lake transparency is measured at all 12 stations using a Secchi disk.

<u>Conductivity</u>

A high-precision conductivity temperature-depth (CTD) profiler is used to record conductivity at 9 stations (2, 3, 4, 5, 6, 7, 8, 10 and 12). During sampling, the CTD is initially lowered just below the surface of the water for 40 seconds during the pump delay time. The CTD is then lowered at a rate of approximately 0.5 meter/second with data collected at approximately 12.5 centimeter depth intervals. The Seabird CTD is programmed to collect data at 250 millisecond intervals. Conductivity data is collected from the CTD field sampling device on a monthly basis.

<u>Dissolved Oxygen</u>

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 are 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.

<u>Ammonium Sampling</u>

Monitoring of ammonium in the epilimnion is conducted using a 9-m integrated sampler at stations 1, 2, 5, 6, 7, 8, and 11. Ammonium is 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 for ammonium analyses are filtered through Gelman A/E glass-fiber filters, and following collection, immediately placed onto dry ice and frozen in order to stabilize the ammonium content (Marvin and Proctor 1965). Ammonium samples are transported on dry ice back to the laboratory transfer station. The ammonium samples are stored frozen until delivered to the University of California Davis Analytical Laboratory (UCDAL) located in Davis, California. Samples are stored frozen until analysis.

Starting in August 2012, the methodology used by UCDAL for ammonium is flow injection analysis. In July 2012, this method was tested on high salinity Mono Lake water and was found to give results comparable to previous years. This method has detection limits of approximately 2.8 µM. Immediately prior to analysis, frozen samples are allowed to thaw and equilibrate to room temperature, and are shaken briefly to homogenize. Samples are heated with salicylate and hypochlorite in an alkaline phosphate buffer (APHA 1998a, APHA 199b, Hofer 2003, Knepel 2003). EDTA

(Ethylenediaminetetraacetic acid) is added in order to prevent precipitation of calcium and magnesium, and sodium nitroprusside is added in order to enhance sensitivity. Absorbance of the reaction product is measured at 660 nm using a Lachat Flow Injection Analyzer (FIA), QuikChem 8000, equipped with a heater module. Absorbance at 660 nm is directly proportional to the original concentration of ammonium, and ammonium concentrations are calculated based on absorbance in relation to a standard solution.

Chlorophyll a Sampling

Monitoring of chlorophyll *a* in the epilimnion is conducted using a 9-m integrated sampler at stations 1, 2, 5, 6, 7, 8, and 11. Chlorophyll is 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 are filtered into opaque bottles through a 120 µm sieve to remove all life stages of *Artemia*. Chlorophyll a samples are kept cold and transported on ice back to the laboratory transfer station located in Sacramento, CA. The determination of chlorophyll *a* is conducted through fluorometric analysis following acetone extraction. Fluorometry was chosen, as opposed to spectrophotometry, due to higher sensitivity of the fluorometric analysis, and because data on chlorophyll *b* and other chlorophyll pigments were not needed.

At the laboratory transfer station in Sacramento, water samples (200 mL) are filtered onto Whatman GF/F glass fiber filters (nominal pore size of 0.7 µm) under vacuum. Filter pads are then stored frozen until they could be mailed overnight in dry ice to the University of Maryland Center for Environmental Science Chesapeake Biological Laboratory (CBL), located in Solomons, Maryland. Sample filter pads are extracted in 90% acetone and then refrigerated in the dark for 2 to 24 hours. Following refrigeration, the samples are allowed to warm to room temperature, and then centrifuged to separate the sample material from the extract. The extract for each sample are then analyzed on a fluorometer. Chlorophyll a concentrations are calculated based on output from the fluorometer. Throughout the process, exposure of the samples to light and heat is avoided.

The fluorometer used in support of this analysis is a Turner Designs TD700 fluorometer equipped with a daylight white lamp, 340-500 nm excitation filter and >665 nm emission filter, and a Turner Designs Trilogy fluorometer equipped with either the non-acid or the acid optical module.

Artemia Population Sampling

The *Artemia* population is sampled by one vertical net tow at each of 12 stations (Figure 3.2.1). Samples are taken with a plankton net (0.91 m x 0.30 m diameter, 118 µm Nitex mesh) towed vertically through the water column. Samples are preserved with 5% formalin in Mono Lake water.

An 8x to 32x stereo microscope is used for all *Artemia* analyses. Depending on the density of shrimp, counts are 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 are split so that approximately 100-200 individuals were subsampled. Shrimp are classified as nauplii (instars 1-7), juveniles (instars 8-11), or adults (instars >12), according to Heath's classification (Heath 1924). Adults are sexed and the reproductive status of adult females was determined. Non-reproductive (non-ovigerous) females are classified as empty. Ovigerous females are classified as undifferentiated (eggs in early stage of development), oviparous (carrying cysts) or ovoviviparous (naupliar eggs present).

An instar analysis is completed for seven of the twelve stations (Stations 1, 2, 5, 6, 7, 8, and 11). Nauplii at these seven stations are further classified as to specific instar stage (1-7). Biomass is determined from the dried weight of the shrimp tows at each station. After counting, samples are rinsed with tap water and dried in aluminum tins at 50°C for at least 48 hours. Samples are weighed on an analytical balance immediately upon removal from the oven.

Artemia Fecundity

When mature females are present, an additional net tow is 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 are 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 are randomly selected, isolated into individual vials, and preserved with 5% formalin. Female length is measured at 8X from the tip of the head to the end of the caudal furca (setae not included). Egg type is noted as undifferentiated, cyst, or naupliar. Undifferentiated egg mass samples are discarded. Brood size is determined by counting the number of eggs in the ovisac and any eggs dropped in the vial. Egg shape is noted as round or indented.

3.2.2 Limnology Data Summary and Analysis

Meteorology

The daily mean wind speed, maximum mean wind speed, and relative humidity are calculated from 10-minute averaged data from the Paoha Island site. Winter temperature is calculated by averaging the monthly average maximum (or minimum) temperature from December of the previous year and January and February of the subsequent year. More specifically, the monthly average from December 2016 is combined with the monthly average from January and February 2017 to obtain the winter average for 2017. Summer temperature is calculated as the average monthly temperature between June and August. Annual precipitation is a sum of precipitation occurring within one calendar year.

Physical and Chemical

An ammonium profile is developed from the samples taken at the eight discrete depths. A chlorophyll profile is developed from the samples taken at the seven discrete depths. In situ, conductivity measurements at Station 6 are corrected for temperature (25˚C) and reported at one meter intervals beginning at one meter in depth down to the lake bottom. Salinity expressed in g/L is calculated based on the equation presented by Jellison in past compliance reports.

Salinity and Mono Lake Elevation

High salinity negatively affects 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). Even though the salinity level in Mono Lake has not neared the tolerance level for *Artemia*, the salinity level is higher than the pre-export period. The pre-diversion salinity was estimated to be 48 g/L (Dana and Lenz 1986) at a lake level around 6,417 feet. As of December 2018, salinity ranged between 86.2 g/L and 93.0 g/L at Station 6 at the lake level of 6,381 feet. Lake elevation data is obtained directly from the LADWP database records. Annual lake elevation for year-to-year comparison is calculated based on the average April (water year) measurements.

Artemia Population Statistics

Calculation of long-term *Artemia* population statistics follows the method proposed by Jellison and Rose (2011). Daily values of adult *Artemia* between sampling dates are linearly interpolated by using an R package *zoo*. The mean, median, peak and centroid day (calculated center of abundance of adults) are then calculated for the time period May 1 through November 30, during which adult *Artemia* population is most abundant.

Long-term statistics are determined by calculating the mean, minimum, and maximum values for the time period 1979-2018.

<u>Artemia</u> 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 deepening 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. In this section, the annual *Artemia* population mean during monomixis and meromixis is quantitatively compared to ammonium, Mono Lake input, and salinity to illustrate the importance of the lake mixing regime to *Artemia* population dynamics.

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 3 months of dormancy in cold (<5°C) water 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 (Jellison et al. 1991). 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) is 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.

Because of the important relationship between water and ambient temperature (Jellison et al. 1989a; Jellison et al. 1990), simple linear regression is performed to examine the relationship between monthly water temperature readings at various depths and monthly ambient temperature readings. The relationships may provide better understanding of the effect of a changing climate on Mono Lake *Artemia* populations.

All analyses are performed using a statistical software, R (the R project statistical computing).

3.2.3 Limnology Results

Meteorology

Wind Speed, relative humidity, air temperature and precipitation data from the weather station at Paoha Island are summarized below for 2018.

Wind Speed and Direction

Mean daily wind-speed varied from 0.97 to 13.12 m/sec in 2018, with an overall mean for this time period of 3.69 m/sec (Figure 3.2.2). The daily maximum 10-min wind speed (5.56 m/sec) on Paoha Island averaged almost twice as much as the mean daily wind speed. The maximum recorded 10-min reading of 30.31 m/sec occurred on the afternoon of November 17. As the case in previous years, winds were predominantly from the south (mean 194.3 degrees).

Air Temperature

Daily air temperatures as recorded at Paoha Island in 2018 ranged from a low of -12.7°C on February 20 to a high of 34.6°C on July 18 (Figure 3.2.3). Daily average winter temperature (January through February) ranged from -8.1°C to 9.4°C. The average maximum daily temperature in winter was 8.3°C, slightly lower than the value observed in 2017 but much higher than previously recorded values. The average maximum daily summer temperature (June through August) was 28.2°C while the average minimum daily summer temperature was 12.5°C. Both values were slightly higher than those observed in 2017.

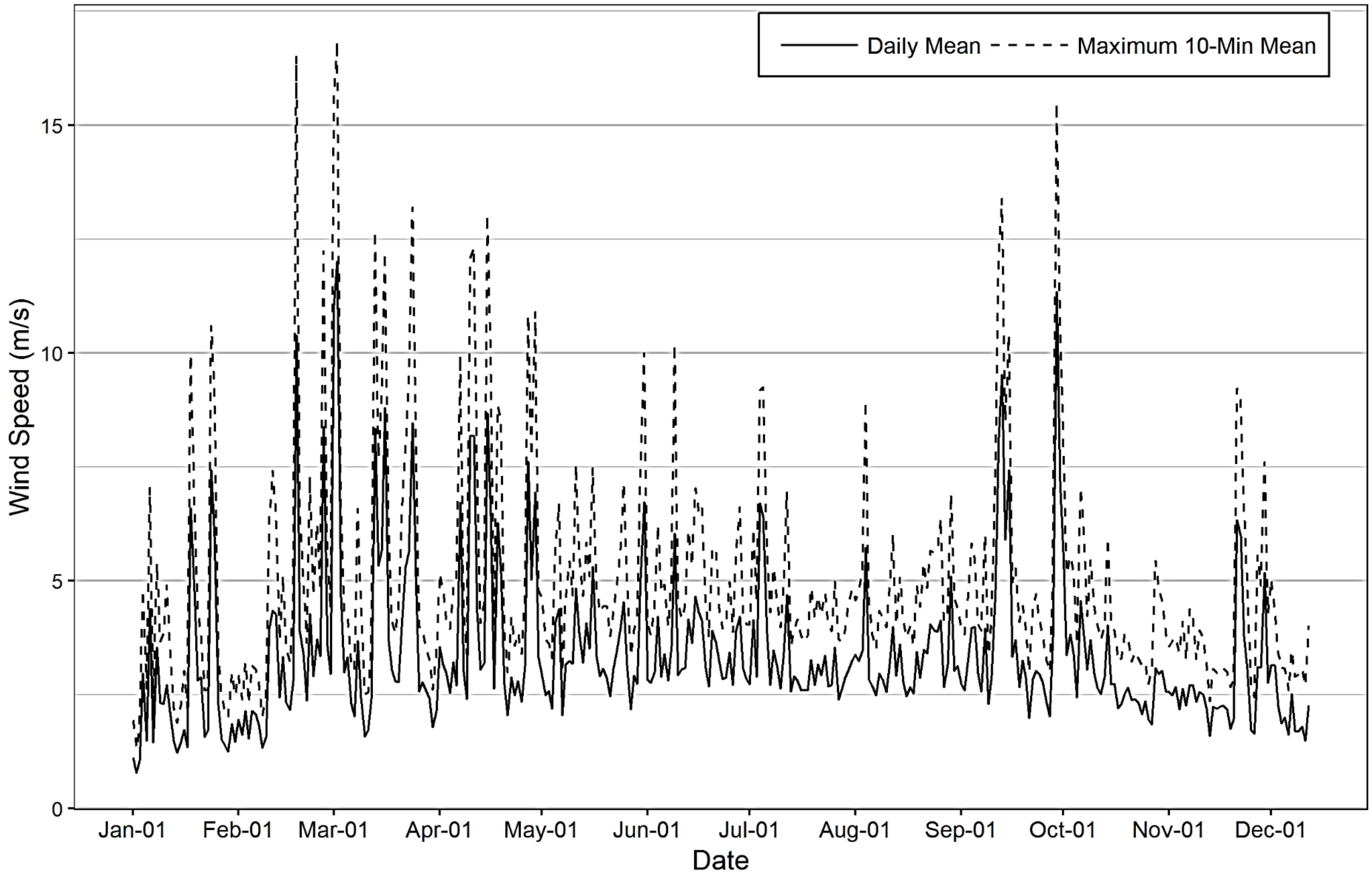

Figure 3.2-2. Daily Mean and Mean Maximum 10-Minute Wind Speed

Wind speed was recorded at Paoha Island from January 1 to December 13, 2018.

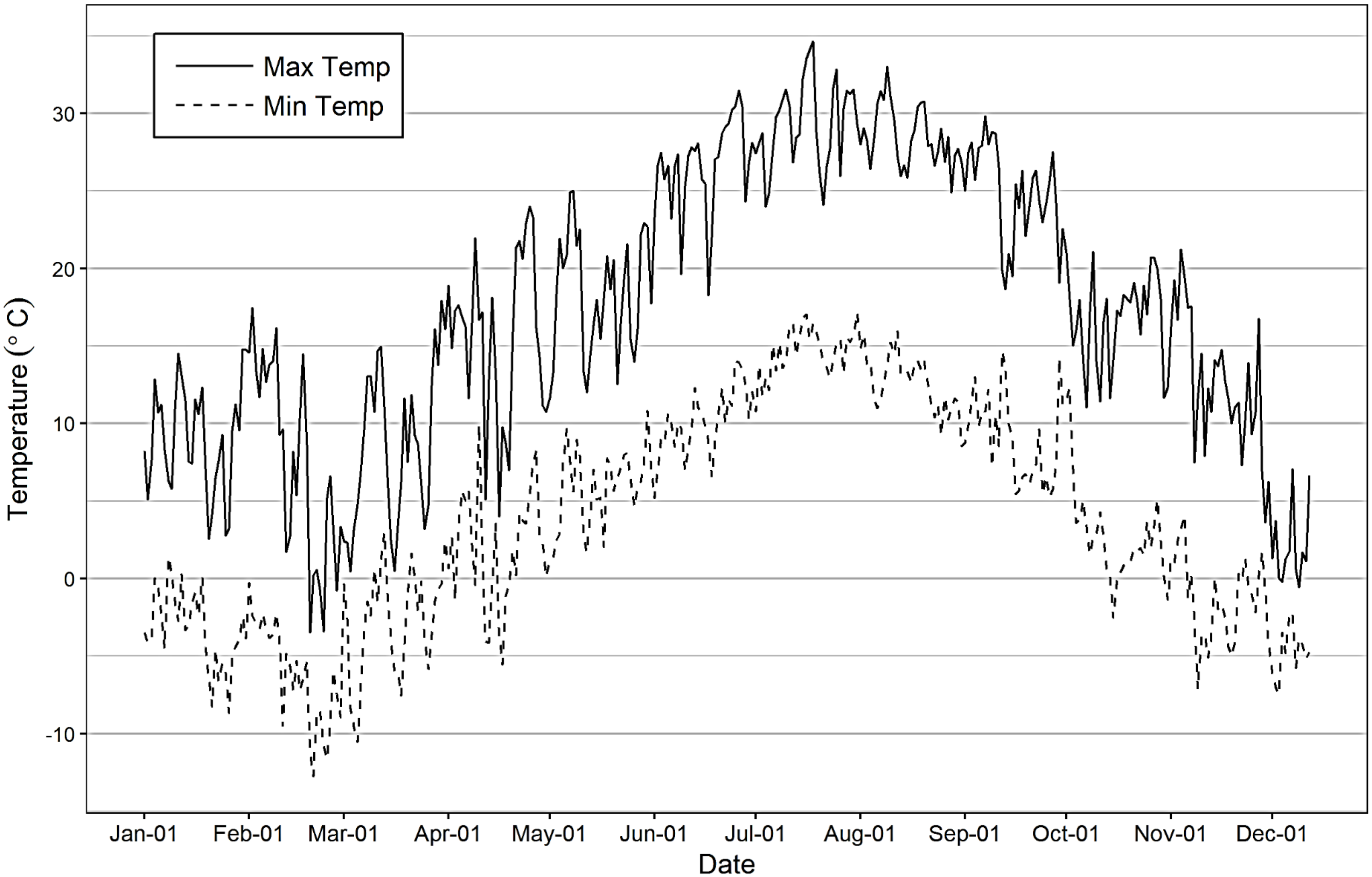

Figure 3.2-3. Minimum and Maximum Daily Temperature (°C)

Air temperature was recorded at Paoha Island from January 1 to December 13, 2018.

Relative Humidity and Precipitation

The mean relative humidity for the period between January 1 and December 13, 2018 was 56% (Figure 3.2.4). The total precipitation during the same period measured at Cain Ranch was 9.54 in. Precipitation events were more frequent in spring and the largest single day total precipitation of 2.28 in. was recorded on March 22 (Figure 3.2.5). In January and February, 0.42 in. of precipitation was recorded. Spring months (March through May) produced 6.13 in. of precipitation followed by much lower summer month precipitation (1.17 in). Fall (September to November) precipitation increased only slightly to 1.44 in. December precipitation was 0.33 in. The highest frequency of days with precipitation (10) occurred in the month of March.

Long Term Trend

The winter of 2017-18 was the second warmest winter since 1951 in terms of the average maximum winter temperature (Figure 3.2.6). In the past five years, the top three warmest winters since 1951 were observed: 2013-14 (3rd), 2014-15 (1st), and 2017-18 (2nd). The winter of 2016-17 was not as warm as these two previous years due in part to more frequent winter storms, but very dry and warm conditions prevailed most of the winter in 2017-18. In terms of the average minimum winter temperature, the winter of 2017-18 was not as extreme as the maximum temperature, but remained more than 1°C warmer than the long term average of -6.06°C. The summer of 2018 was 3rd and 4th warmest in terms of average maximum and minimum temperature, respectively (Figure 3.2.7). The average minimum temperature was more than 2°C warmer than the long term average. Winter precipitation (the months of December-February) in 2017-18 (0.42 in) was ranked the 2nd driest in 87 years and a mere 8% of the long term average (4.96 in) while summer precipitation was ranked 36th in 88 years and 110% of the long term average (Figure 3.2.8, Figure 3.2.9). The winter preceding the 2018 monitoring year was extremely dry and warmer, and summer of 2018 was warmer but wetter.

There is no clear long-term trend for average summer and winter temperatures except for average summer minimum temperature ($r=0.55$, $p<0.0001$). A combination of above average summer minimums since 1995, and below average summer minimum temperature during the earlier part of the record (between 1962 and 1987), contributed to this significant positive trend of increasing minimum summer temperatures. The average winter minimum temperature has been above the long-term average (-6.1°C) for the three winters prior to the winter of 2017-18, and the winter of 2014-15 was particularly warm as the highest average minimum since 1951 was recorded.

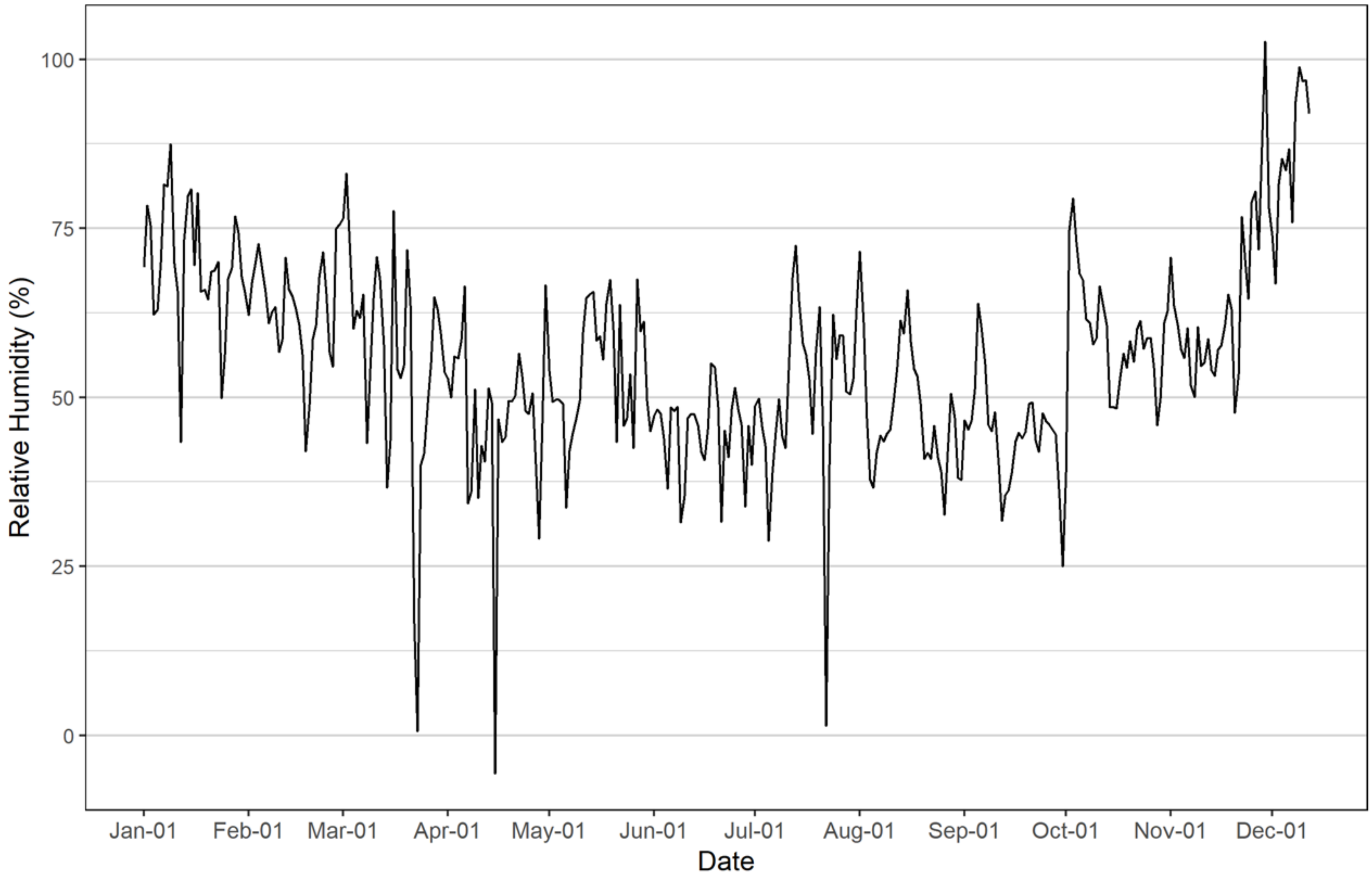

Figure 3.2-4. Mean Daily Relative Humidity (%)

Relative humidity was recorded at Paoha Island from January 1 to December 13, 2018.

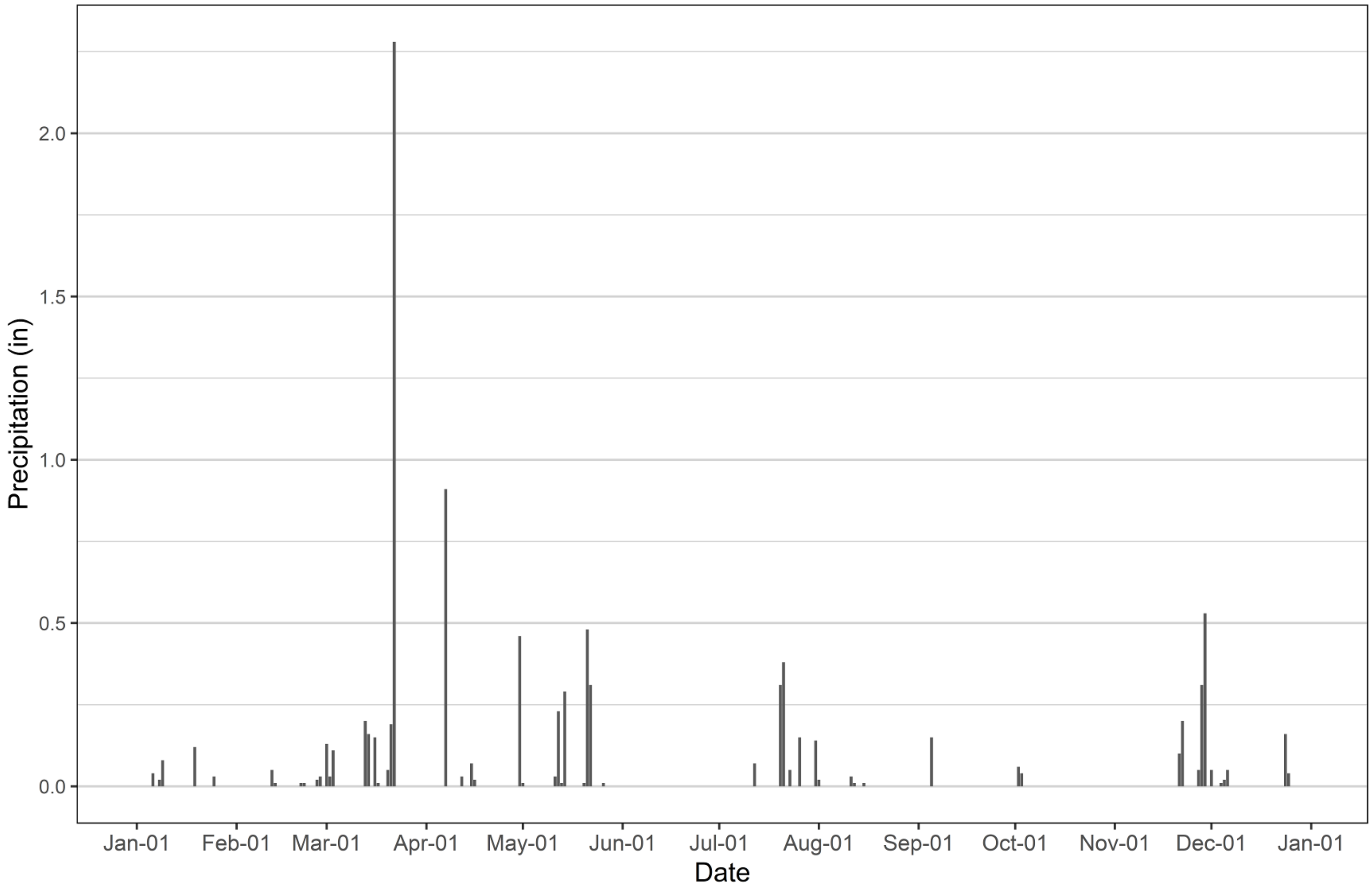

Figure 3.2-5. Total Daily Precipitation (mm)

Precipitation was recorded at Cain Ranch from January 1 to December 31, 2018.

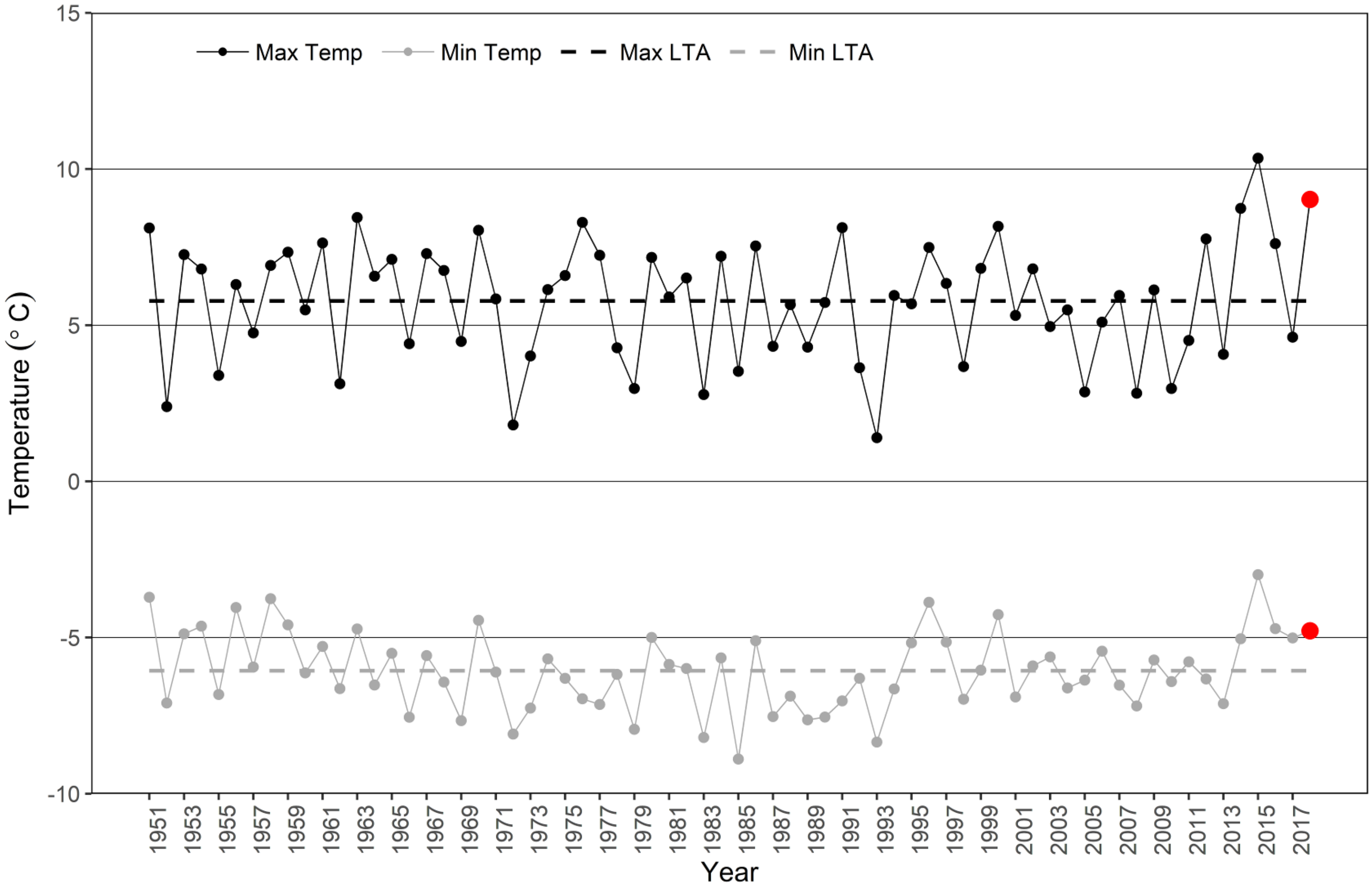

Figure 3.2-6. Average Temperature during Winter Months (December through February)

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.

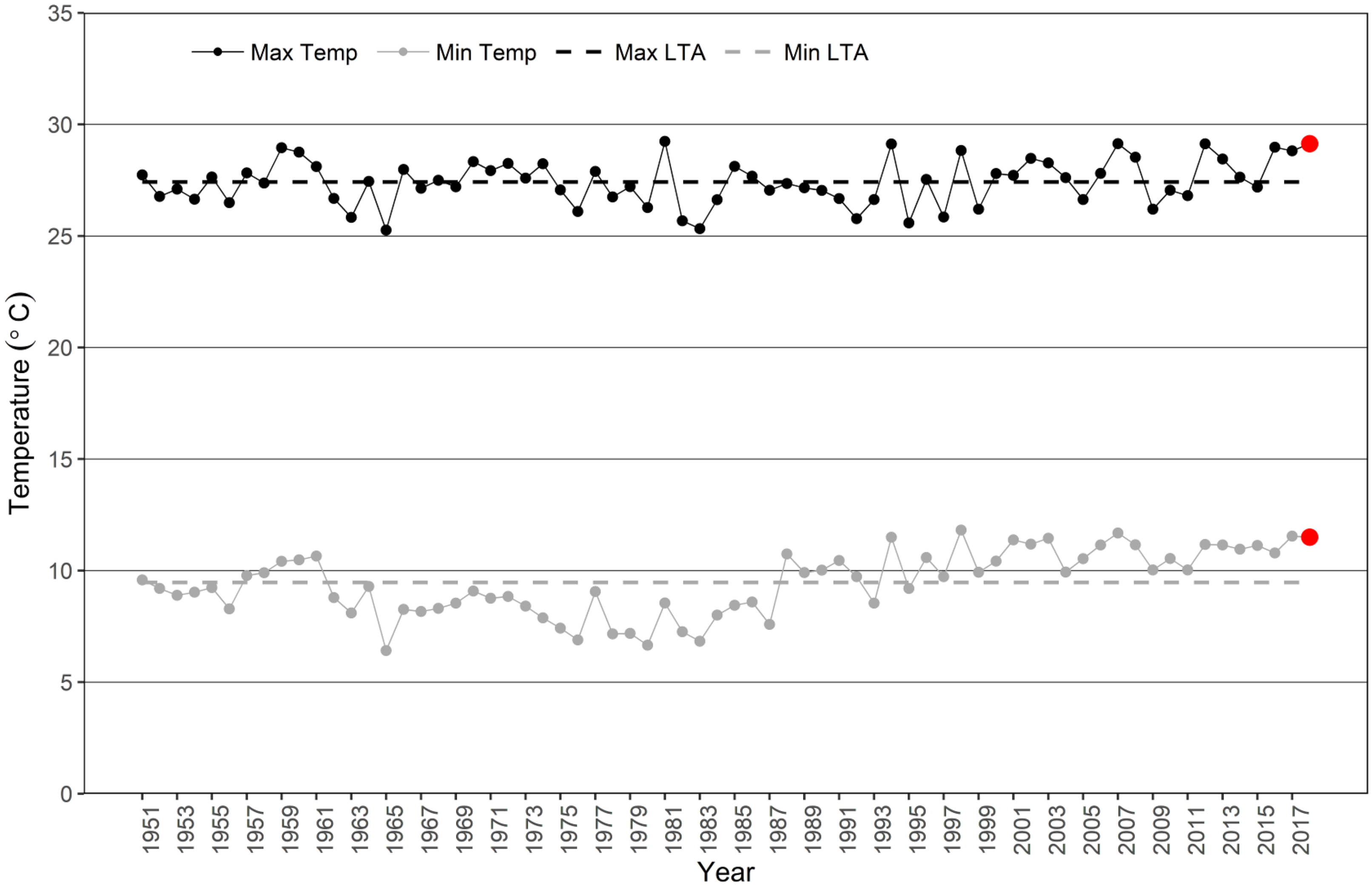

Figure 3.2-7. Average Temperature during Summer Months (June through August)

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.

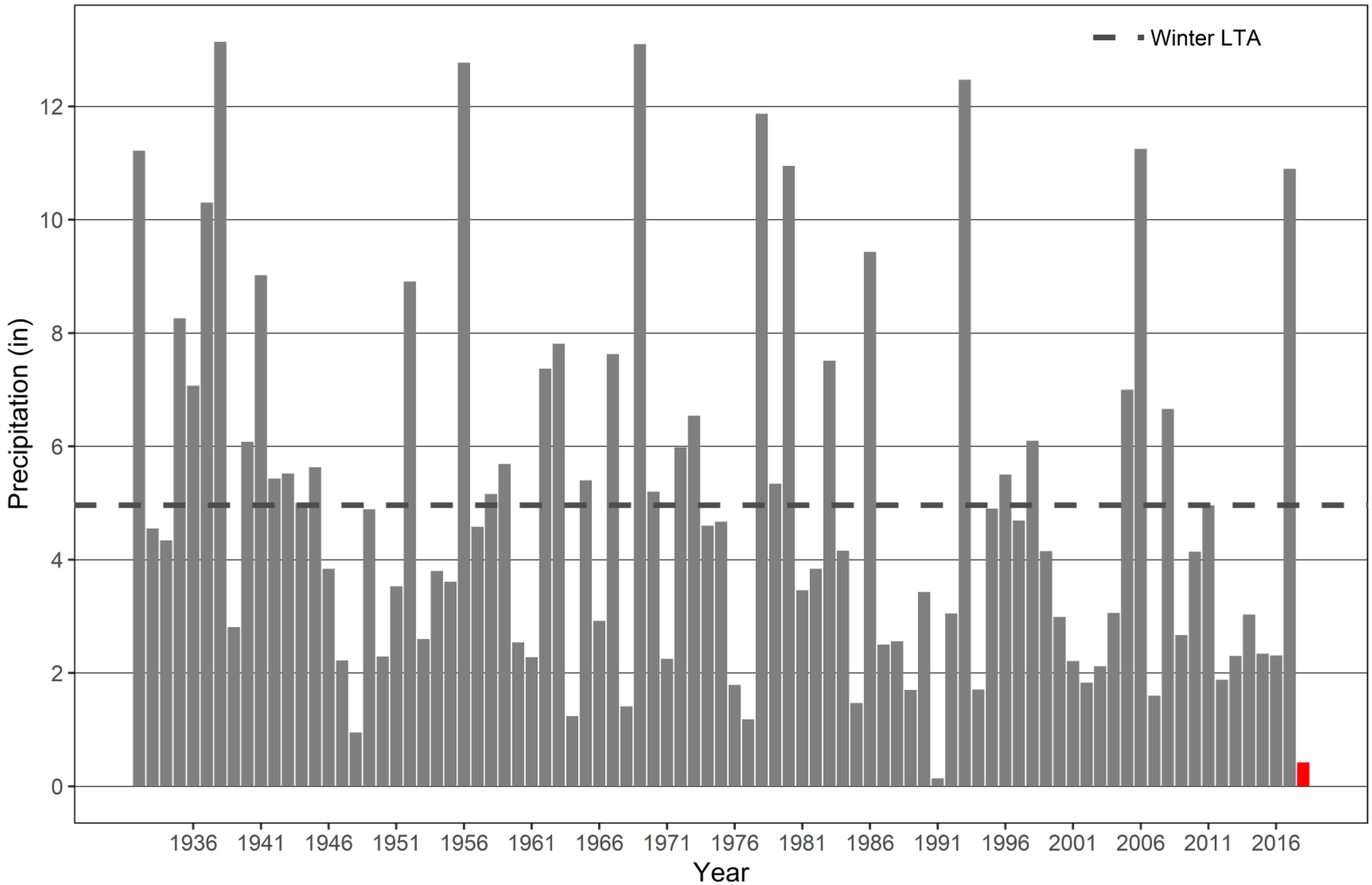

Figure 3.2-8. Total Winter Precipitation (December through February)

Precipitation was recorded at LADWP Cain Ranch since 1932.

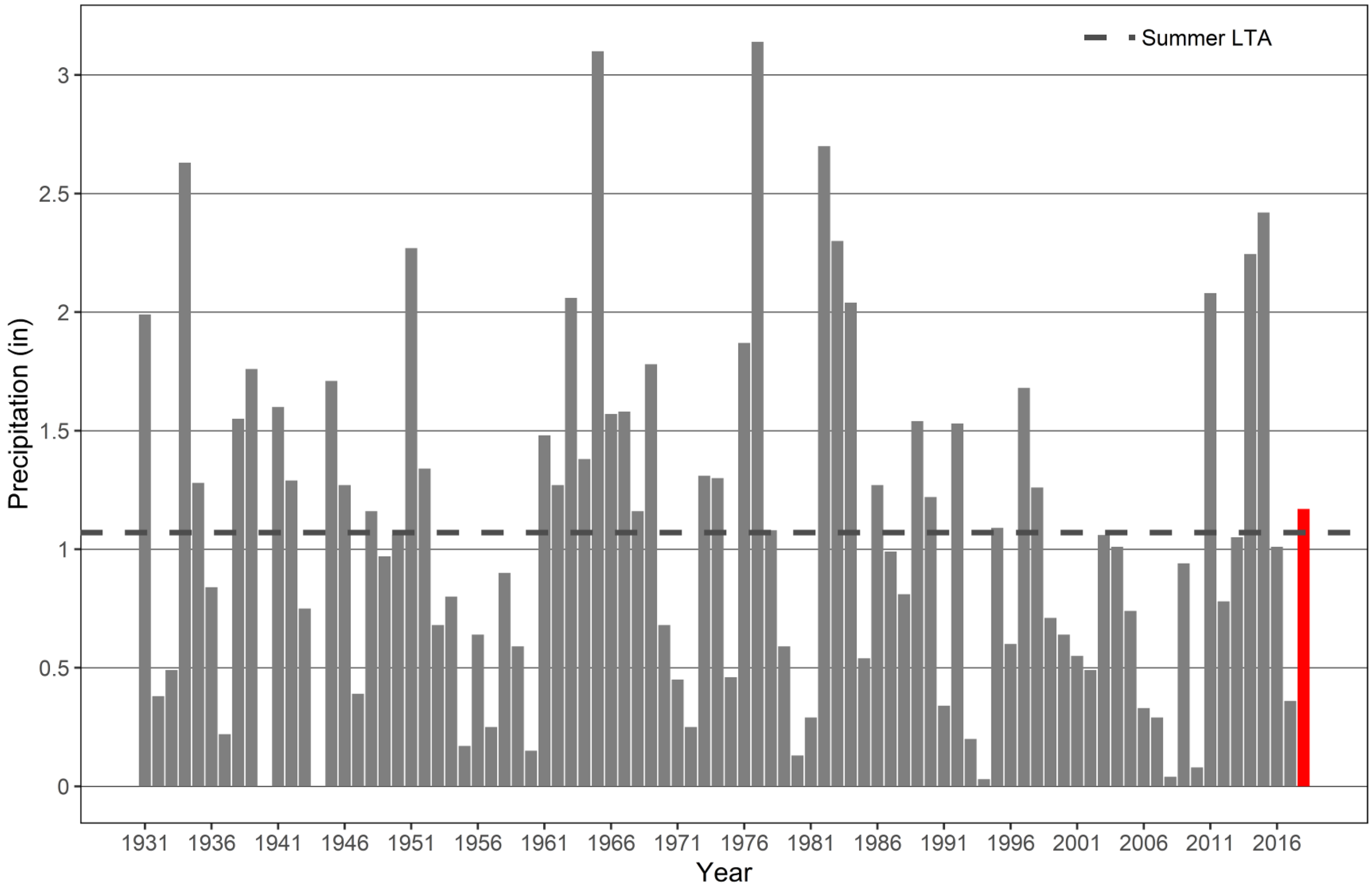

Figure 3.2-9. Total Summer Precipitation (June through August)

Precipitation was recorded at LADWP Cain Ranch since 1932

Since 1998 and before the winter of 2016-17, only three winters showed precipitation above the long-term average of 86 years (5.0 in); 2004-5, 2005-6, and 2007-8 (Figure 3.1-1). The average winter precipitation for the past 10 years (2007 through 2016), excluding 2011 has been 8.3 inches, 75% of the long-term average. Since 1990, only eight years show precipitation above the long-term average; however, four out of five summers show precipitation close to or above the long-term average during the severe drought between 2012 and 2016.

Physical and Chemical

Mono Lake Surface Elevation

The average monthly surface elevation of Mono Lake in January 2018 was 6381.2 feet, much higher than the average January elevation of 6,377.2 feet in 2017 (Figure 3.2.10). Water Year 2016-17 was second wettest on record in terms of input from two major tributaries (Rush and Lee Vining Creeks); 308% of the long term average. The lake level rose 4 feet from January to the year's peak at 6381.2 feet in September, and the year ended at 6381.1 feet. Runoff during Water Year 2017-2018 was 91% of the long-term average (122.247 acre-feet between 1935 and 2018) and ranked 44th out of 85 years. Mono Lake input from Rush and Lee Vining Creeks between October 2017 and September 2018 was 97,022 AF. This value corresponds to 100% of Normal relative to the long-term average starting in 1982. Due to the above normal input, the lake level remained above 6,381 feet until November when the lake level dropped to 6,380.9 feet and remained so until the end of the year. It appears that the combined input of 100% of Water Year Normal flow in Rush and Lee Vining Creeks is necessary to maintain the lake level steady throughout the year at the lake level of around 6,381 feet; however, the combined input of 100% of Normal is not enough to raise the lake level.

Transparency

Transparency of Mono Lake during the summer improved from 0.93 m in June to 2.54 m in July to 3.53 m in August (Table 3.2.2. Figure 3.2.11). Transparency from February through June remained below 1 m. As *Artemia* grazing reduced midsummer phytoplankton, lake-wide transparency and Secchi depth improved between June and August. The duration of improved transparency was shorter in 2018 compared to 2017 during which improved transparency lasted from July to October. Further, the magnitude of transparency decreased in 2018 compared to 2017 as the maximum lake-wide depth decreased to 3.53 m in 2018 from 5.78 m in 2017. A combination of lower *Artemia* population abundance and lower Mono Lake input may be the reasons for the deterioration of lake clarity.

Beginning in 2014, maximum transparency progressively worsened each year during the driest five year period of runoff on record: 1.5 m in 2014, 0.9 m in 2015 and 0.6 m in 2016. This trend, however, was reversed in 2017 due to an almost record-breaking runoff even though Secchi depths still lagged behind historical values (Figure 3.2.12, Figure 3.2.13, and Figure 3.2.14). The annual maximum Secchi reading in 2018 was better than values found between 2014 and 2016, but remained below the historical average of 7.13 m. The input flow peaked in June 7 with estimated combined flow[1] from Rush and Lee Vining Creeks of 503 cfs, and the exceeding probability associated with this flow was approximately 37% or 2.8 years recurrence interval based on daily flow data available since 1990. The peak inflow could have been higher if not for the premature peak flow of Lee Vining Creek, which took place on April 8. The annual peak flow occurred two weeks earlier than 2017 and magnitude was one third of that observed in 2017. A combination of the above two factors may have contributed to improvement in clarity during 2018 monitoring.

[1] Combined flow between Rush and Lee Vining Creeks.

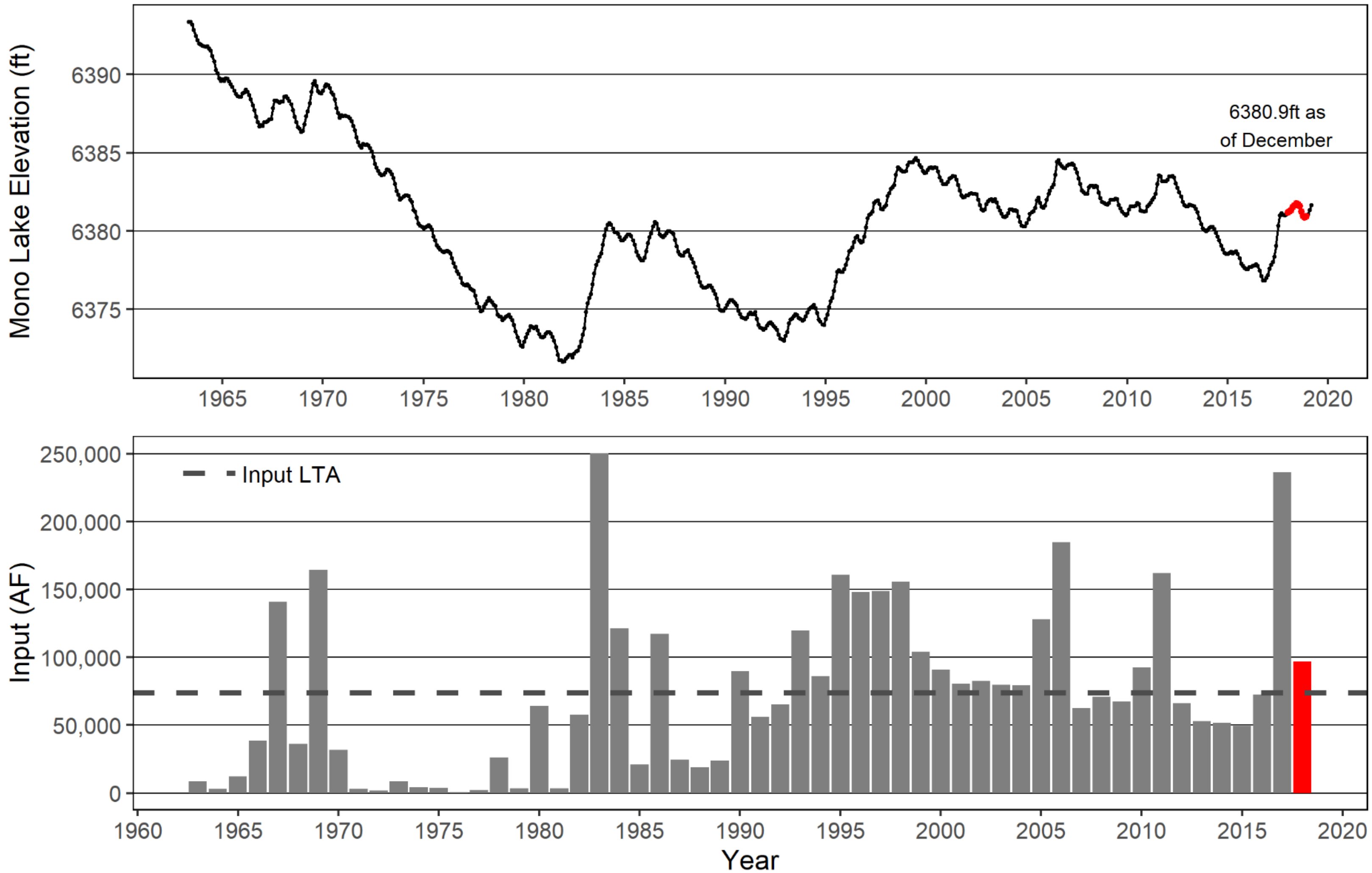

Figure 3.2-10. Mono Lake Surface Elevation (top) and Combined Inflow of Rush and Lee Vining Creeks (bottom)

Mono Lake elevation and input data since 1967 were presented as monthly flow volume of all tributaries to Rush Creek did not become available until 1967

Table 3.2-2. Secchi Depths (m), February-December in 2018

	Sampling Month										
Station	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Western Sector											
1	0.7	0.8	0.5		1	2.9	3.3		0.8	0.8	0.7
2	0.7	0.7	0.5		1	2.8	3.5	2	0.9	0.7	0.6
3	0.6	0.87	0.6	0.5	0.9	2.7	3.7	1.4	0.75	0.8	0.6
4	0.7	0.8	0.6	0.5	1	2.5	3.8	1.8	1	0.9	0.8
5	0.8	0.7	0.4	0.5	1	3.2	3.7	1.6	0.9	0.9	0.5
6	0.7	0.7	0.4	0.6	0.95	2.3	3.5	1.7	0.8	0.9	0.5
AVG	0.70	0.76	0.50	0.53	0.98	2.73	3.58	1.70	0.86	0.83	0.62
SE	0.03	0.03	0.04	0.03	0.02	0.13	0.07	0.10	0.04	0.03	0.05
Eastern Sector											
7	0.7	0.6	0.5	0.6	0.9	2.6	3.5	1.6	0.9	0.7	0.5
8	0.7	0.7	0.5	0.5	0.95	2.5	3.7	1.7	0.8	0.8	0.5
9	0.7	0.8	0.6	0.6	0.8	2	3	1.7	0.9	0.7	0.5
10	0.6	0.8	0.45	0.5	0.9	2.2	3.8		1	0.8	0.6
11	0.6	0.6	0.4	0.5	0.9	2.3	3.4	1.9	1	0.7	0.5
12	0.6	0.7	0.5		0.9	2.5	3.4		0.85	0.9	0.6
AVG	0.65	0.70	0.49	0.54	0.89	2.35	3.47	1.73	0.91	0.77	0.53
SE	0.02	0.04	0.03	0.02	0.02	0.09	0.11	0.06	0.03	0.03	0.02
Total Lakewide											
AVG	0.68	0.73	0.50	0.53	0.93	2.54	3.53	1.71	0.88	0.80	0.58
SE	0.02	0.02	0.02	0.02	0.02	0.09	0.07	0.06	0.02	0.02	0.03

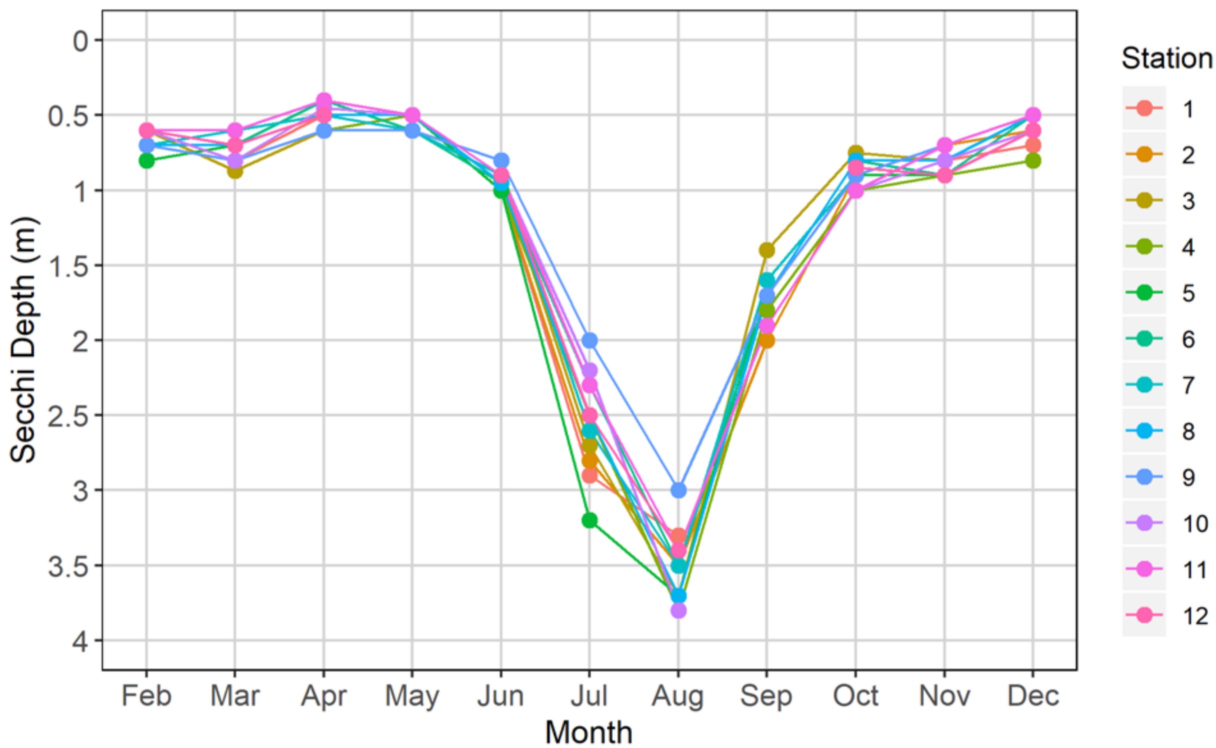

Figure 3.2-11. Lake-wide Secchi Depths in 2018

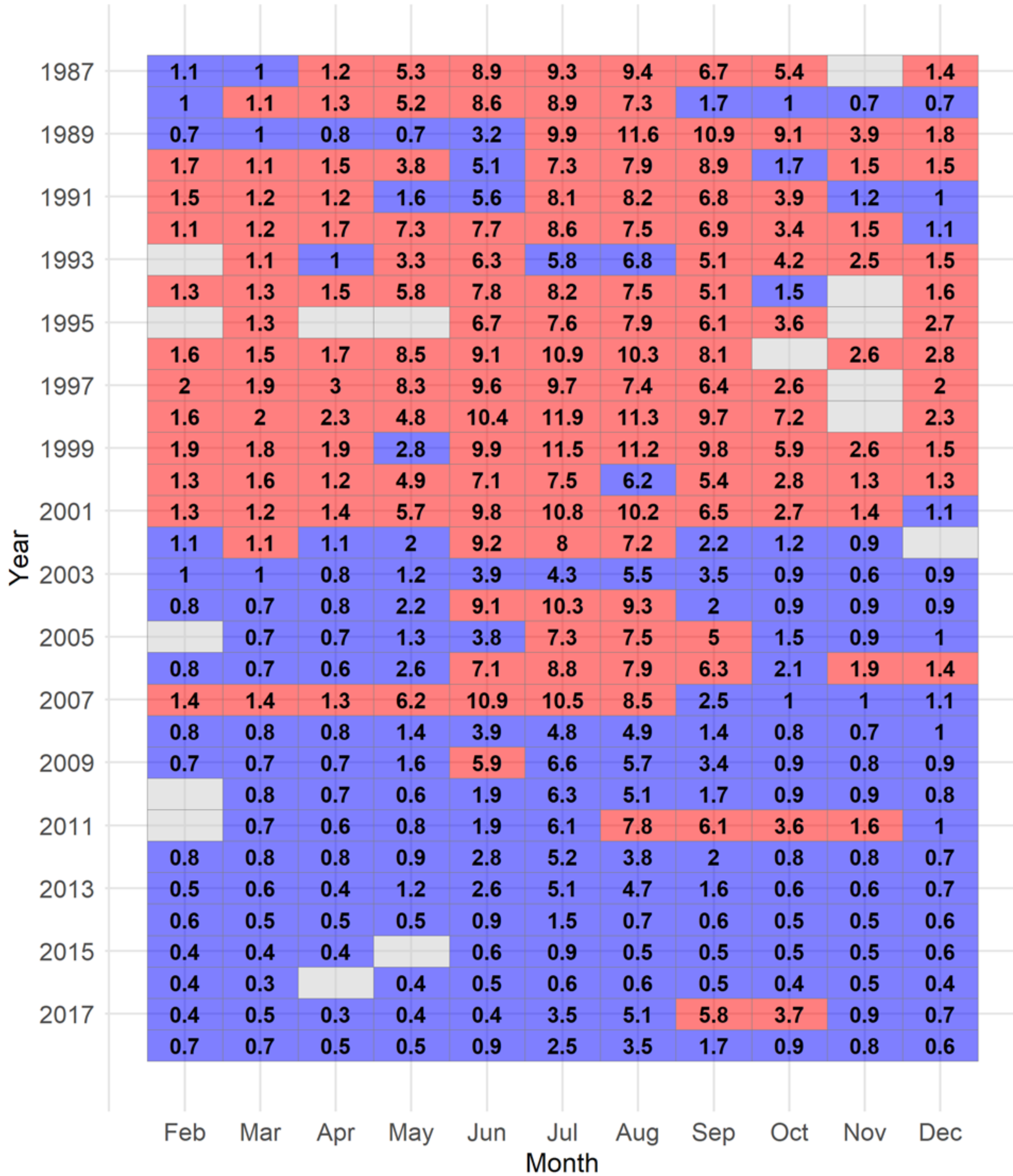

Figure 3.2-12. Lake-wide Average Secchi Depths (m) of 12 Stations

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.

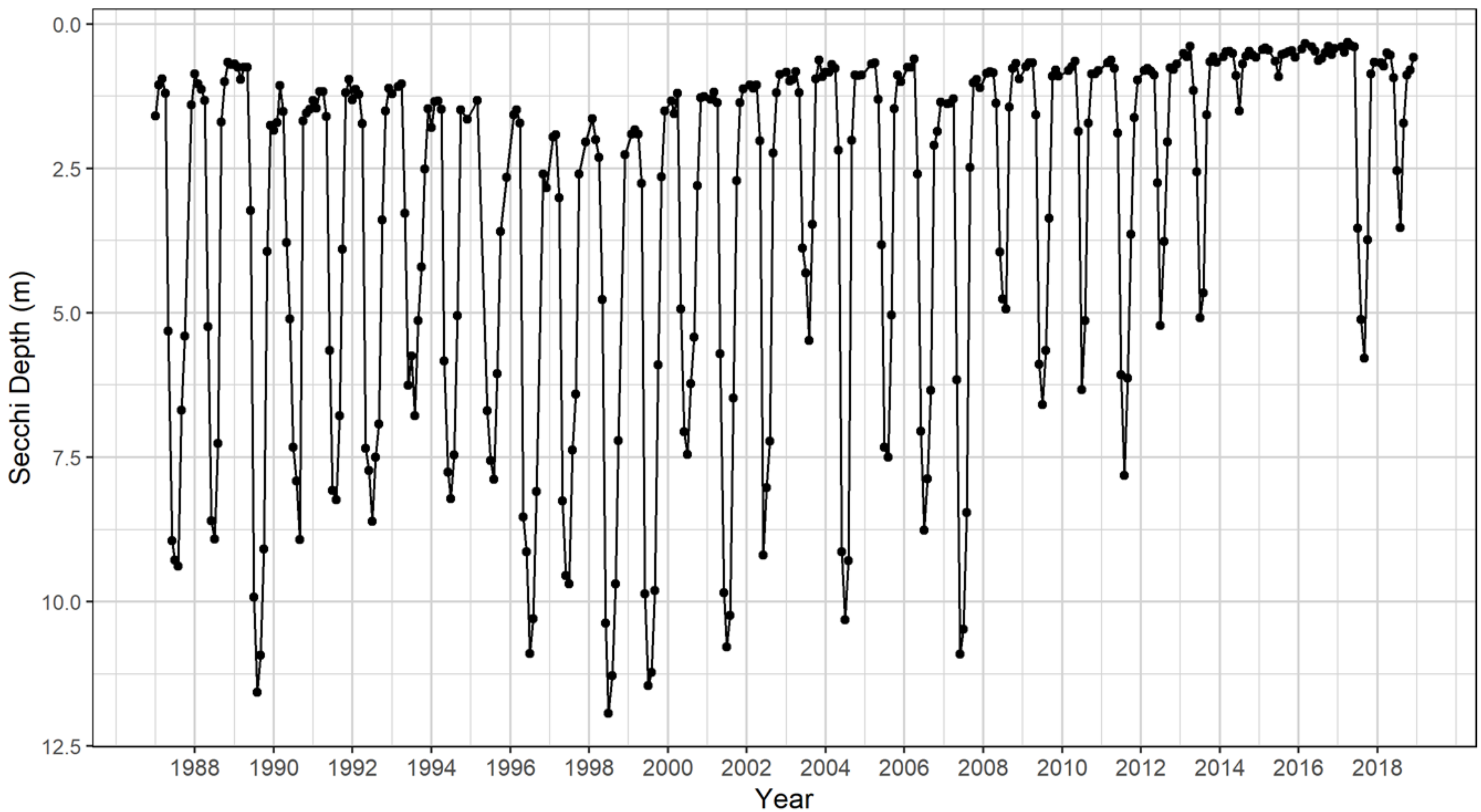

Figure 3.2-13. A Time Series Plot of Lake-wide Average of Secchi Depths (m)

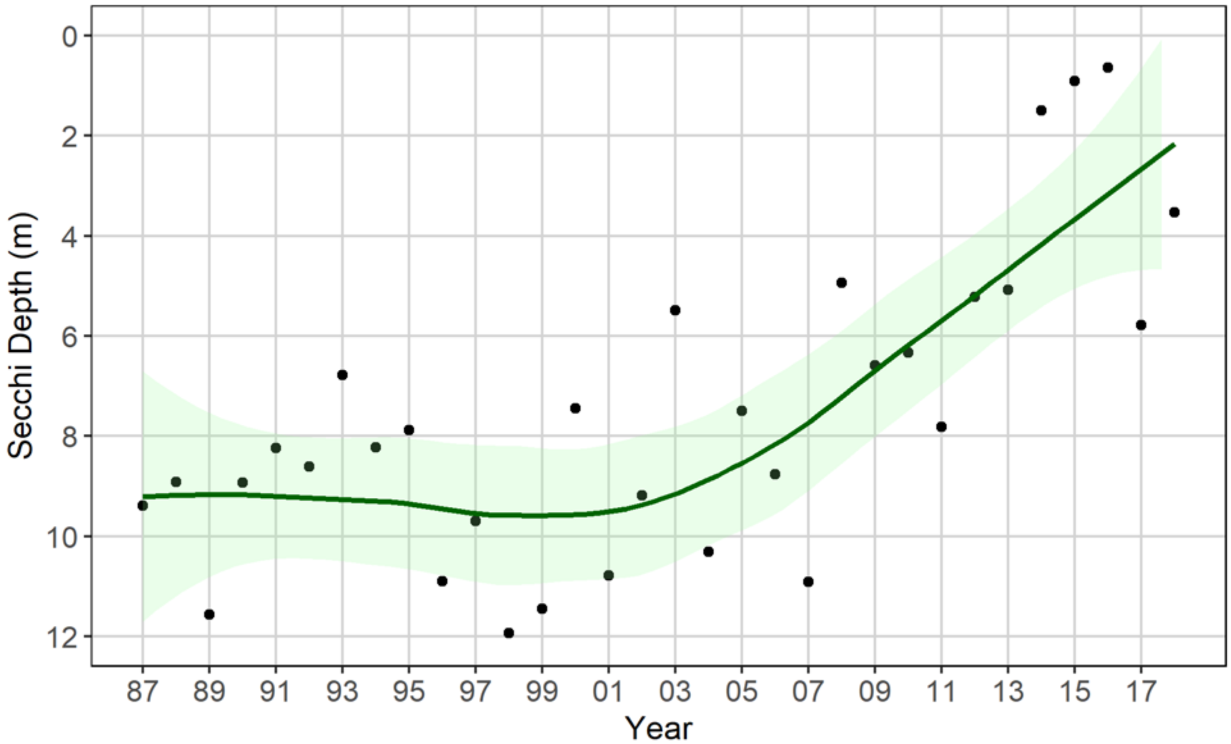

Figure 3.2-14. Annual Maximum Readings of Secchi Depths (m)

Water Temperature

The water temperature data from Station 6 indicate that in 2018, Mono Lake started to become thermally stratified in spring and remained so until December when the lake became isothermal (Table 3.2.3, Figure 3.2.15). By mid-June, the thermocline formed at 5 to 6 m (as indicated by the greater than 1°C change per meter depth) and remained between 6 and 15 m into November even though the thermal stratification became weaker with time. Mono Lake became isothermal in December.

Average water temperature in the epilimnion and hypolimnion remained mostly below normal in summer but above normal in spring and winter months in 2018 (Figure 3.2.16, Figure 3.2.17). A very high February epilimnion water temperature is most likely due to a very warm January and a lower than normal epilimnion water temperature is most likely due to a cooler and wetter March (Figure 3.2.18). In spite of the monthly average air temperature between April and September being above the long-term average, the epilimnion water temperature remained below normal. In November, however, the monthly average air temperature was below normal, resulting in a lower epilimnion water temperature. Because Mono Lake never quite became isothermal in 2017, the hypolimnion water temperature was higher than normal in December of 2017 and this effect remained into the summer of 2018. The epilimnion water temperature remained mostly below the long-term average, such that warming of the upper part of the hypolimnion appeared to be limited. The limited warming, in turn, resulted in lower than normal hypolimnion water temperature. Mono Lake also remained stratified in 2018, potentially helping to keep the hypolimnion water temperature lower.

Table 3.2-3. Water Temperature (°C) at Station 6, February-December in 2018

Depth (m)	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	3.9	2.5	7.5	13.8	19.0	22.6	21.2	17.5	13.0	9.5	5.6
2	3.7	2.4	7.4	13.4	18.8	22.4	21.3	17.7	13.0	9.5	5.6
3	3.6	2.4	7.4	13.4	17.8	22.4	21.3	18.0	13.1	9.5	5.6
4	3.6	2.4	7.4	12.9	17.1	22.3	21.2	18.2	13.1	9.5	5.6
5	3.7	2.4	7.4	11.9	16.5	22.1	21.2	18.2	13.1	9.6	5.6
6	3.7	2.4	7.5	11.3	14.0	21.3	21.2	18.2	13.1	9.6	5.6
7	3.8	2.5	7.5	10.4	12.0	19.6	20.7	18.2	13.1	9.6	5.7
8	3.7	2.4	7.5	9.4	10.2	15.1	19.1	18.2	13.1	9.6	5.6
9	3.7	2.4	7.5	8.7	8.9	12.4	14.5	18.2	13.1	9.6	5.6
10	3.7	2.4	7.5	8.3	8.4	10.0	10.9	16.3	13.1	9.7	5.6
11	4.2	2.4	7.5	7.7	7.6	8.8	9.4	11.0	13.1	9.7	5.6
12	5.4	3.0	6.2	6.9	7.3	7.8	8.4	9.4	13.1	9.6	5.6
13	6.1	4.6	5.3	6.3	6.7	7.1	7.8	8.1	10.7	9.2	5.6
14	6.4	5.3	5.3	6.0	6.2	6.4	7.1	6.8	7.8	8.8	5.6
15	6.5	5.5	5.4	5.7	5.8	5.9	6.5	6.3	6.6	7.7	6.7
16	6.4	5.8	5.4	5.5	5.7	5.8	6.2	6.0	6.2	6.6	6.7
17	6.2	5.8	5.5	5.5	5.6	5.7	5.9	5.8	5.9	6.1	6.2
18	6.1	5.8	5.5	5.5	5.6	5.7	5.8	5.7	5.8	5.8	5.9
19	6.0	5.8	5.6	5.5	5.6	5.6	5.7	5.7	5.7	5.8	5.8
20	5.8	5.8	5.6	5.5	5.6	5.6	5.7	5.6	5.7	5.7	5.7
21	5.7	5.8	5.6	5.6	5.6	5.6	5.6	5.6	5.7	5.7	5.7
22	5.6	5.7	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.7	5.7
23	5.6	5.7	5.7	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
24	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
25	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
26	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
27	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
28	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
29	5.2	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
30	5.2	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.5	5.6	5.6
31	5.2	5.4	5.5	5.6	5.6	5.6	5.5	5.5	5.5	5.5	5.5
32	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
33	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
34	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
35	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
36	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
37	5.0	5.3	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
38	5.0	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
39	5.0	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
40	5.0	5.2	-	-	-	-	-	-	-	-	5.5

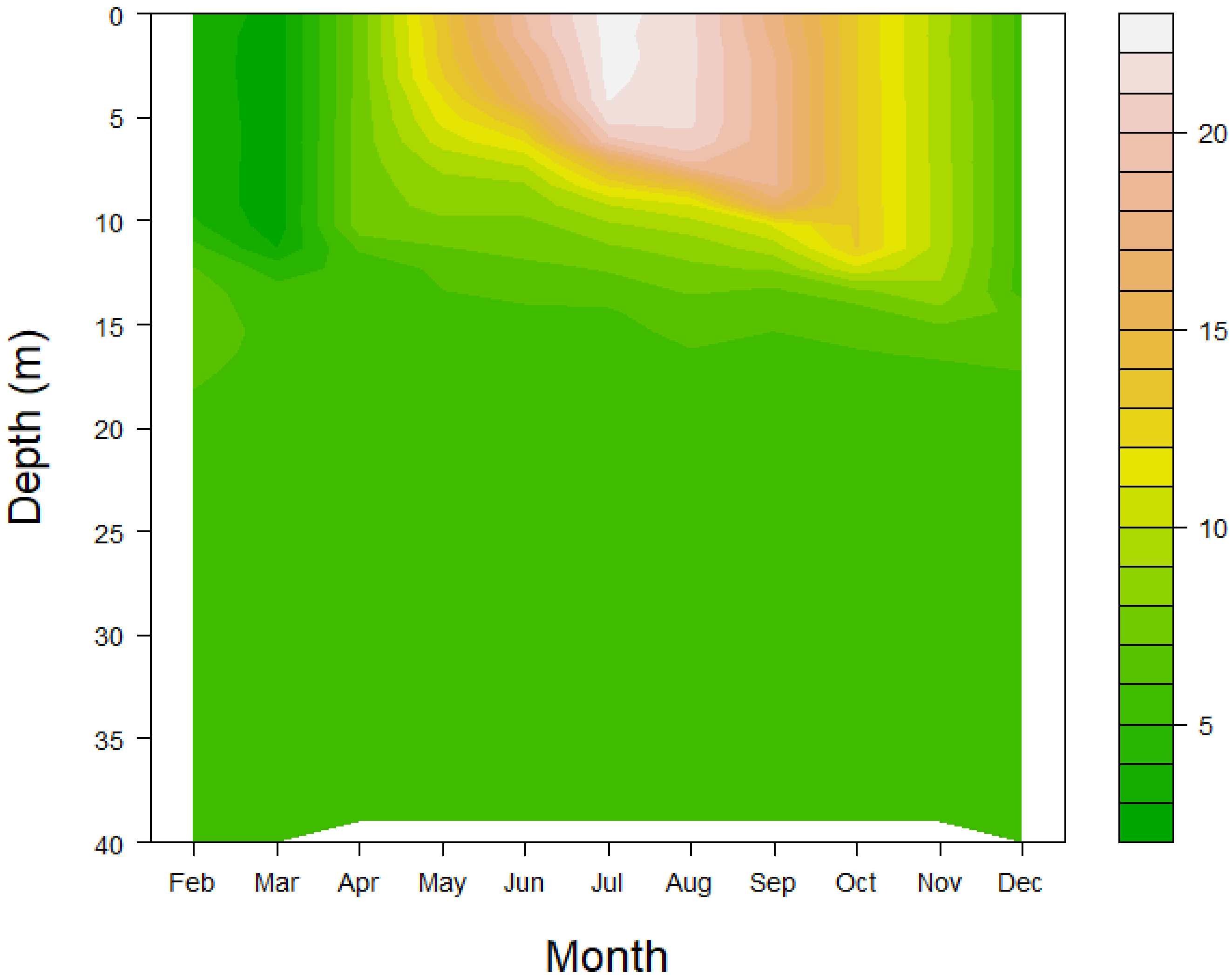

Figure 3.2-15. Water Temperature Profile (°C) at Station 6, February-December in 2018

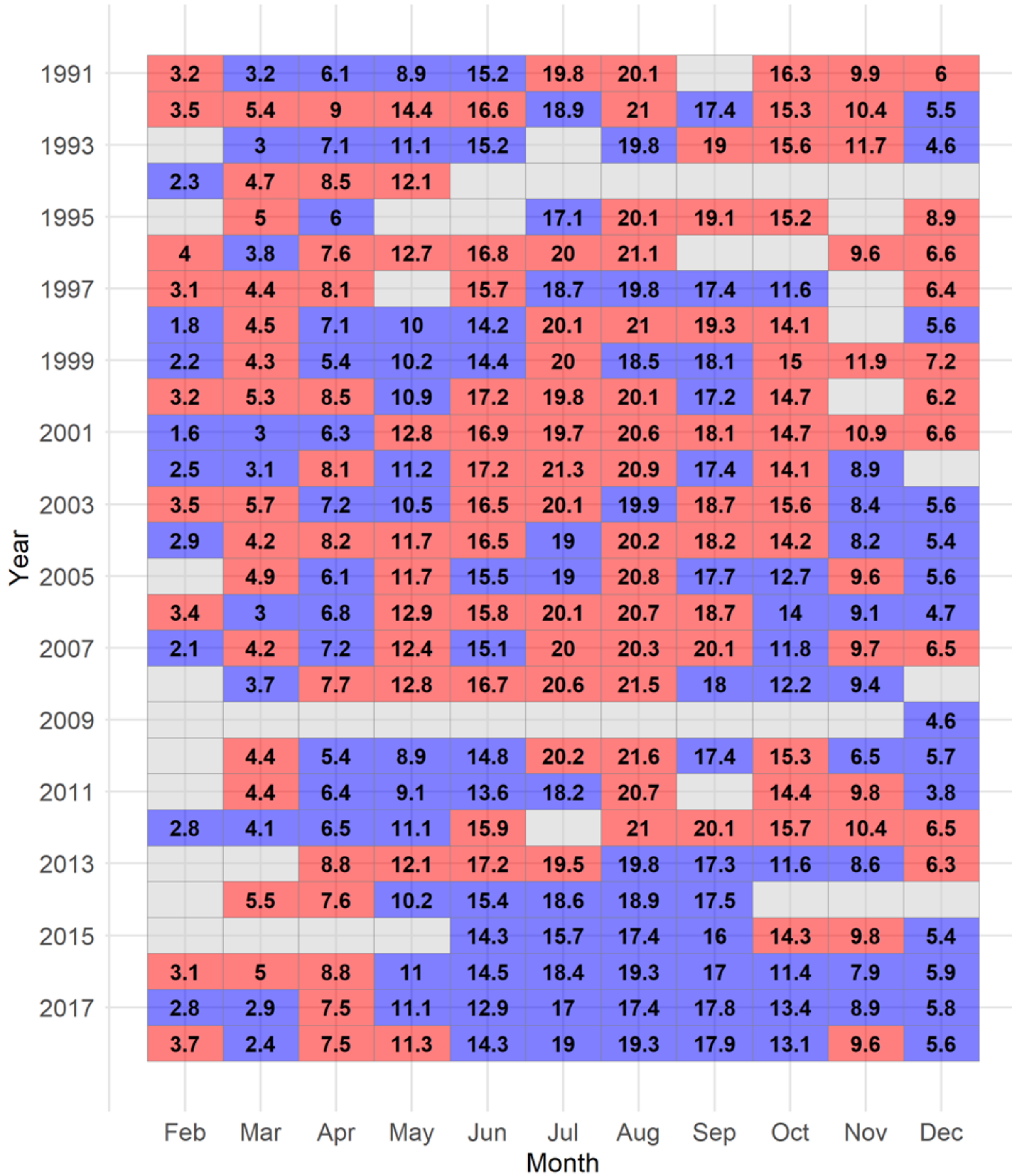

Figure 3.2-16. Average Water Temperature (°C) between 1 and 10 m at Station 6

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

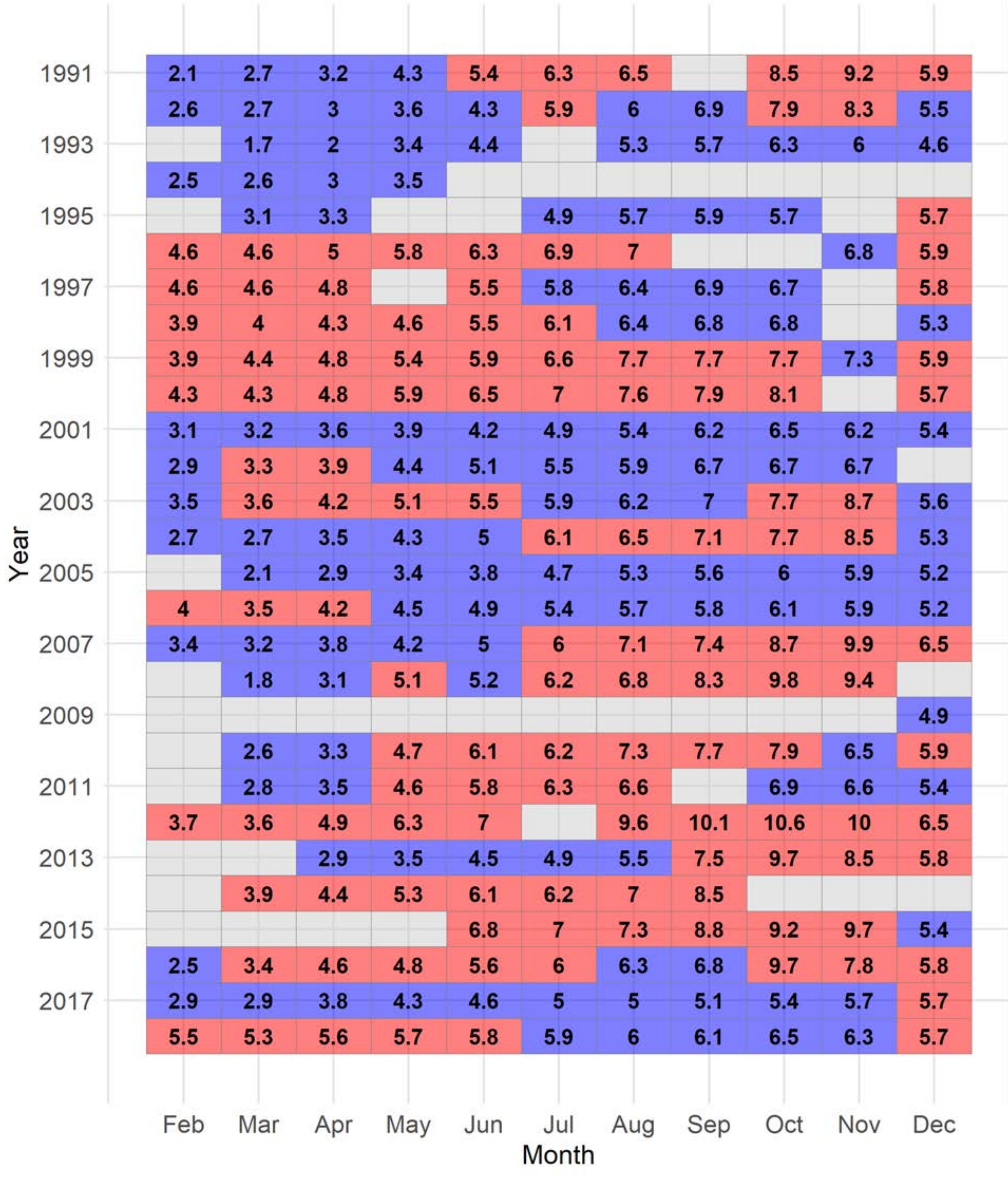

Figure 3.2-17. Average Water Temperature (°C) between 11 and 38 m at Station 6

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.

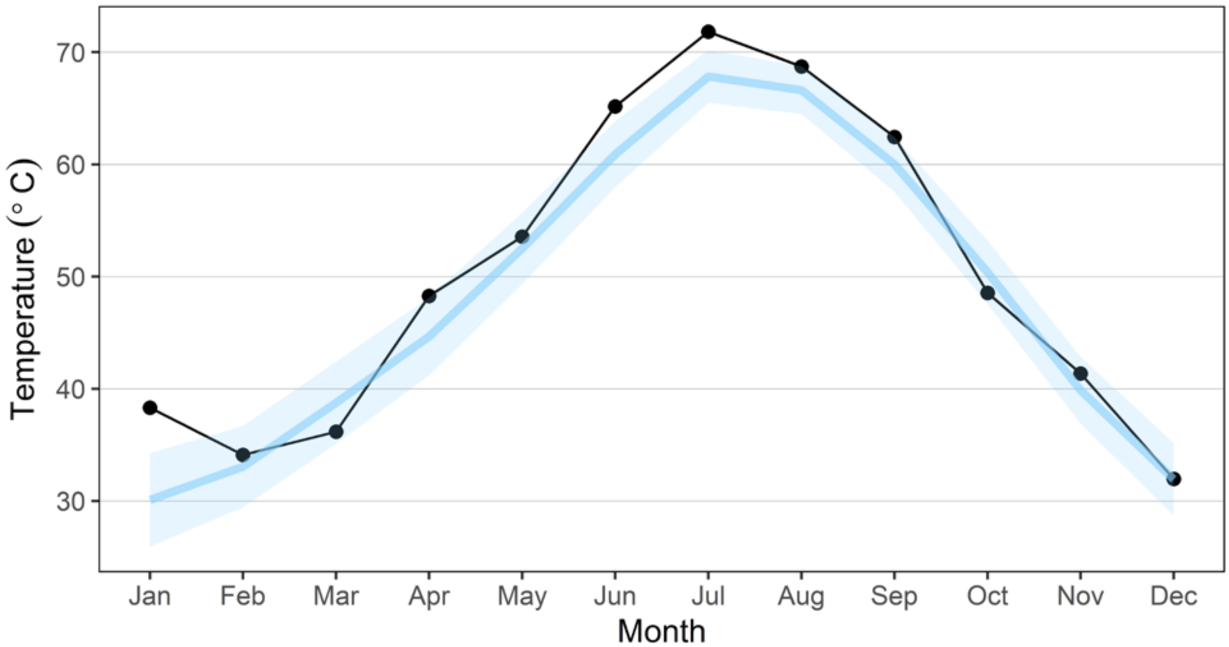

Figure 3.2-18. Monthly Average Temperature in 2018 compared to the Long Term Average

The blue line indicates the long-term average since 1951 while the shaded band indicates one standard deviation above and below the mean. 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

<u>Conductivity</u>

Conductivity data was collected from the CTD field sampling device on a monthly basis. In situ conductivity measurements at Station 6 were corrected for temperature (25˚C) and reported at one meter intervals beginning at one meter in depth down to the lake bottom. The chemocline established in 2017 remained throughout 2018 even though it was weakened at the end of 2018. The chemocline was observed between 10 and 11 m in February, moved up during summer, and re-established around between 13 and 15 m in fall and winter (Table 4, Figure 19). Epilimnetic specific conductivity began to decrease in April with onset of snowmelt driven runoff and peaked in July, approximately a month after the peak input from Rush and Lee Vining Creeks which took place on June 7. The lowest conductivity of any depth in 2018 was 78.1 mS/cm. The largest vertical range in specific conductivity (11.9 mS/cm) was observed in July compared to 17.1 mS/cm observed in September, 2017. A vertical range above 10 mS/cm was found between June and September. The chemocline remained between 14 and 15 m in December.

Table 3.2-4. Conductivity (mS/cm at 25°C) at Station 6, February-December in 2018

Depth (m)	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	3.9	2.5	7.5	13.8	19.0	22.6	21.2	17.5	13.0	9.5	5.6
2	3.7	2.4	7.4	13.4	18.8	22.4	21.3	17.7	13.0	9.5	5.6
3	3.6	2.4	7.4	13.4	17.8	22.4	21.3	18.0	13.1	9.5	5.6
4	3.6	2.4	7.4	12.9	17.1	22.3	21.2	18.2	13.1	9.5	5.6
5	3.7	2.4	7.4	11.9	16.5	22.1	21.2	18.2	13.1	9.6	5.6
6	3.7	2.4	7.5	11.3	14.0	21.3	21.2	18.2	13.1	9.6	5.6
7	3.8	2.5	7.5	10.4	12.0	19.6	20.7	18.2	13.1	9.6	5.7
8	3.7	2.4	7.5	9.4	10.2	15.1	19.1	18.2	13.1	9.6	5.6
9	3.7	2.4	7.5	8.7	8.9	12.4	14.5	18.2	13.1	9.6	5.6
10	3.7	2.4	7.5	8.3	8.4	10.0	10.9	16.3	13.1	9.7	5.6
11	4.2	2.4	7.5	7.7	7.6	8.8	9.4	11.0	13.1	9.7	5.6
12	5.4	3.0	6.2	6.9	7.3	7.8	8.4	9.4	13.1	9.6	5.6
13	6.1	4.6	5.3	6.3	6.7	7.1	7.8	8.1	10.7	9.2	5.6
14	6.4	5.3	5.3	6.0	6.2	6.4	7.1	6.8	7.8	8.8	5.6
15	6.5	5.5	5.4	5.7	5.8	5.9	6.5	6.3	6.6	7.7	6.7
16	6.4	5.8	5.4	5.5	5.7	5.8	6.2	6.0	6.2	6.6	6.7
17	6.2	5.8	5.5	5.5	5.6	5.7	5.9	5.8	5.9	6.1	6.2
18	6.1	5.8	5.5	5.5	5.6	5.7	5.8	5.7	5.8	5.8	5.9
19	6.0	5.8	5.6	5.5	5.6	5.6	5.7	5.7	5.7	5.8	5.8
20	5.8	5.8	5.6	5.5	5.6	5.6	5.7	5.6	5.7	5.7	5.7
21	5.7	5.8	5.6	5.6	5.6	5.6	5.6	5.6	5.7	5.7	5.7
22	5.6	5.7	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.7	5.7
23	5.6	5.7	5.7	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
24	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
25	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
26	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
27	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
28	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
29	5.2	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
30	5.2	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.5	5.6	5.6
31	5.2	5.4	5.5	5.6	5.6	5.6	5.5	5.5	5.5	5.5	5.5
32	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
33	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
34	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
35	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
36	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
37	5.0	5.3	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
38	5.0	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
39	5.0	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
40	5.0	5.2	-	-	-	-	-	-	-	-	5.5

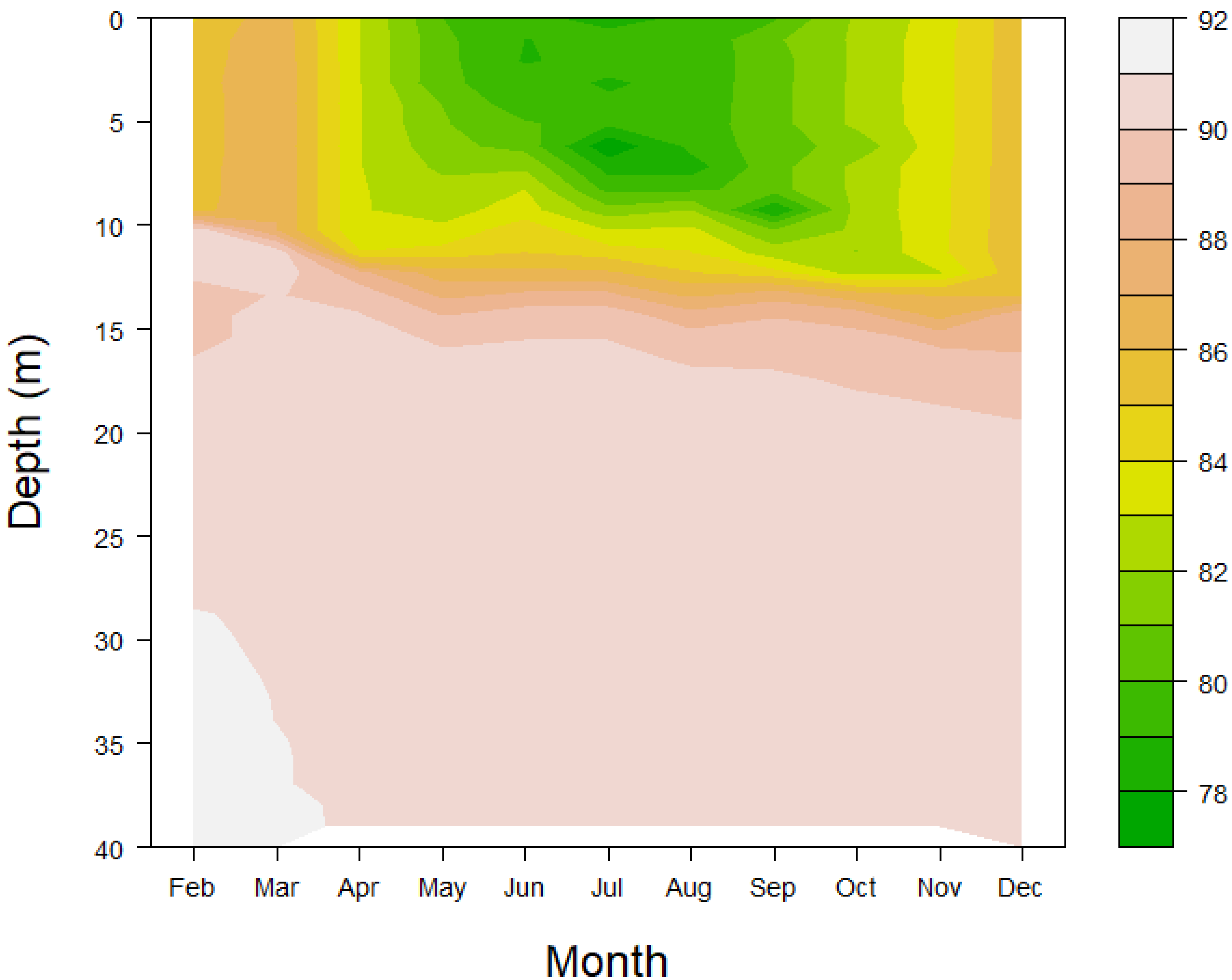

Figure 3.2-19. Conductivity Profile (mS/cm) at Station 6, February-December in 2018

<u>Salinity</u>

Salinity expressed in g/L was calculated based on the equation presented by Jellison in past compliance reports and presented in Figure 3.2.20 and Figure 3.2.21 (see also Figure 3.2-44). Salinity in the epilimnion was found to be lower than normal between April and November, but higher than normal in February, March and December. Even though salinity in February and March was above normal, values were much lower than those observed in 2016 and again 2017 for these 2 months. Salinity in December was higher than normal mainly due to the weakening chemocline resulting from decreased influx of freshwater. Salinity in October and November was below normal but higher than observed in 2017. Salinity in the hypolimnion remained above 90 g/L and higher than normal for all months even though values in 2018 were lower than those in 2017. The trend of increasing hypolimnetic salinity which started at the end of meromixis in 2008 halted in 2018.

Mono Lake water was less salty at shallower depths but continued to remain saltier at deeper depths. Due to the extremely dry conditions that persisted between 2012 and 2015 the lake level dropped from 6,383.5 feet in May 2012 to 6,377.0 feet in December 2016. During the same period, the salinity level increased from 72.6 g/L (July 2012) to 91.4 g/L in the epilimnion and from 80.6 g/L (July 2012) to 91.5 g/L in the hypolimnion. As a result, the salinity level at the beginning of 2017 was higher than any other monitoring years since 1991. As a result of the almost record breaking runoff in 2017, salinity was observed to decline in 2018. A runoff of similar magnitude to 2017 is expected in 2019, which would strengthen the chemocline and help to lower salinity further.

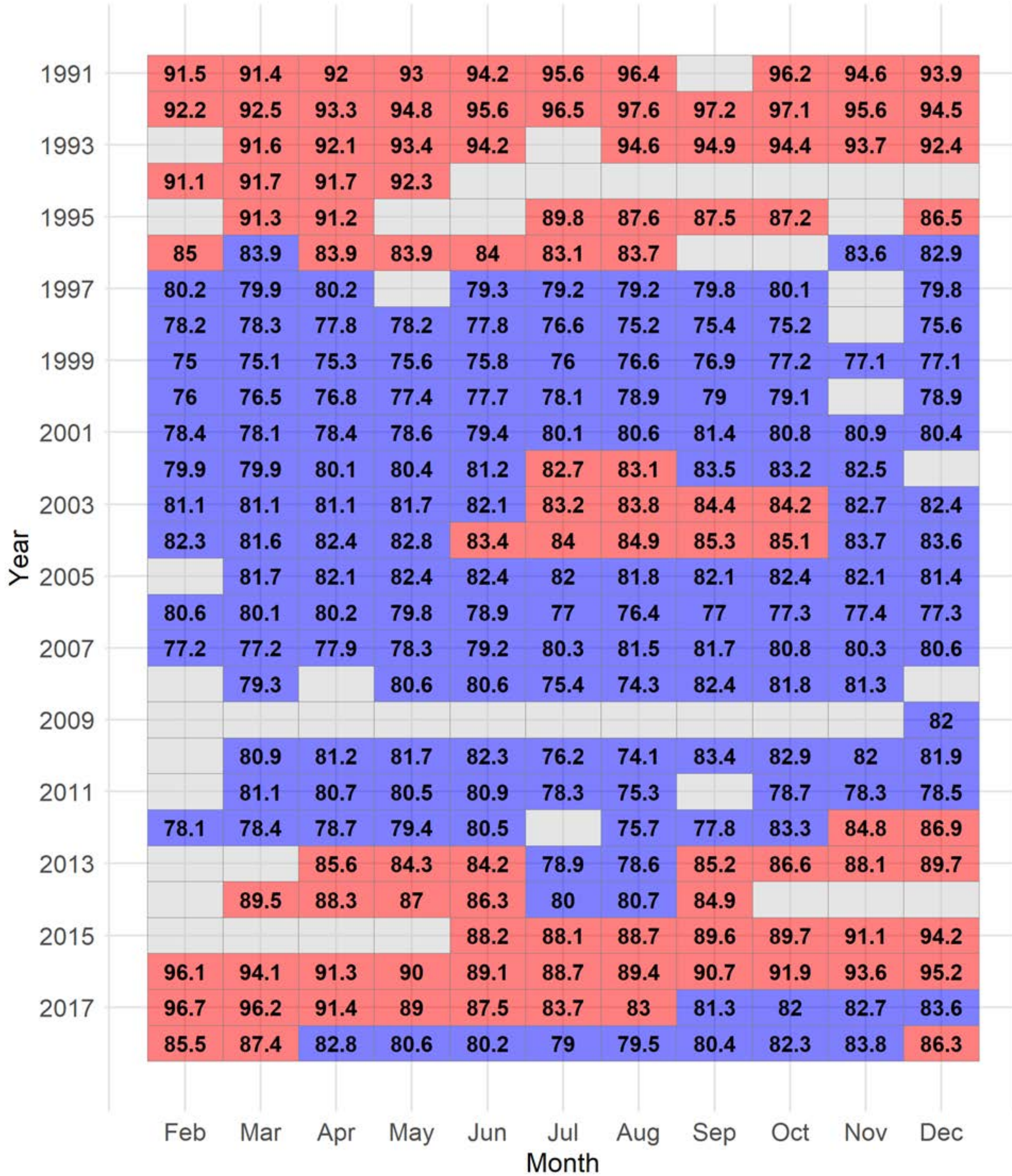

Figure 3.2-20. Average Salinity (g/L) between 1 and 10 m at Station 6

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.

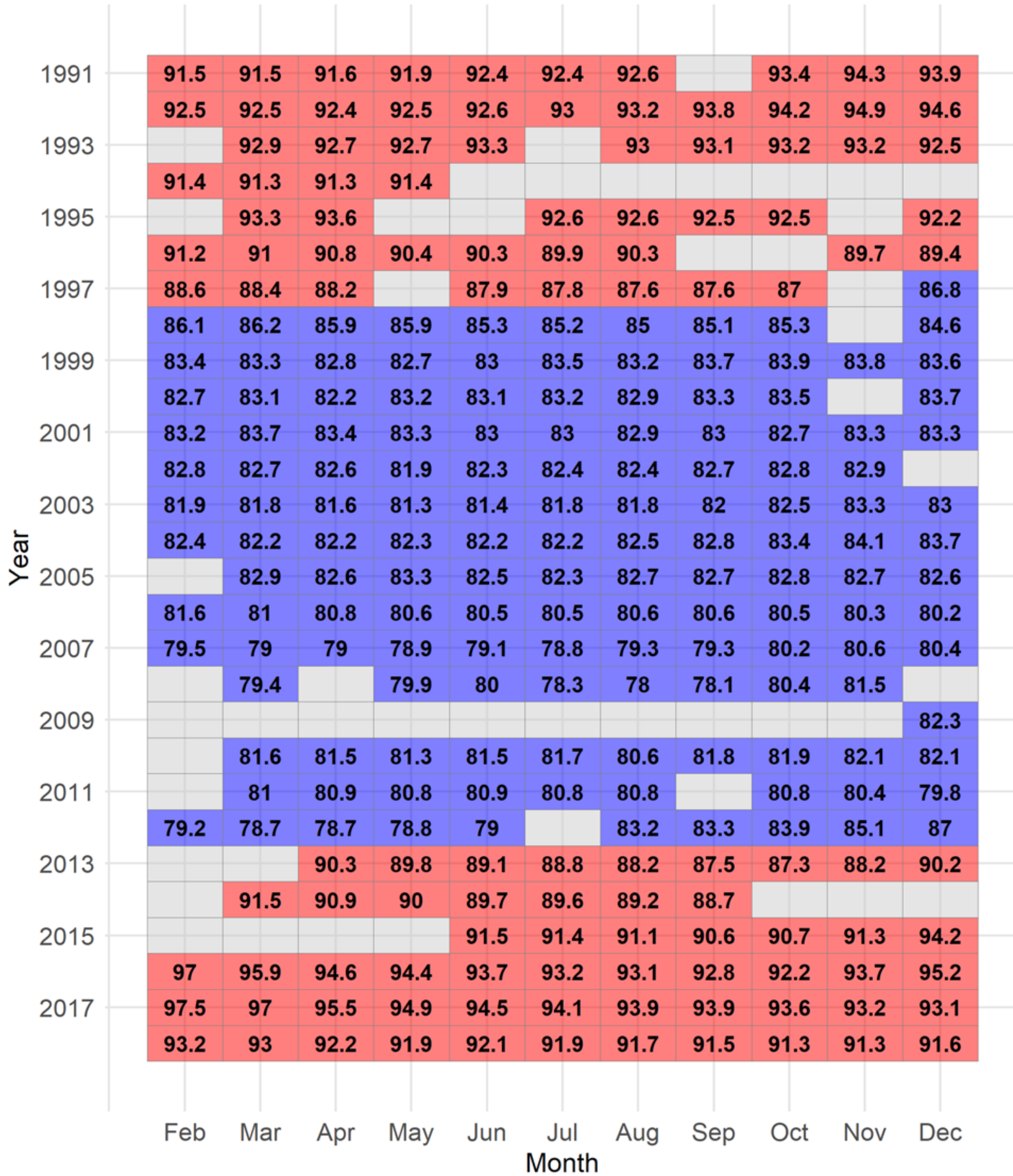

Figure 3.2-21. Average Salinity (g/L) between 11 and 38 m at Station 6

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

In 2018, dissolved oxygen (DO) concentrations in the upper mixed layer (between 10 and 15 m) remained much higher than the hypolimnion DO (>15 m) as the chemocline persisted throughout the year even though it weakened over time (Table 3.2.5, Figure 3.2.22). The lowest epilimnetic values occurred during the December survey when dissolved oxygen was 3.5–3.6 mg/L in the upper 5 m of the water column. Hypolimnetic DO concentration remained mostly anoxic (<0.5 mg/L) throughout 2018. The anoxic condition was found at 13 m in February, at 12 m in July and 16 m in December. Mono Lake became isothermal in December, but the chemocline persisted throughout 2018, resulting in the anoxic condition remaining below 16 m to the end of the year.

Average DO concentrations in the upper mixing layer (depth between 1 and 15 m) ranged from 4 mg/L in June and September to 7.3 mg/L in March in 2018, and remained above the long term average (Figure 3.2.23). Epilimnion DO levels in 2018 were much higher than those observed in previous 4 years. Elevated epilimnion DO levels appeared to have started in October 2017; coinciding with large influx of freshwater. Reduced salinity along with chemocline preventing dissolve oxygen from mixing downward may have helped phytoplankton populations to thrive. High epilimnion DO levels during meromixis were observed between 1995 and 2001 and again between 2011 and 2012, but not between 2005 and 2007. Below 15 m average DO concentrations remained either suboxic or anoxic throughout 2018 and the average for 2018 was lowest since 1994 (Figure 3.2.24).

Table 3.2-5. Dissolved Oxygen* (mg/L) at Station 6, February-December in 2018

Depth (m)	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	3.9	2.5	7.5	13.8	19.0	22.6	21.2	17.5	13.0	9.5	5.6
2	3.7	2.4	7.4	13.4	18.8	22.4	21.3	17.7	13.0	9.5	5.6
3	3.6	2.4	7.4	13.4	17.8	22.4	21.3	18.0	13.1	9.5	5.6
4	3.6	2.4	7.4	12.9	17.1	22.3	21.2	18.2	13.1	9.5	5.6
5	3.7	2.4	7.4	11.9	16.5	22.1	21.2	18.2	13.1	9.6	5.6
6	3.7	2.4	7.5	11.3	14.0	21.3	21.2	18.2	13.1	9.6	5.6
7	3.8	2.5	7.5	10.4	12.0	19.6	20.7	18.2	13.1	9.6	5.7
8	3.7	2.4	7.5	9.4	10.2	15.1	19.1	18.2	13.1	9.6	5.6
9	3.7	2.4	7.5	8.7	8.9	12.4	14.5	18.2	13.1	9.6	5.6
10	3.7	2.4	7.5	8.3	8.4	10.0	10.9	16.3	13.1	9.7	5.6
11	4.2	2.4	7.5	7.7	7.6	8.8	9.4	11.0	13.1	9.7	5.6
12	5.4	3.0	6.2	6.9	7.3	7.8	8.4	9.4	13.1	9.6	5.6
13	6.1	4.6	5.3	6.3	6.7	7.1	7.8	8.1	10.7	9.2	5.6
14	6.4	5.3	5.3	6.0	6.2	6.4	7.1	6.8	7.8	8.8	5.6
15	6.5	5.5	5.4	5.7	5.8	5.9	6.5	6.3	6.6	7.7	6.7
16	6.4	5.8	5.4	5.5	5.7	5.8	6.2	6.0	6.2	6.6	6.7
17	6.2	5.8	5.5	5.5	5.6	5.7	5.9	5.8	5.9	6.1	6.2
18	6.1	5.8	5.5	5.5	5.6	5.7	5.8	5.7	5.8	5.8	5.9
19	6.0	5.8	5.6	5.5	5.6	5.6	5.7	5.7	5.7	5.8	5.8
20	5.8	5.8	5.6	5.5	5.6	5.6	5.7	5.6	5.7	5.7	5.7
21	5.7	5.8	5.6	5.6	5.6	5.6	5.6	5.6	5.7	5.7	5.7
22	5.6	5.7	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.7	5.7
23	5.6	5.7	5.7	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
24	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
25	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
26	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
27	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
28	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
29	5.2	5.5	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6	5.6
30	5.2	5.4	5.5	5.6	5.6	5.6	5.6	5.6	5.5	5.6	5.6
31	5.2	5.4	5.5	5.6	5.6	5.6	5.5	5.5	5.5	5.5	5.5
32	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
33	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
34	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
35	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
36	5.1	5.3	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
37	5.0	5.3	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
38	5.0	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
39	5.0	5.2	5.4	5.5	5.5	5.5	5.5	5.5	5.5	5.5	5.5
40	5.0	5.2	-	-	-	-	-	-	-	-	5.5

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

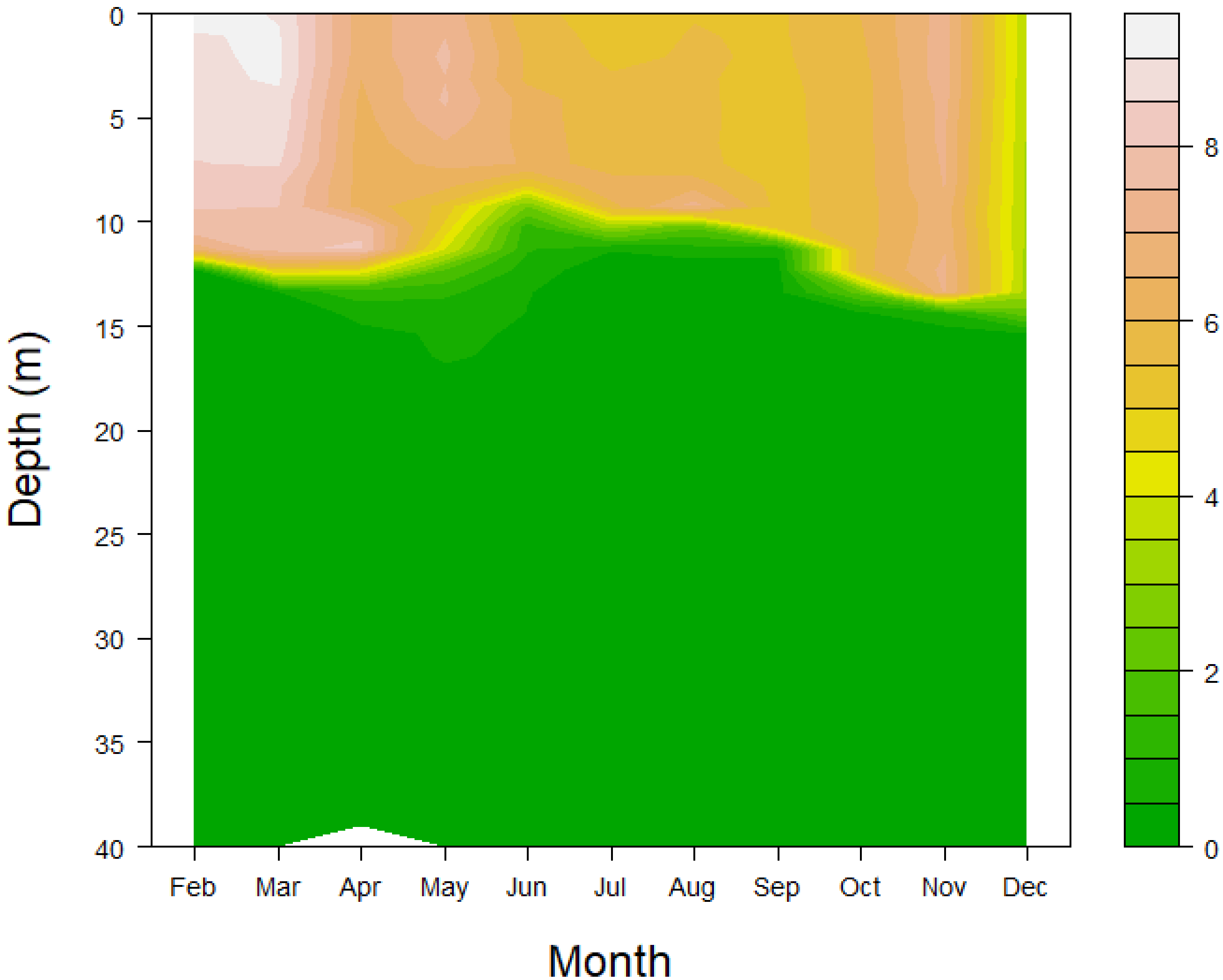

Figure 3.2-22. Dissolved Oxygen (mg/L) Profiles at Station 6, February-December in 2018

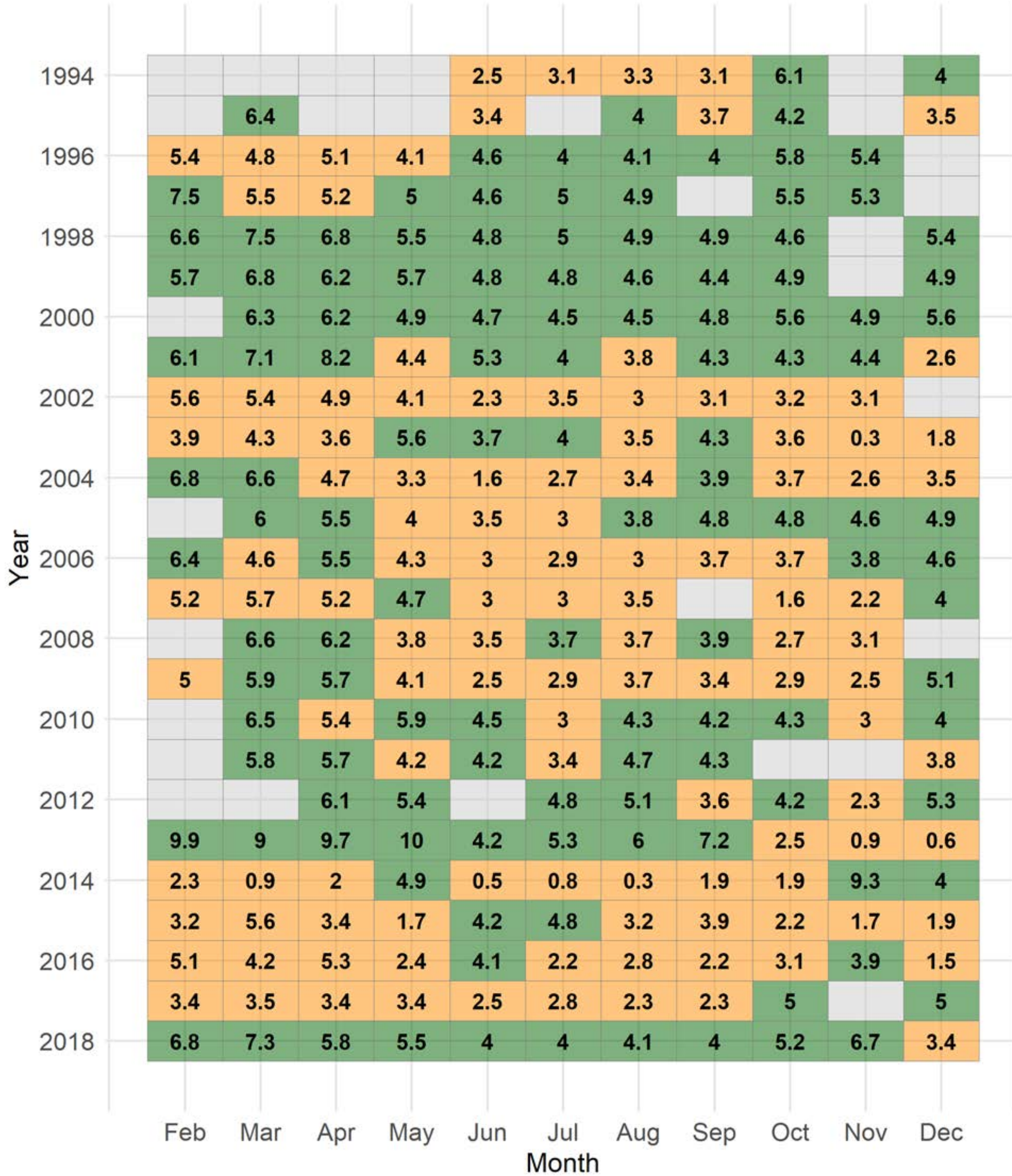

Figure 3.2-23. Average Dissolved Oxygen (mg/L) at Station 6 between 1 and 15 m

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.

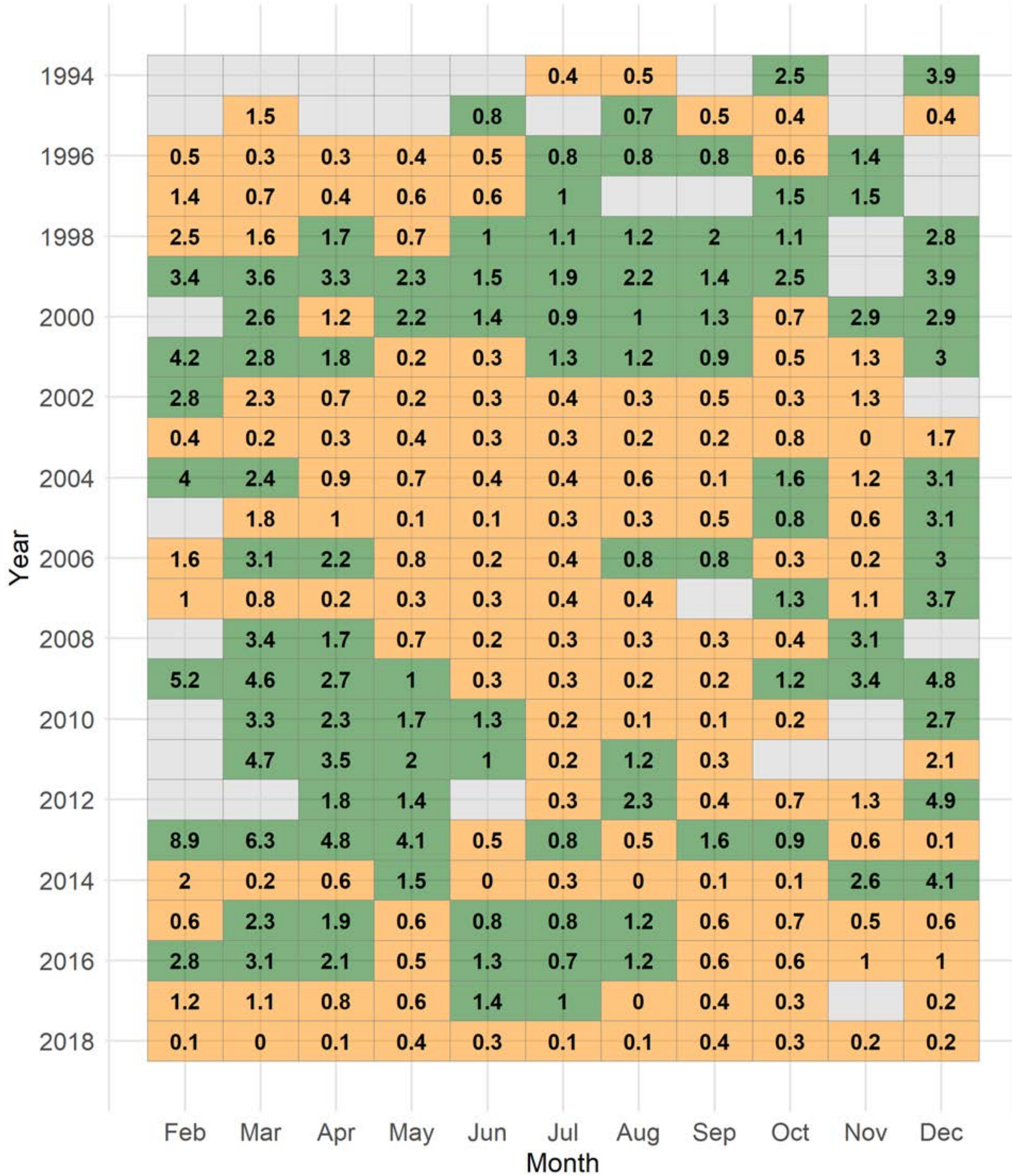

Figure 3.2-24. Average Dissolved Oxygen (mg/L) at Station 6 between 16 and 38 m

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.

Ammonium

Ammonium levels were low (<2.8 µM) in the epilimnion throughout the year while ammonium continued to accumulate at depths at or below 20 m (Table 3.2.6, Figure 3.2.25). Ammonium levels at 12 m slightly increased in June as *Artemia* activity started to pick up, but quickly depleted and decreased below the detectable level by September. At or below 16 m of depth, the ammonium level continued to increase as *Artemia* carcasses and fecal pellets sank deeper, and declined slightly in November and December with declining *Artemia* activity and further sinking of carcasses and fecal pellets. At the depth of 35 m, the ammonium level continued to rise throughout the year and reached the maximum level in December. Holomixis never occurred in 2018 as the lake remained stratified throughout the year at a depth between 10 and 15 m. A very low epilimnetic ammonium level was observed across the other 6 stations (Table 3.2.7).

Average ammonium values in the epilimnion between 1 and 10 m were below the detectable level of 2.8 µM. The minimum detectable level of 2.8 µM makes a historical comparison difficult especially for the epilimnion as an arbitrary value (2 µM) has been substituted for <2.8 µM which may not reflect actual values. Historically, average ammonium values less than 1 µM have been recorded. In 2018, the above normal epilimnetic ammonium level was found in March, April, and September; but this result may be attributable to the detectable limit of the lab instrument (Figure 3.2.26). Epilimnetic ammonium levels should have been comparable to what was recorded during the meromixis between 1995 and 2002. In this section, hypolimnion was referred as depths below 20 m in order to clearly demonstrate continuous accumulation of ammonium at the depth below 20 m. In spite of continuous accumulation of ammonium below 20 m throughout the year, hypolimnetic ammonium levels remained below the long term average and much lower than historical levels found during meromictic years (Figure 3.2.27). During the second meromixis event between 1995 and 2002, hypolimnetic ammonium level almost continuously rose from 50.4 µM in September, 1995, to 613.5 µM in August, 2001, and remained above 100 µM for almost 8 years. The hypolimnetic accumulation level in 2018 exceeded the levels during two brief meromixis events in 2005-2007 and 2011. With close to 150% snowpack in the Rush Creek drainage, Mono Lake will remain stratified and ammonium will continue to accumulate in the hypolimnion in 2019.

Table 3.2-6. Ammonium (µM) at Station 6, February-December in 2018

Depth (m)	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	-	-	-	-	-	-	-	-	-	-	-
2	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
3	-	-	-	-	-	-	-	-	-	-	-
4	-	-	-	-	-	-	-	-	-	-	-
5	-	-	-	-	-	-	-	-	-	-	-
6	-	-	-	-	-	-	-	-	-	-	-
7	-	-	-	-	-	-	-	-	-	-	-
8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
9	-	-	-	-	-	-	-	-	-	-	-
10	-	-	-	-	-	-	-	-	-	-	-
11	-	-	-	-	-	-	-	-	-	-	-
12	4.4	<2.8	<2.8	<2.8	47.1	6.1	3.3	<2.8	<2.8	<2.8	<2.8
13	-	-	-	-	-	-	-	-	-	-	-
14	-	-	-	-	-	-	-	-	-	-	-
15	-	-	-	-	-	-	-	-	-	-	-
16	41.6	45.5	47.7	49.9	47.7	55.4	48.2	68.7	66.0	55.4	55.4
17	-	-	-	-	-	-	-	-	-	-	-
18	-	-	-	-	-	-	-	-	-	-	-
19	-	-	-	-	-	-	-	-	-	-	-
20	50.4	52.7	53.2	59.3	63.8	71.0	66.5	57.7	71.5	81.5	85.4
21	-	-	-	-	-	-	-	-	-	-	-
22	-	-	-	-	-	-	-	-	-	-	-
23	-	-	-	-	-	-	-	-	-	-	-
24	51.6	53.8	59.3	63.2	63.8	73.2	71.5	72.6	82.6	90.4	87.0
25	-	-	-	-	-	-	-	-	-	-	-
26	-	-	-	-	-	-	-	-	-	-	-
27	-	-	-	-	-	-	-	-	-	-	-
28	52.7	59.3	61.0	74.3	69.9	74.3	85.9	89.8	83.7	92.0	90.4
29	-	-	-	-	-	-	-	-	-	-	-
30	-	-	-	-	-	-	-	-	-	-	-
31	-	-	-	-	-	-	-	-	-	-	-
32	-	-	-	-	-	-	-	-	-	-	-
33	-	-	-	-	-	-	-	-	-	-	-
34	-	-	-	-	-	-	-	-	-	-	-
35	63.8	64.9	72.1	-	77.1	80.9	89.3	90.9	78.7	109.2	114.2

Laboratory detection limit of 2.8µm.

Table 3.2-7. 9-meter Integrated Values for Ammonium (µm), February-December in 2018

Station	Feb	Mar	Apr	May*	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	<2.8	<2.8	<2.8	NA	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
2	<2.8	<2.8	<2.8	NA	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
5	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
6	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
7	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
11	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
Mean	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8	<2.8
SE	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

* Stations 1 and 2 in May were not sampled due to inclement weather.
Laboratory detection limit of 2.8µm.

It is notable that the large meromixis event between 1995 and 2002 appears to have a longer lasting effect on hypolimnetic ammonium levels as the hypolimnetic ammonium levels remained much higher than recent years except 2018. Lower accumulation of ammonium in recent years may be attributable to a lack or meromixis or/and weaker meromixis event preceding the monomixis. As a result, a strong negative trend is observed for all months.

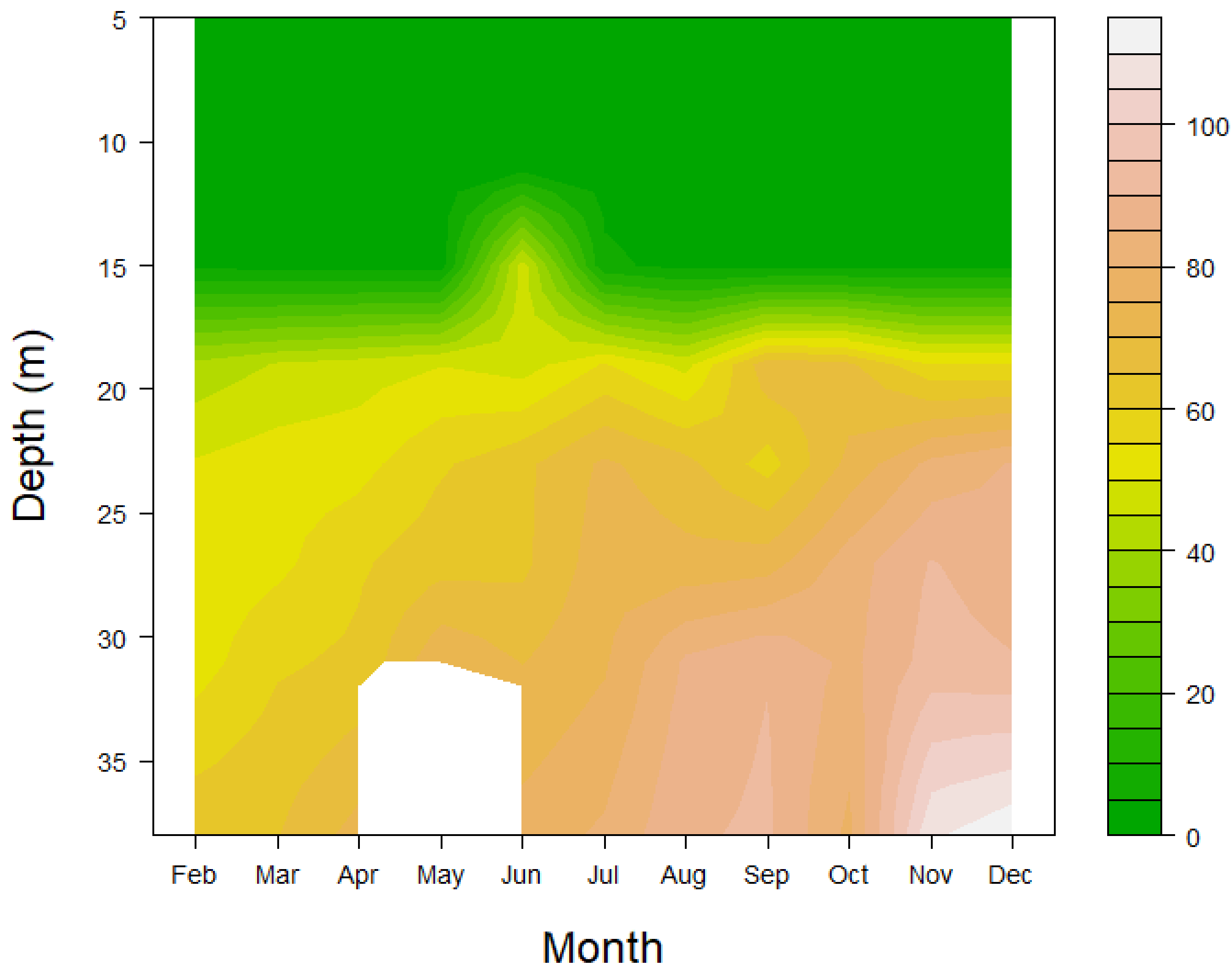

Figure 3.2-25. Ammonium Profiles (µm) at Station 6, February-December in 2018

The sample at 35 m in May was not processed.

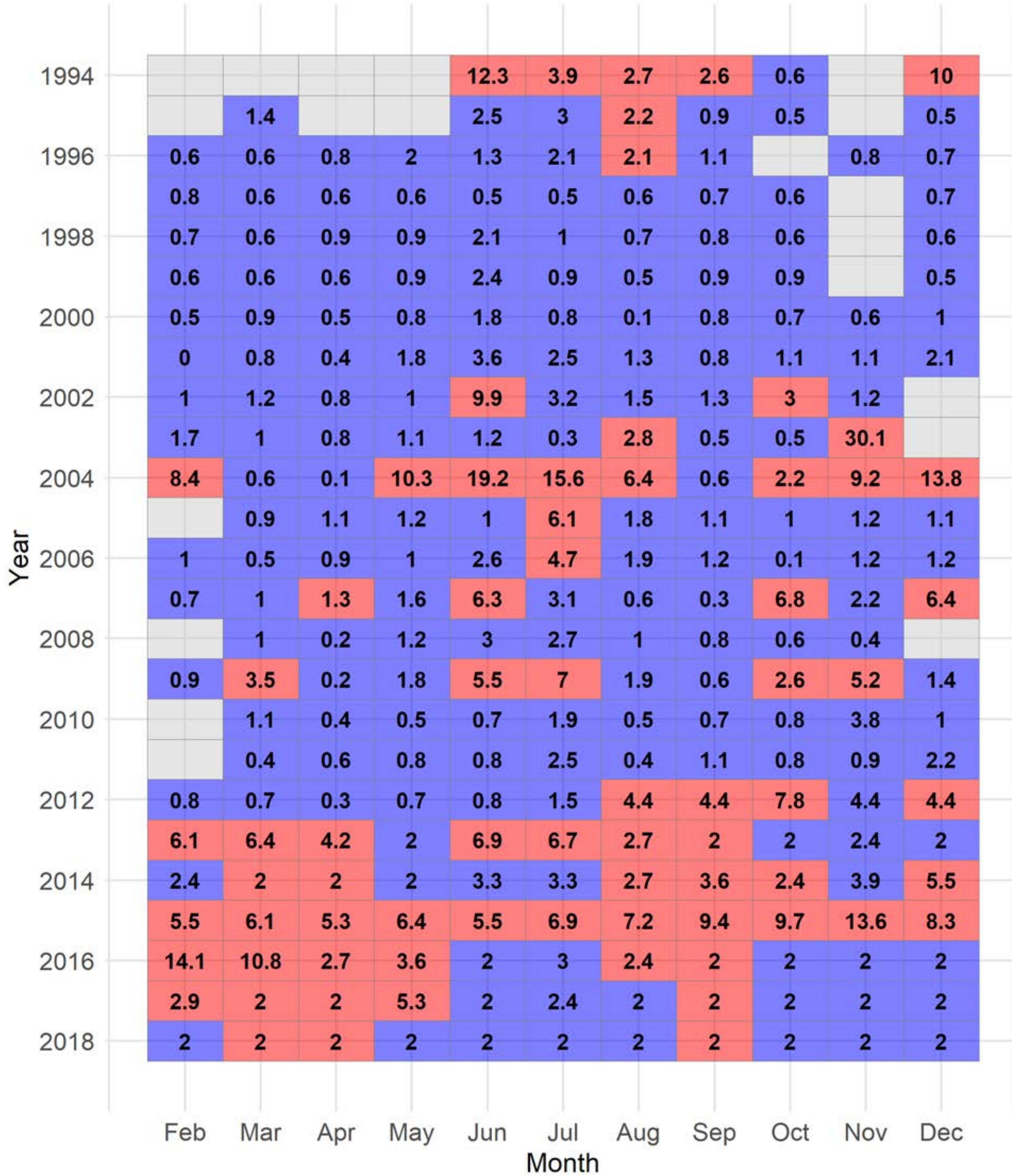

Figure 3.2-26. Average Ammonium (µm) at Station 6 at 2 and 8 m

An arbitrary value of 2 was used for values below the laboratory detection limit of 2.8µm. 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.

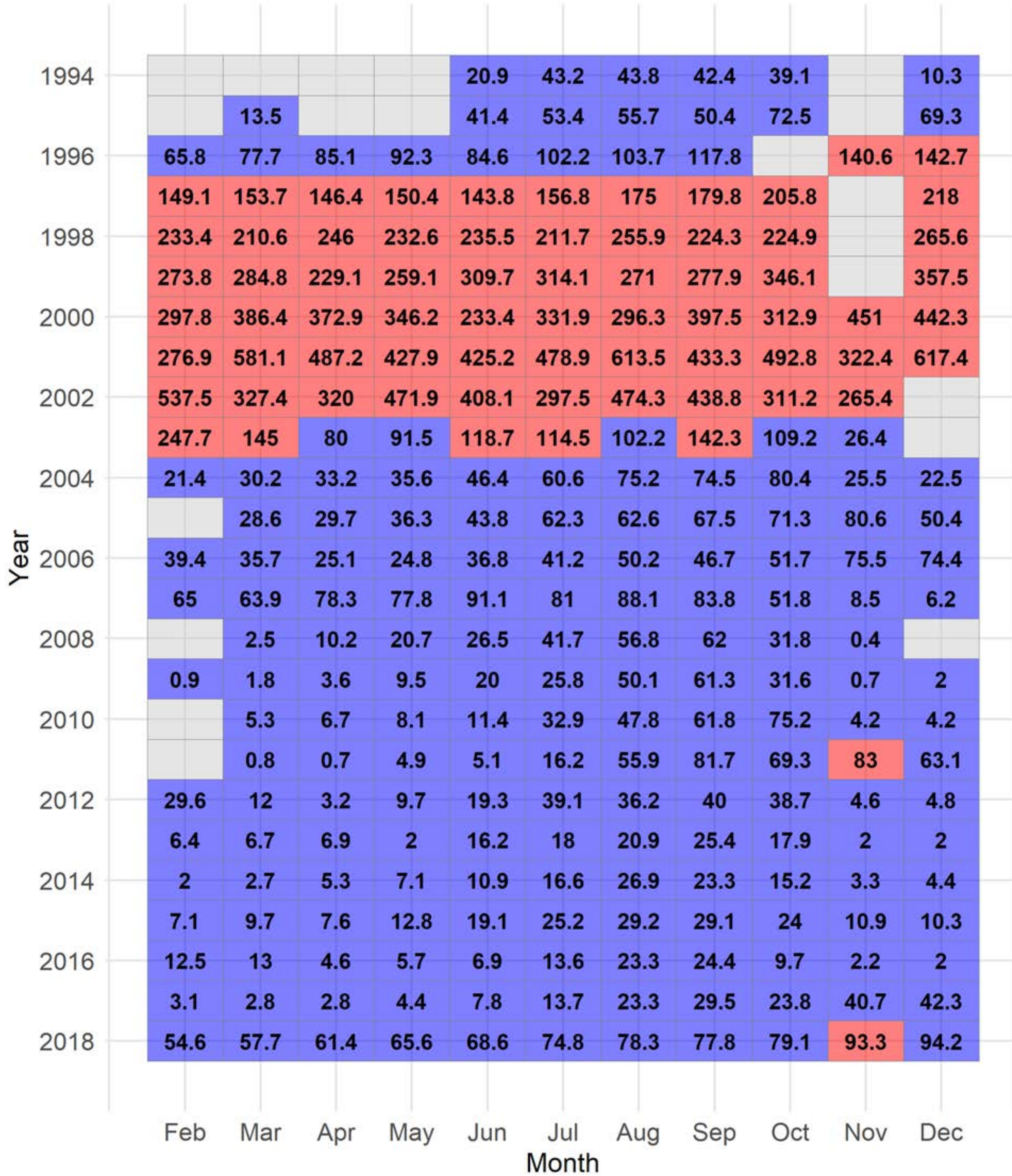

Figure 3.2-27. Average Ammonium (µm) at Station 6 at and below 20 m

An arbitrary value of 2 was used for values below the laboratory detection limit of 2.8µm. 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. An arbitrary value of 2 was used for values below the laboratory detection limit of 2.8µm

<u>Phytoplankton</u>

Seasonal changes were noted in the phytoplankton community, as measured by chlorophyll *a* concentration (Table 3.2.8, Table 3.2.9, Figure 3.2.28). Chlorophyll levels started to increase with warming temperature but quickly dropped to the lowest level (<2 µg/L) in August and recovered through the remainder of the year. This pattern was observed in epilimnion for all stations, but hypolimnetic chlorophyll levels declined between September and December. The initial rise was due to warming temperature followed by dampening of the initial peak caused by increased *Artemia* activities in summer. After the summer low, epilimnetic chlorophyll continued to rise for the remainder of the year while hypolimnetic chlorophyll peaked again in September and slowly decreased for the remainder of the year. The decrease in later months may be attributable to the recovery of epilimnetic chlorophyll which would limit light penetration. Chlorophyll levels were lower at the shallower depths at Station 6 and higher at deeper depths throughout 2018. The epilimnetic chlorophyll level (between 2 and 8 m) was highest in April (26.9 µg/L) and lowest in August (1.7 µg/L) while the hypolimnetic chlorophyll level (≤12 m) was highest in May (68.64 µg/L) and lowest in February (37.5 µg/L).

Within the epilimnion, lake-wide mean chlorophyll levels decreased throughout the spring and reached the lowest level at 1.7 µg/L at Station 6 and 1.4 µg/L lake-wide in August as *Artemia* grazing intensified (Figure 3.2.29, Figure 3.2.30). These values were comparable to 1.8 µg/L recorded at Station 6, but lower than the lake-wide value of 5.4 µg/L in September, 2017. These 2018 values were also lower than the long term average; yet, the August Secchi reading in 2018 was much lower (less transparent) than the 2017 value (3.5 m compared to 5.1 m in 2017) and lower than the historical values which routinely exceeded 8 m.

Chlorophyll levels in the epilimnion are generally lower during meromixis and higher during monomixis, particularly in spring and winter months. During the monomixis between 1988 and 1994 (years between 1988 and 1989 are the breakdown period) chlorophyll levels as high as 104 µg/L were observed. Even with elevated chlorophyll levels in spring, the levels plunged down below 1 µg/L in early summer and remained mostly below 5 µg/L until October. With successive monomixis events, however, elevated chlorophyll levels were found lasting into early summer and starting in September until the last monomixis. As discussed in previous reports between 2014 and 2016 chlorophyll levels remained above 10 µg/L (except July, 2014) throughout the year. This period coincided with the driest 5 year period on record and also very low *Artemia* population abundance. Mean *Artemia* abundance between 2014 and 2016 was the lowest abundance for a 3-year period on record. Low population and earlier *Artemia* peaks may have attributed for higher chlorophyll levels throughout summer months and peaks in late fall or winter.

The year 2018 marked the second year of the current meromixis, and epilimnetic chlorophyll levels were much lower than the preceding monomixis; however, the levels were still higher than the levels found during the second meromixis. Hypolimnetic chlorophyll levels in 2018 were lower than the preceding monomixis; however, were mostly above normal throughout the year and much higher than the levels observed during the second meromixis (Figure 3.2.31).

Table 3.2-8. Chlorophyll a (µg /L) at Station 6, February-December in 2018

Depth (m)	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	-	-	-	-	-	-	-	-	-	-	-
2	10.3	13.7	28.0	13.9	8.3	3.2	1.3	5.9	16.8	23.3	29.2
3	-	-	-	-	-	-	-	-	-	-	-
4	-	-	-	-	-	-	-	-	-	-	-
5	-	-	-	-	-	-	-	-	-	-	-
6	-	-	-	-	-	-	-	-	-	-	-
7	-	-	-	-	-	-	-	-	-	-	-
8	29.4	12.7	25.7	31.3	20.9	6.6	2.2	5.8	15.8	15.3	24.4
9	-	-	-	-	-	-	-	-	-	-	-
10	-	-	-	-	-	-	-	-	-	-	-
11	-	-	-	-	-	-	-	-	-	-	-
12	16.6	19.0	26.3	68.6	35.2	53.4	57.9	84.2	16.5	16.2	22.3
13	-	-	-	-	-	-	-	-	-	-	-
14	-	-	-	-	-	-	-	-	-	-	-
15	-	-	-	-	-	-	-	-	-	-	-
16	34.9	59.9	64.3	68.5	44.0	36.3	41.5	73.2	69.5	58.8	65.6
17	-	-	-	-	-	-	-	-	-	-	-
18	-	-	-	-	-	-	-	-	-	-	-
19	-	-	-	-	-	-	-	-	-	-	-
20	43.8	35.6	76.8	60.3	78.3	58.8	58.5	51.6	73.3	64.7	55.6
21	-	-	-	-	-	-	-	-	-	-	-
22	-	-	-	-	-	-	-	-	-	-	-
23	-	-	-	-	-	-	-	-	-	-	-
24	30.8	57.5	75.7	77.1	55.3	46.7	82.2	52.8	76.0	51.4	50.9
25	-	-	-	-	-	-	-	-	-	-	-
26	-	-	-	-	-	-	-	-	-	-	-
27	-	-	-	-	-	-	-	-	-	-	-
28	61.4	35.8	71.9	-	57.5	65.8	50.1	48.5	70.6	63.2	43.6

Table 3.2-9. 9-meter Integrated Values for Chlorophyll a (µg/L), February-December in 2018

Station	Feb	Mar	Apr	May*	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1	17.2	13.7	17.1	-	9.3	2.2	1.5	6.1	18.2	20.2	32.8
2	20.8	15.1	23.5	-	7.5	2.0	1.5	5.1	18.7	21.1	28.8
5	11.0	13.9	27.0	18.5	6.3	2.7	1.7	7.3	25.6	19.0	31.0
6	11.5	11.3	26.1	26.3	3.5	2.1	1.6	6.5	16.7	17.7	29.5
7	12.7	9.9	31.4	19.4	5.9	2.7	1.2	4.3	16.1	13.5	28.9
8	17.9	17.7	26.1	14.5	7.8	4.2	1.1	5.6	16.5	19.9	20.6
11	13.9	11.8	40.2	-	6.1	2.3	1.1	2.9	19.9	12.8	27.7
Mean	15.0	13.3	27.3	19.7	6.6	2.6	1.4	5.4	18.8	17.7	28.5
SE	1.4	1.0	2.7	2.4	0.7	0.3	0.1	0.6	1.2	1.3	1.5

* Stations 1, and 2 in May were not sampled due to inclement weather. Chlorophyll a sample from Station 11 was not processed in the lab.

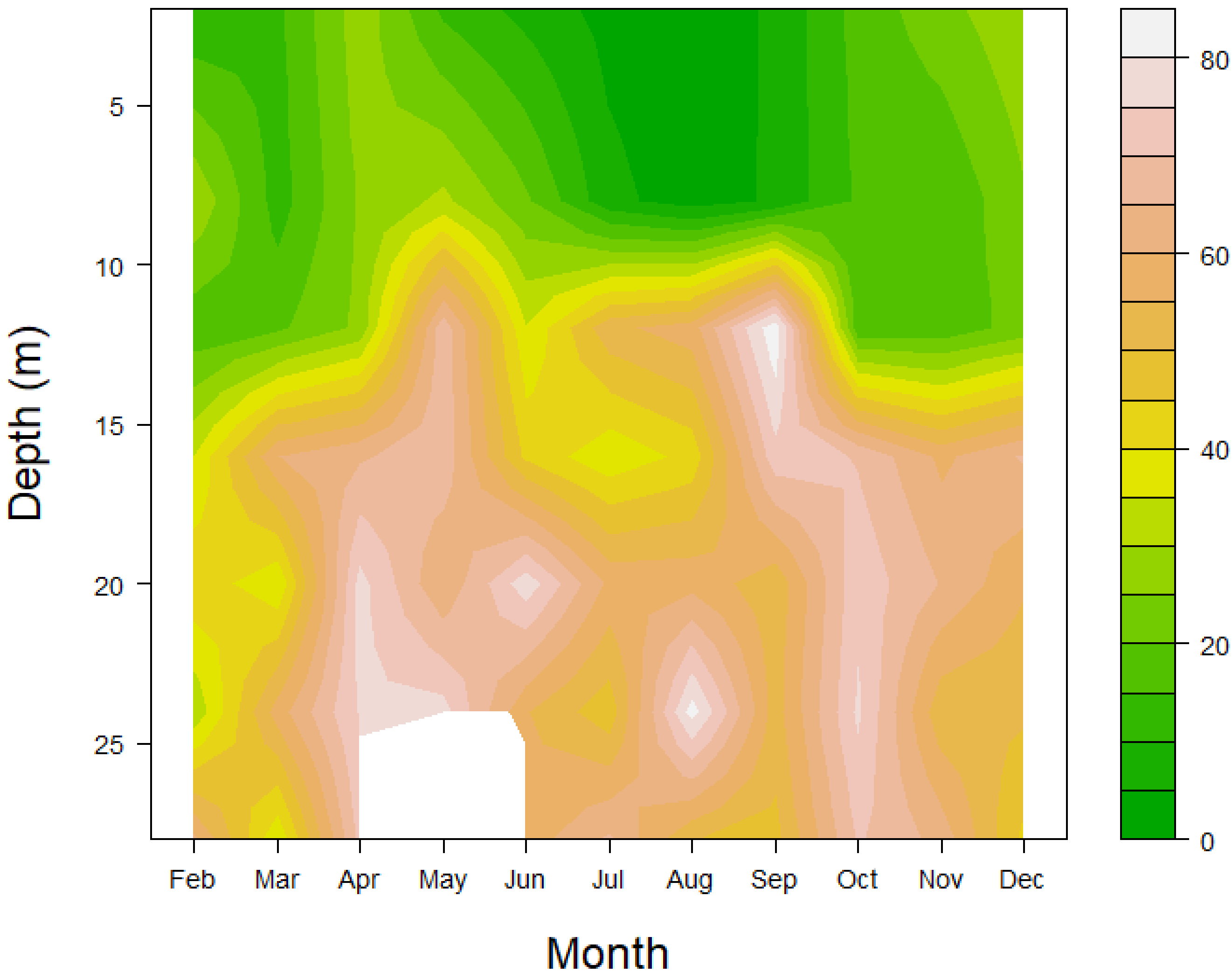

Figure 3.2-28. Chlorophyll a Profiles (µg/L) at Station 6, February-December in 2018

The sample at 28 m in May was not processed.

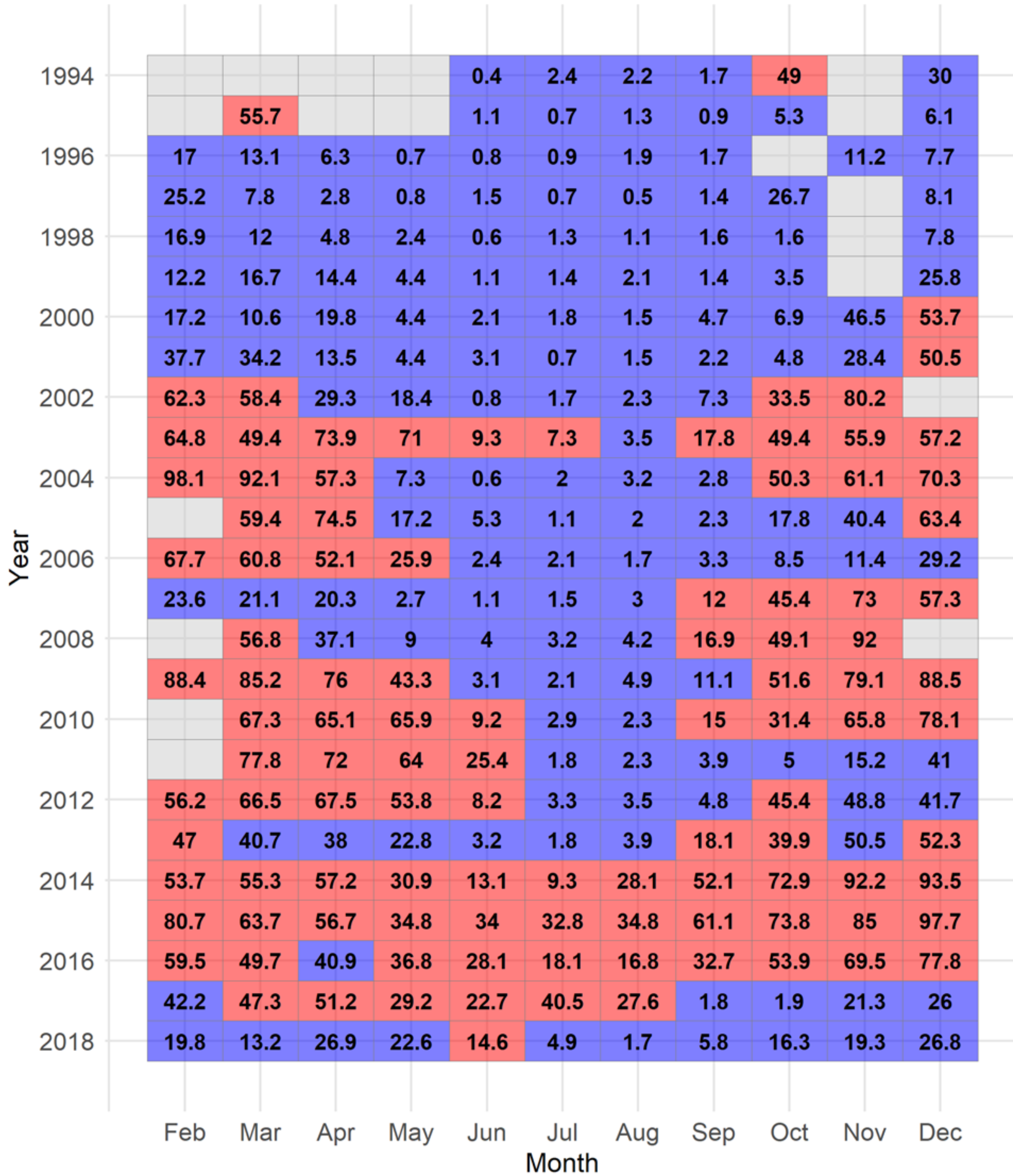

Figure 3.2-29. Average Chlorophyll a (µg/L) at Station 6 between 1 and 10 m

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.

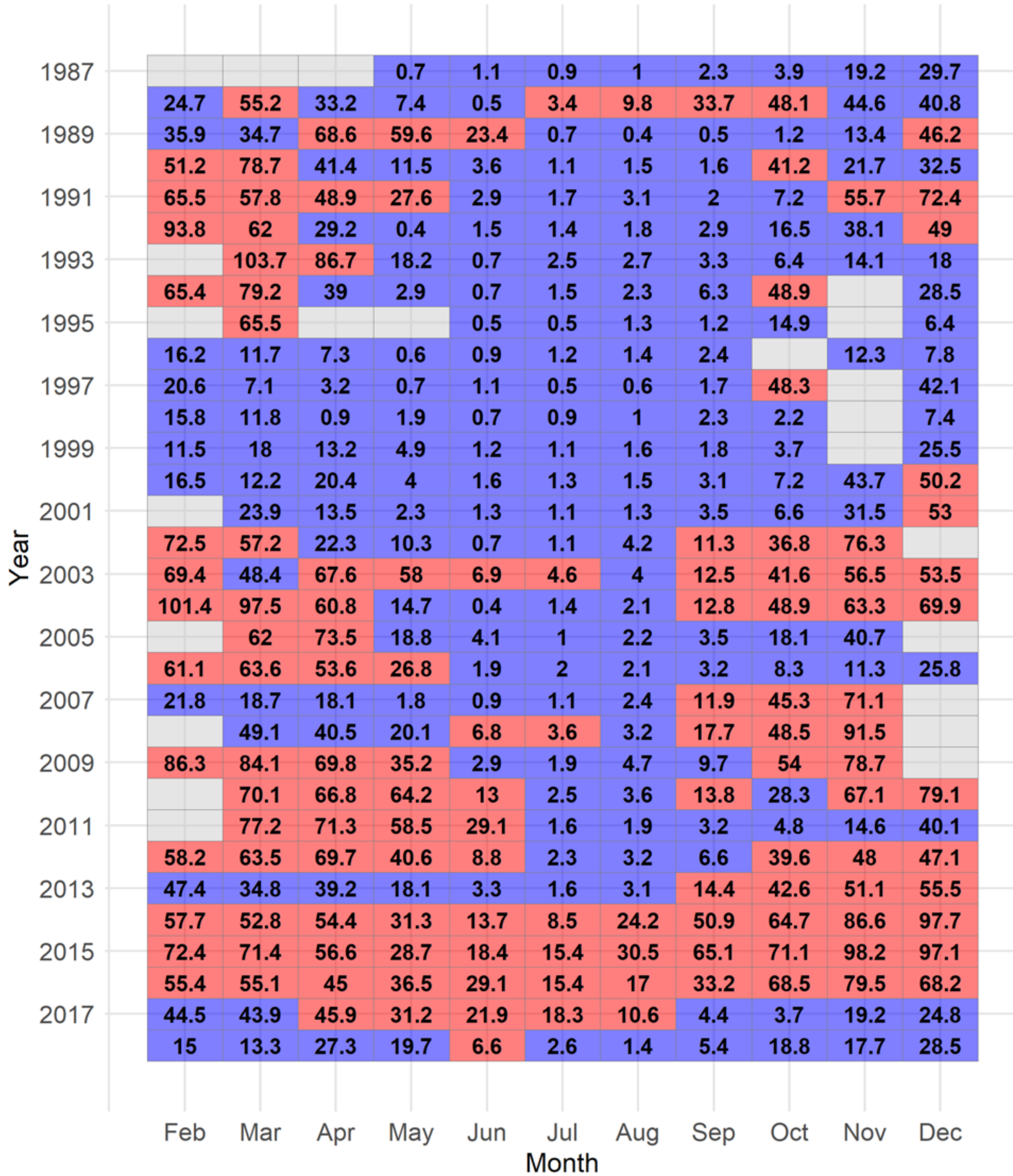

Figure 3.2-30. Average Lake-wide 9m Integrated Chlorophyll a (µg/L)

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.

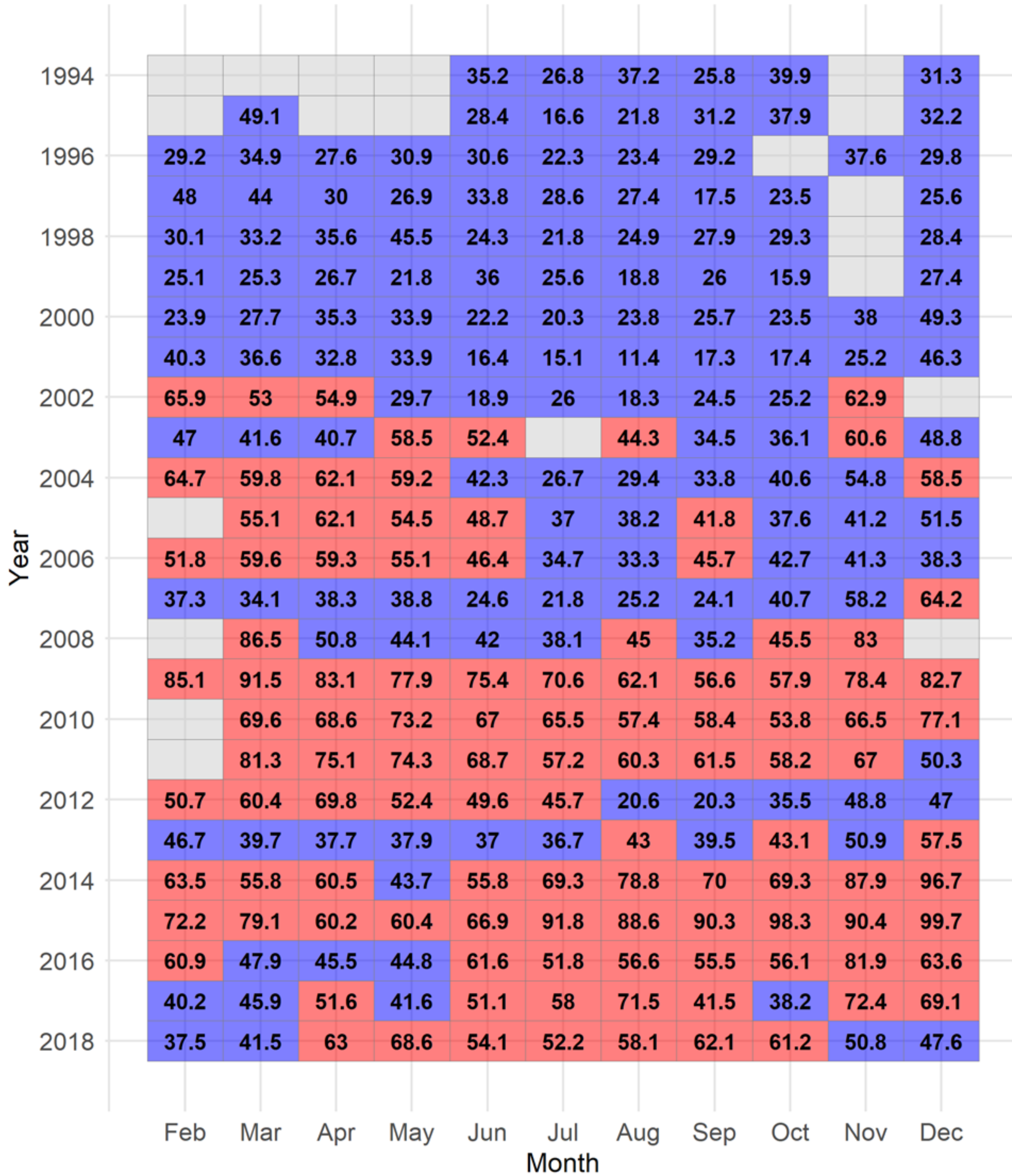

Figure 3.2-31. Average Chlorophyll a (µg/L) at Station 6 between 11 and 28 m

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.

<u>*Artemia*</u> *Population and Biomass*

Artemia population data is presented in Table 3.2.10 through Table 3.2.12 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 above 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.

<u>*Artemia* Population</u>

Hatching of overwintering cysts had already initiated by February as the mid-February sampling detected an instar lake-wide mean abundance of 23,011 +/- 4,467 m^{-2}. Almost all the instars in mid-February were instar age classes 1 and 2 (Figure 3.2.32). Instar abundance increased through spring to a peak of 66,481 +/- 16,402 m^{-2} in April. Between February and April, adults continued to be essentially absent. A peak monthly abundance of total *Artemia* for the entire lake occurred in April (60,939 +/-8,340 m^{-2}). The peak instar abundance occurred in May for 2016 and in April for 2017. Adults started to mature in May as a proportion of adult increased from 19% in May to 90% in August. The instar analysis indicated a diverse age structure of instars 1-7 and juveniles (instars 8-11) starting in April and lasting to July when abundance of each age class started to decline even though all age classes existed. In June, females with cysts were first recorded. The abundance of females with cysts peaked at 7,134 +/- 1,104 m^{-2} in July and high abundance was maintained through November. By July, hatching and instar development decreased significantly, with instars and juveniles comprising only 13% of the population compared to 81% in May. The highest adult *Artemia* abundance occurred in July (21,836 +/- 4,016 m^{-2}) and remained above 10,000 m^{-2} most likely through September.

Table 3.2-10. Artemia Lake-wide and Sector Population Means (per m^2 or m^{-2}) in 2018

	Instars		Adult	Adult	Adult Female	Ad Female Ovigery Classification				Total
	1-7	8-11	Total	Males	Total	empty	undif	cysts	naup	*Artemia*
Lakewide										
Feb	23,011	35	54	28	25	0	13	13	0	23,099
Mar	29,949	25	28	13	16	16	0	0	0	30,003
Apr	60,939	54	0	0	0	0	0	0	0	60,993
May	21,135	22,913	10,463	10,463	0	0	0	0	0	54,511
Jun	2,146	11,308	13,038	10,235	2,803	107	1,033	711	952	26,492
Jul	1,226	2,236	21,836	13,261	8,575	427	707	7,134	307	25,297
Aug	1,550	151	15,408	9,352	6,056	542	265	5,193	57	17,109
Sep	-	-	-	-	-	-	-	-	-	-
Oct	881	187	9,046	5,418	3,628	150	225	3,176	77	10,114
Nov	1,290	321	4,736	2,497	2,239	153	169	1,856	61	6,347
Dec	1,579	235	868	411	457	27	33	373	24	2,681
Western Sector										
Feb	16,299	50	13	13	0	0	0	0	0	16,362
Mar	13,215	0	6	0	6	6	0	0	0	13,221
Apr	46,546	0	0	0	0	0	0	0	0	46,546
May	14,447	13,099	6,318	6,318	0	0	0	0	0	33,863
Jun	2,254	13,360	13,924	12,126	1,797	0	1,073	617	107	29,537
Jul	1,109	3,277	30,072	19,183	10,889	479	907	8,999	504	34,458
Aug	1,500	277	18,162	11,721	6,440	429	202	5,772	38	19,939
Sep	-	-	-	-	-	-	-	-	-	-
Oct	1,078	151	4,767	3,176	1,591	66	132	1,361	32	5,996
Nov	1,289	164	2,121	1,185	936	54	60	785	38	3,573
Dec	422	173	715	362	353	35	35	262	22	1,311
Eastern Sector										
Feb	29,723	19	95	44	50	0	25	25	0	29,837
Mar	46,684	50	50	25	25	25	0	0	0	46,785
Apr	75,332	107	0	0	0	0	0	0	0	75,439
May	25,594	29,457	13,226	13,226	0	0	0	0	0	68,276
Jun	2,039	9,256	12,153	8,343	3,810	215	993	805	1,797	23,447
Jul	1,342	1,194	13,599	7,338	6,261	375	507	5,268	110	16,136
Aug	1,601	25	12,654	6,982	5,672	655	328	4,613	76	14,280
Sep	-	-	-	-	-	-	-	-	-	-
Oct	684	224	13,325	7,660	5,665	233	318	4,991	123	14,233
Nov	1,292	479	7,351	3,809	3,542	252	277	2,927	85	9,122
Dec	2,735	296	1,021	460	561	19	32	485	25	4,052

Table 3.2-11. Standard Errors (SE) of Artemia Sector Population Means (per m² or m^{-2}) from Table 3.2.10 in 2018

	Instars		Adult	Adult	Adult Female	Ad Female Ovigery Classification				Total
	1-7	8-11	Total	Males	Total	empty	undif	cysts	naup	*Artemia*
Lakewide										
Feb	4,467	19	18	16	14	0	13	8	0	4,471
Mar	5,603	17	17	13	13	13	0	0	0	5,611
Apr	8,340	54	0	0	0	0	0	0	0	8,357
May	3,483	5,501	2,160	2,160	0	0	0	0	0	10,904
Jun	197	1,416	1,321	1,392	775	50	176	207	764	2,544
Jul	182	512	4,016	2,759	1,337	93	119	1,104	146	4,452
Aug	223	49	2,062	1,383	747	127	62	644	21	2,195
Sep	-	-	-	-	-	-	-	-	-	-
Oct	132	33	2,305	1,280	1,034	40	87	893	37	2,347
Nov	146	74	1,084	562	539	48	52	458	23	1,173
Dec	605	66	203	86	120	9	10	105	8	781
Western Sector										
Feb	5,531	36	13	13	0	0	0	0	0	5,526
Mar	1,800	0	6	0	6	6	0	0	0	1,798
Apr	7,469	0	0	0	0	0	0	0	0	7,469
May	3,059	3,672	2,010	2,010	0	0	0	0	0	8,624
Jun	266	2,487	1,644	1,320	433	0	321	210	68	4,167
Jul	155	821	5,928	4,125	1,937	158	123	1,601	275	6,604
Aug	171	61	3,120	1,939	1,205	157	106	1,076	26	3,116
Sep	-	-	-	-	-	-	-	-	-	-
Oct	110	33	1,301	799	511	21	51	431	15	1,328
Nov	228	35	412	189	225	16	17	199	10	350
Dec	139	59	186	77	120	14	14	96	11	359
Eastern Sector										
Feb	6,260	13	23	28	25	0	25	16	0	6,266
Mar	4,783	32	32	25	25	25	0	0	0	4,760
Apr	12,937	107	0	0	0	0	0	0	0	12,962
May	4,765	7,999	2,926	2,926	0	0	0	0	0	15,238
Jun	310	979	2,158	2,314	1,432	80	178	376	1,510	2,720
Jul	342	211	2,948	1,562	1,403	108	177	1,187	46	3,165
Aug	435	25	2,439	1,552	972	203	64	730	34	2,881
Sep	-	-	-	-	-	-	-	-	-	-
Oct	221	57	3,791	2,136	1,667	62	166	1,417	70	3,961
Nov	205	113	1,504	816	740	77	82	652	46	1,689
Dec	1,028	118	370	160	212	12	15	186	13	1,342

Table 3.2-12. Percentage in Different Classes of Artemia Population Means from Table 3.2.10 in 2018

	Instars		Instar	Adult	Adult	Adult Female	Ad Female Ovigery Classification				Ovigerous
	1-7	8-11	%	Total	Males	Total	empty	undif	cysts	naup	Female%
Lakewide											
Feb	100	0.2	100	0.2	0.1	0.1	0	50	50	0	100
Mar	100	0.1	100	0.1	0.04	0.1	100	0	0	0	0
Apr	100	0.1	100	0	0	0	0	0	0	0	0
May	39	42	81	19	19	0	0	0	0	0	0
Jun	8	43	51	49	39	11	4	38	26	35	96
Jul	5	9	14	86	52	34	5	9	88	4	95
Aug	9	1	10	90	55	35	9	5	94	1	91
Sep	-	-	-	-	-	-	-	-	-	-	-
Oct	9	2	11	89	54	36	4	6	91	2	96
Nov	20	5	25	75	39	35	7	8	89	3	93
Dec	59	9	68	32	15	17	6	8	87	5	94
Western Sector											
Feb	100	0.3	100	0.1	0.1	0	0	0	0	0	0
Mar	100	0	100	0.05	0	0.05	100	0	0	0	0
Apr	100	0	100	0	0	0	0	0	0	0	0
May	43	39	81	19	19	0	0	0	0	0	0
Jun	8	45	53	47	41	6	0	60	34	6	100
Jul	3	10	13	87	56	32	4	9	86	5	96
Aug	8	1	9	91	59	32	7	3	96	1	93
Sep	-	-	-	-	-	-	-	-	-	-	-
Oct	18	3	20	80	53	27	4	9	89	2	96
Nov	36	5	41	59	33	26	6	7	89	4	94
Dec	32	13	45	55	28	27	10	11	82	7	90
Eastern Sector											
Feb	100	0.1	100	0.3	0.1	0.2	0	50	50	0	100
Mar	100	0.1	100	0.1	0.1	0.1	100	0	0	0	0
Apr	100	0.1	100	0	0	0	0	0	0	0	0
May	37	43	81	19	19	0	0	0	0	0	0
Jun	9	39	48	52	36	16	6	28	22	50	94
Jul	8	7	16	84	45	39	6	9	90	2	94
Aug	11	0.2	11	89	49	40	12	7	92	2	88
Sep	-	-	-	-	-	-	-	-	-	-	-
Oct	5	2	6	94	54	40	4	6	92	2	96
Nov	14	5	19	81	42	39	7	8	89	3	93
Dec	67	7	75	25	11	14	3	6	90	5	97

<u>Instar Analysis</u>

The instar analysis, conducted at seven stations, shows patterns similar to those shown by the lake-wide and sector analysis, but provides more insight into *Artemia* reproductive cycles occurring at the lake (Figure 3.2.32). Instars 1 were most abundant in March but declining sharply after March while instars 2 peaked more broadly from February to April as overwintering cysts were hatching. By May all age classes of instars 1-7 and juveniles were present and comprised approximately 81% of the *Artemia* population while adults comprised the remainder (19%). A proportion of instars and juvenile combined fell to 50% in June and down to 10% by August. The presence of late stage instars and juveniles throughout the monitoring year indicate survival and recruitment into the population. Adult abundance decreased from 89% in October to 32% in December while instar and juvenile age classes increased from 11% to near 68% over the same period. Instar abundance peaked in April and immediately started to decline recording the lowest abundance in October. Since October, instar abundance of both age classes rebounded indicating late hatching of the second generation even though there was no distinct peak found indicating higher generations.

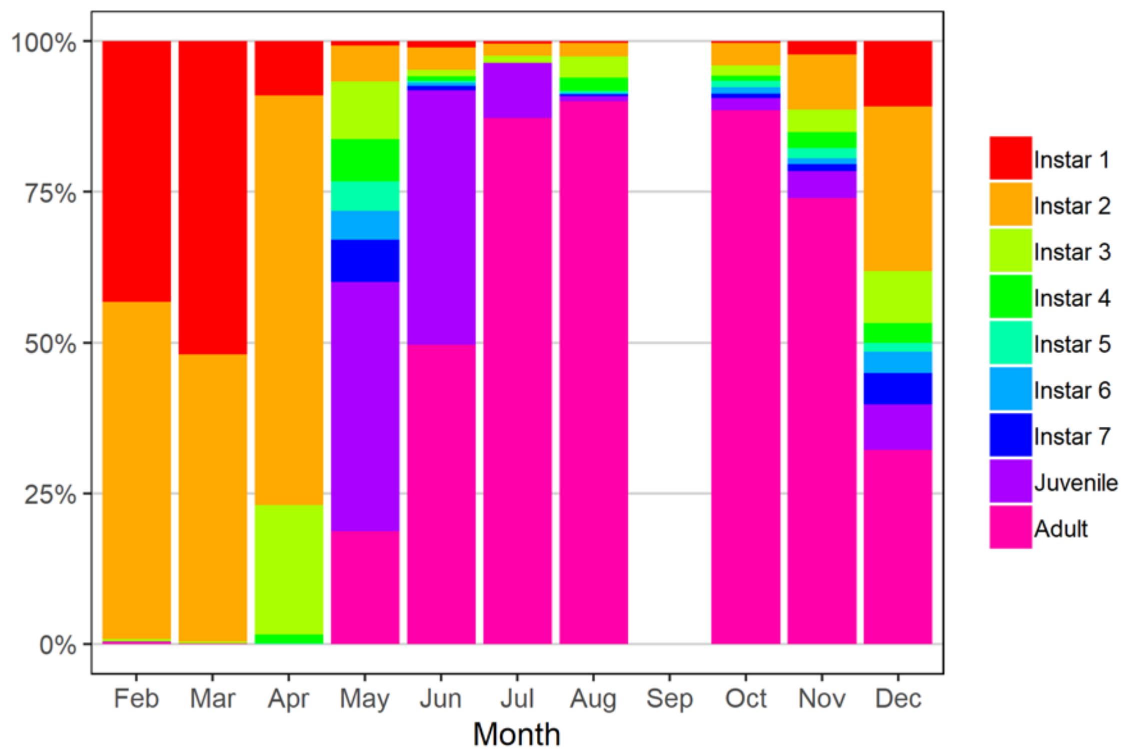

Figure 3.2-32. Compositional Changes of Artemia Instars and Adults in 2018

Biomass

Mean lake-wide *Artemia* biomass remained above 10 g/m^2 between June and August peaking at 15.3 g/m^2 in July (Table 3.2-13). Mean biomass was only slightly higher than 10 g/m^2 in August and below 10 g/m^2 in October (6.91 g/m^2); such that mean biomass in September would have been below 10 g/m^2. As observed in 2017, peak mean biomass was higher in the western sector than in the eastern sector. This pattern was observed also in 2015 indicating the reversed pattern found in 2016 was rather atypical.

Table 3.2-13. Artemia Mean Biomass (g/m^2) in 2018

Month	Lakewide	Western Sector	Eastern Sector
Feb	0.55	0.42	0.69
Mar	0.45	0.32	0.58
Apr	2.21	2.31	2.10
May	7.99	4.65	10.2
Jun	12.2	13.1	11.3
Jul	15.3	20.4	9.04
Aug	11.3	12.8	9.82
Sep	-	-	-
Oct	6.91	3.67	10.2
Nov	4.30	1.88	6.73
Dec	0.91	0.76	1.07

Reproductive Parameters and Fecundity Analysis

By June, fecund females were plentiful enough to conduct fecundity analysis (Table 3.2-10, Table 3.2-14, Figure 3.2-33). Females were virtually absent between February and May and started to appear in June when 96% of females were ovigerous, with 26% oviparous (cyst-bearing), 35% ovoviviparous (naupliar eggs) and 38% undifferentiated eggs. From July through December, over 90% of females were ovigerous with the majority (87-94%) oviparous.

The lake-wide mean fecundity declined from June to July but continued to increase from July to October, resulting in July being the lowest among 4 months. The lake-wide mean fecundity was initially 29.8 +/- 1.1 egg per brood in June, decreased to 21.8 +/- 0.8 eggs per brood in July, and rebounded to 32.5 +/- 1.6 in October. Although fecund females were documented during population analysis in November, densities were too low to conduct the analysis. The majority of fecund females (>88%) were oviparous between July and October. Little difference was observed in fecundity between the western and eastern sectors. Typically, mean female lengths are positively correlated with mean eggs per brood, and 2018 followed this pattern (r=0.36, *P*<0.0001). The largest monthly mean of females was found in October (9.8 mm) when the mean brood size was largest (32.5 +/- 1.6 eggs per brood). These numbers were lower than what observed in 2017 (10.0 mm and 36.4 eggs per brood).

Table 3.2-14. Artemia Fecundity Summary in 2018

	# of Eggs/Brood				Female Length (mm)		
Month	Mean	SE	% Cyst	% Indented	Mean	SE	n
Lakewide							
Jun	29.8	1.1	98.6	42.3	9.4	0.1	7
Jul	21.8	0.8	98.6	59.4	8.9	0.1	7
Aug	23.2	1.1	98.6	54.3	9.2	0.1	7
Sep	-	-	-	-	-	-	-
Oct	32.5	1.6	98.6	59.2	9.8	0.1	6
Western Sector							
Jun	29.5	1.6	97.6	41.5	9.2	0.1	4
Jul	22.6	1.0	97.4	56.4	9.0	0.1	4
Aug	23.5	1.5	100	60.0	9.3	0.1	4
Sep	-	-	-	-	-	-	-
Oct	31.6	2.3	97.6	48.8	9.7	0.1	4
Eastern Sector							
Jun	30.2	1.5	100	43.3	9.6	0.1	3
Jul	20.8	1.2	100	63.3	8.8	0.1	3
Aug	22.8	1.6	96.7	46.7	9.2	0.1	3
Sep	-	-	-	-	-	-	-
Oct	33.7	2.1	100	73.3	10.0	0.1	2

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

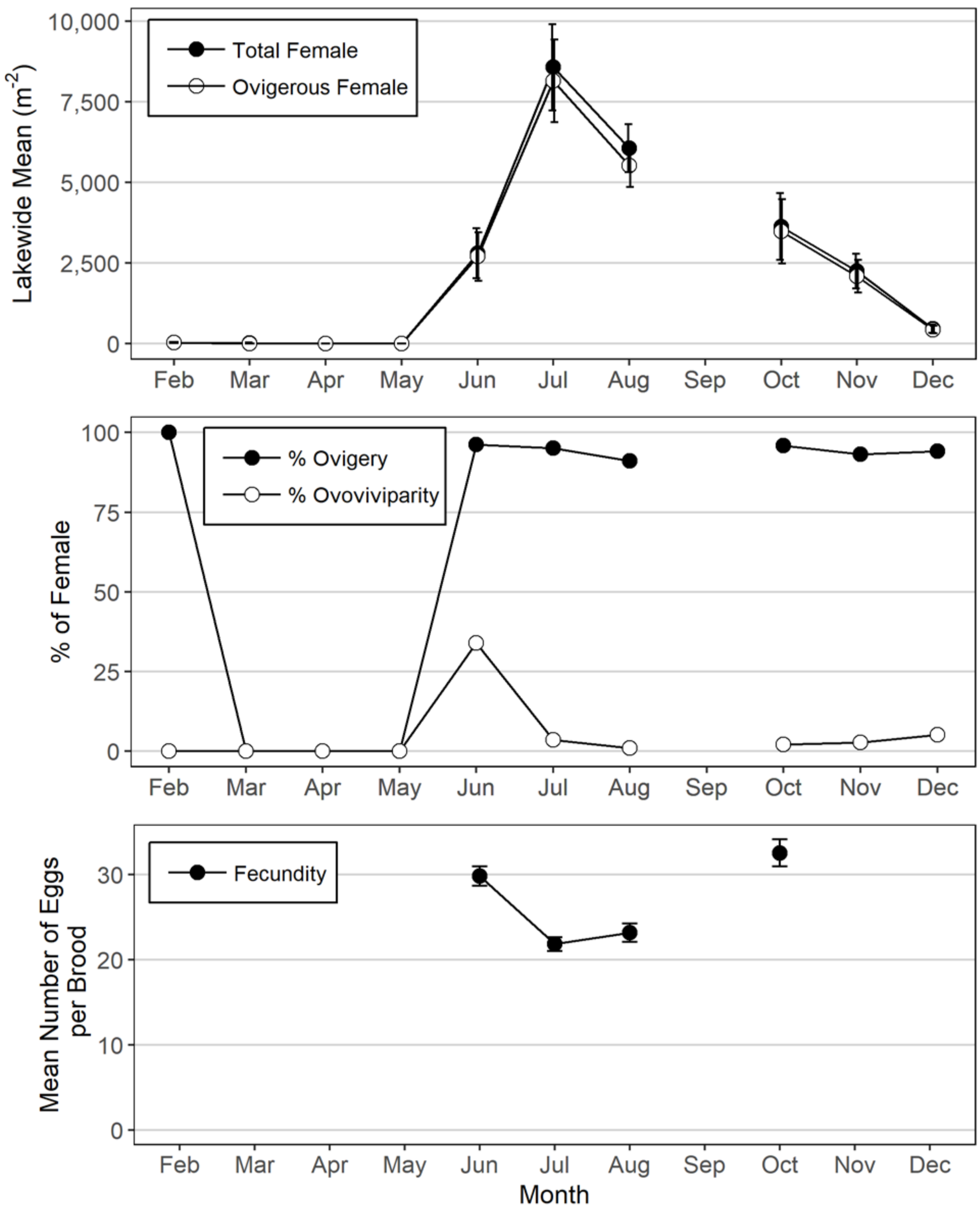

Figure 3.2-33. Artemia Reproductive Parameters and Fecundity, June-October in 2018

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

The upward trend of mean adult *Artemia* after the lowest mean on record was observed in 2015 did not continue into 2018 as the mean value decreased from 15,158 m^{-2} in 2017 to 12,120 m^{-2} in 2018 (Table 3.2.15). The mean and median adult abundance in 2018 was 12,120 $m^{-2}$and 12,024 m^{-2}, respectively, and remained below the long-term average. Due to low abundance during the previous 5 years, the 5-year running average between 2014 and 2018 was lowest on record for mean and median. The centroid is the calculated center of abundance of adults. The third year in row the centroid day did not follow the declining trend; instead it remained above 215 days (216 days in 2018, August 3), 6 days later than the long term average of 210 days which corresponds to July 28 or 29 depending on whether a year is a leap year or not (Figure 3.2.34). Because of recent larger centroid values atypical centroid, the declining trend has weakened as demonstrated in Figure 3.2.34. The trend line was steeper when the previous 3 years were excluded (blue colored line) and the model fit was much better as well (R^2=0.61 compared to R^2=0.39). The mean, median, peak and centroid data for 2015 was misreported in the 2015 annual report and has been corrected and reported in this document. The corrected mean, median and peak are higher by 12%, 4% and 21% respectively.

A year following the onset of monomixis has coincided with high adult *Artemia* abundance at Mono Lake as nutrients which are previously contained in the hypolimnion becomes fully available for phytoplankton throughout the water column (Figure 3.2.35). Adult *Artemia* abundance in 1989 and 2004 was the second and third highest adult density recorded since 1979. It appears the longer the period of meromixis, the higher the peak of *Artemia* population when meromixis breaks. The last two meromixis events which only lasted 1 to 2 years resulted in shorter peaks. Mono Lake became meromictic in 2017 for the fifth time on record and remained meromictic throughout 2018. The current snowpack numbers indicate that the meromictic regime will re-strengthen and Mono Lake will remain meromictic into 2021. At least 4 years of meromixis should result in higher adult *Artemia* abundance than the abundance observed during the last two episodes of meromixis.

Table 3.2-15. 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
Mean	18,951	17,676	43,714	210
Min	7,676	5,786	18,498	179
Max	36,643	36,909	105,245	252

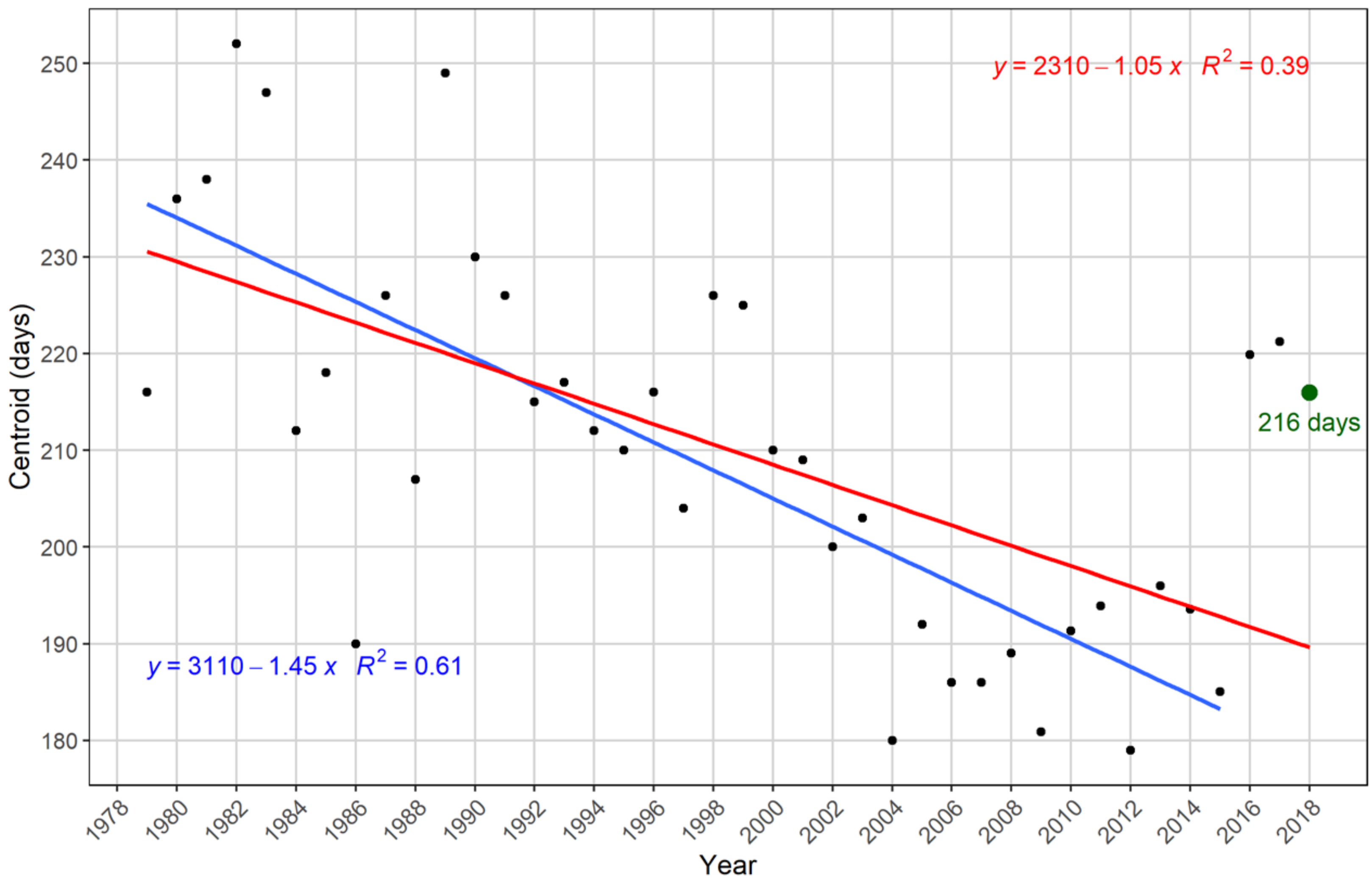

Figure 3.2-34. Adult Artemia Population Centroid

A green dot indicates a value in 2018. A blue line indicates a linear trend line between 1979 and 2015 while a red line indicates a linear trend for all monitoring years.

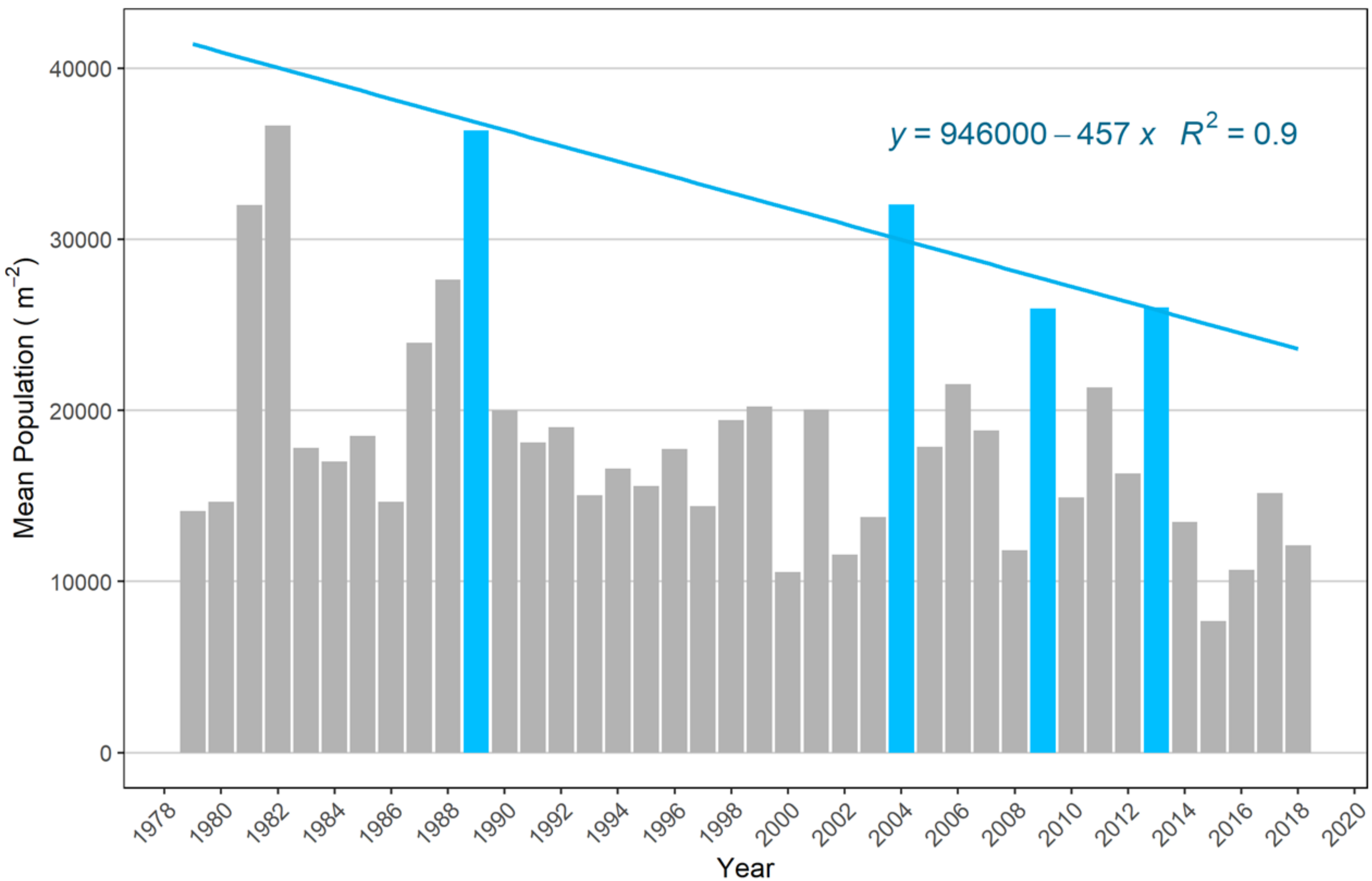

Figure 3.2-35. Mean Lake-wide Adult Artemia Population (per m2)

Years with a blue colored bar indicate years with peak *Artemia* abundance occurring subsequent to the onset of monomixis. A blue line indicates a temporal trend of peak *Artemia* abundance of 4 years (1989, 2004, 2009, and 2013).

The examination of monthly average *Artemia* abundance reveals a temporal shift in peak monthly abundance to earlier months for both adult and instars (Figure 3.2.36, Figure 3.2.37). In 2018 peak monthly average adult *Artemia* occurred in July while peak monthly average instar was observed in April. Monthly *Artemia* population abundance has been monotonically declining for late summer to fall (later period); but shows more complicated pattern for late spring to summer (earlier period). With the end of meromixis in 2003, monthly abundance in earlier period started to increase or monthly peaks started to occur between May and July instead of between July and August. As *Artemia* population in general plummeted during the driest 5-year period between 2012 and 2016, *Artemia* population abundance in both periods started to decline even though the peak monthly abundance remained to occur between May and July. In past 2 years with the onset of meromixis, peak monthly abundance was observed in July.

In 2018 monthly average instar peaked in April and both March and April's monthly averages were found above average (Figure 3.2.37). The peak in April was comparable to the peaks from the previous 3 years even though the 2018 peak was slightly lower than those of the previous 3 years. A temporal shift in instar abundance is demonstrated in Figure 3.2.39. Two periods (February to May and June to December) were compared using log transformed instar *Artemia* population abundance. Averages of 2 periods started to diverge with the onset of the meromixis in 1995 with some overlaps, and separation became more accentuated after 2008. Two slopes were significantly different ($P<0.0001$). More instars are present in earlier months of the year (the first generation hatching of cysts) but fewer instars are present in later months of the year (the second or third generation hatching of nauplii).

It appears the prolonged monomixis since 2003, which was interrupted by 2 short meromixis events, may be responsible for the shift in monthly adult and instar population peaks. More detailed discussions are given in Analysis of Long Term Trends.

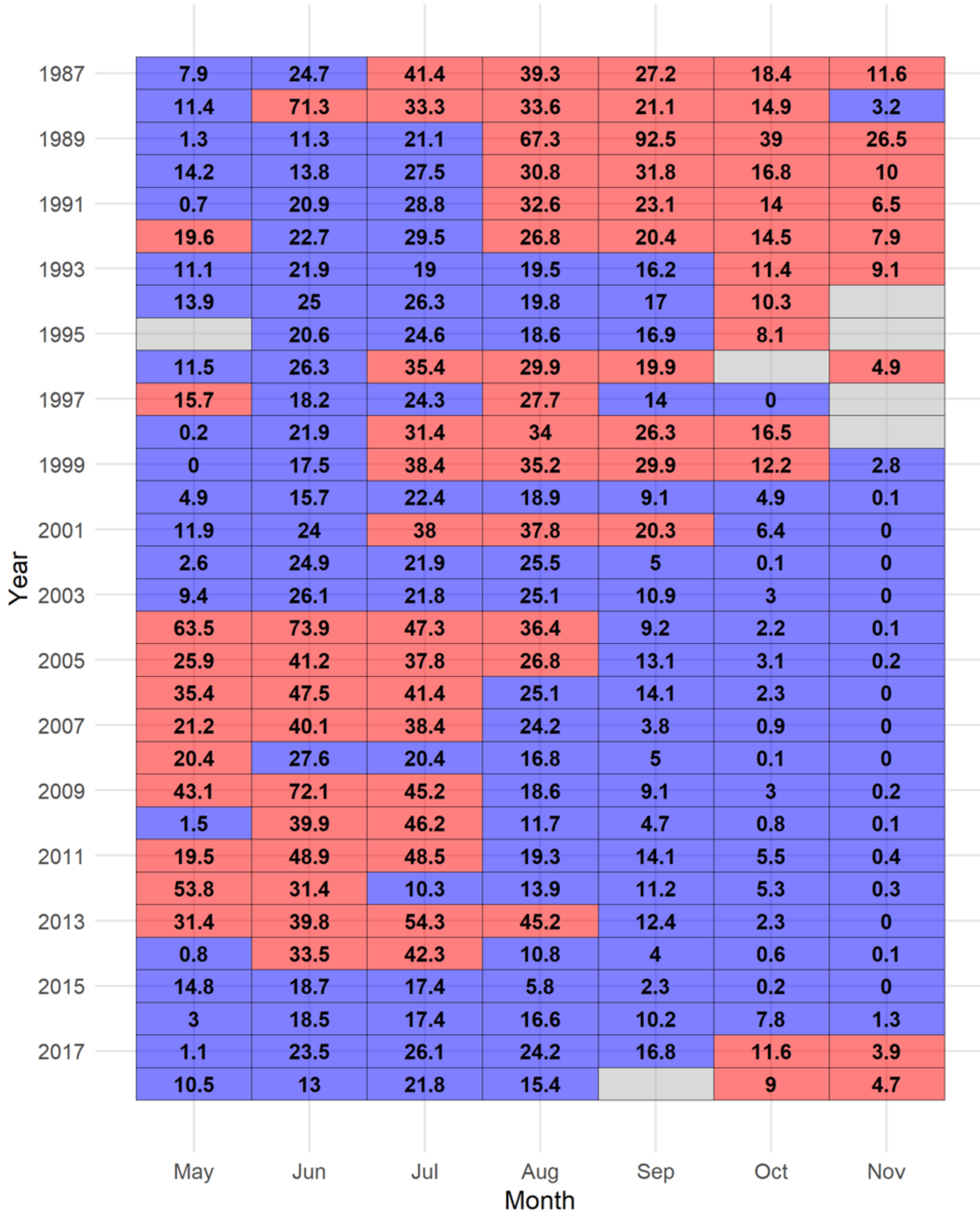

Figure 3.2-36. Monthly Average Adult Artemia Abundance of 12 Stations

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.

Year	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1987	3.3	22.3	18.5	33.6	13.3	5.9	1.7	0.5	2	9.2	
	0.4	50.2	46.8	26.9	24.7	5.7	4.8	1.3	2.6	1.2	0
1989	0	17.8	9.3	2.9	250.9	7.1	1	0.2	0.9	1.5	5.1
	2	7.5	12	89.7	234.8	12.4	7.1	1.1	1.9	3.6	8.8
1991	5.7	33.6	26.6	32	39.5	18	3.6	1	1.4	3.8	3.6
	6.6	14.8	26.5	28.9	14.3	5.4	6.1	2	3.4	9.6	15.3
1993		13.3	23.1	11.5	74.5	27.5	11.2	4.6	5.1	8.9	4.4
	5.7	18.4	25.9	12.7	48.9	15.9	6.7	4.9	3.4		1.4
1995		16.1			23.5	21	8.7	1.8	2.2		0.5
	11.1	30.3	81	26.3	10.6	3.1	2.4	0.9		1.7	1.1
1997	7.5	12.8	36.4	35.4	12.6	6.4	3.7	1.2	0.9		0.7
	10.5	23.7	48.4	64.4	18.1	4.2	2.5	1.3	0.8		1.2
1999	26.2	34.7	42	60.6	11.2	5.4	3	1.6	1.1	1.3	0.6
	12.9	14.1	23.2	24.7	93.1	9.5	2.9	2.6	2.1	0.3	0.5
2001	3.4	3.2	30.1	36	23.1	7.8	3.3	2.5	2.8	0.3	0.4
	0.9	20.7	36.9	18.3	66.2	10	2.4	1.6	0.2	0.1	
2003	3.2	4.4	15.3	6.6	90.3	42.4	8.8	2.3	1.2	0	0
	47.3	68.7	49.1	20.7	15.2	5.7	3.4	2.4	0.9	0.2	0.3
2005		31.8	33.6	9.9	15.5	11.5	6.9	2.9	2.6	1.3	0.3
	13.7	46.8	92.9	10.1	12.2	10.1	3.6	2.2	0.9	0.3	0.9
2007	2.7	14.4	51.9	45.7	24.9	10.4	2.8	1.1	0.6	0.1	0.1
	1.1	10.7	26.7	13.4	83.5	13.6	6.8	2.3	0.2	0	0.1
2009	19.3	43.3	54.1	27.3	11.1	6.9	2.4	2.6	1.5	0.6	0.5
		31.4	64.6	67	9.2	4	2.8	2.2	0.7	0.2	0.3
2011		39.9	110.2	97.5	15.7	4.7	2.1	3	2.2	0.9	0.7
	12.9	31.2	40.2	29.6	18.4	1.2	1.2	1.3	1.6	0.4	0.8
2013	1.5	28.1	81.4	30.2	11.9	3.6	1.3	1.1	1	0.2	0.8
	35.4	150.9	119.7	60.4	3.8	0.6	0.7	0.5	0.5	0	0.1
2015	10.1	18.5	66.8	24.9	19.1	2.2	0.8	0.6	0.2	0.1	0.4
	22.5	59.6	64.3	40.7	14.6	3.6	0.7	0.4	0.8	1.7	1.6
2017	9.9	41.6	66.5	29	7.6	5.8	1.1	0.7	1.1	2.3	2.2
	23	29.9	60.9	21.1	2.1	1.2	1.6		0.9	1.3	1.6

Month

Figure 3.2-37. Monthly Average Instar 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.

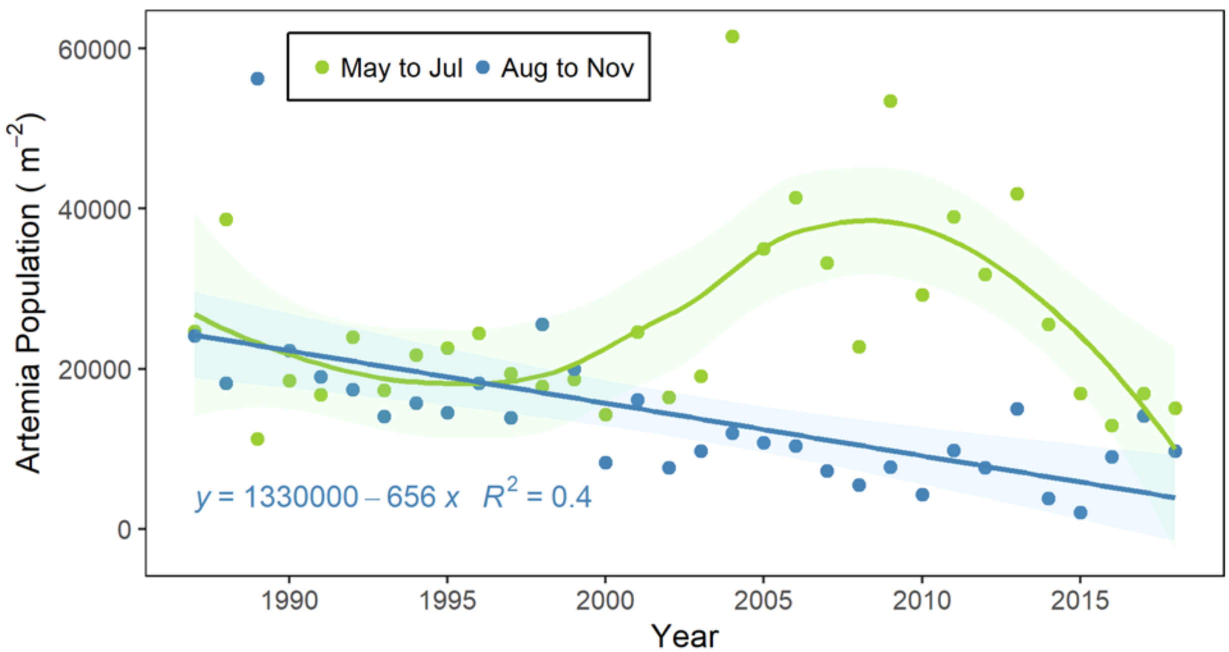

Figure 3.2-38. Comparison of Mean Lake-wide Adult Artemia Population (per m2) between Earlier and Later Months

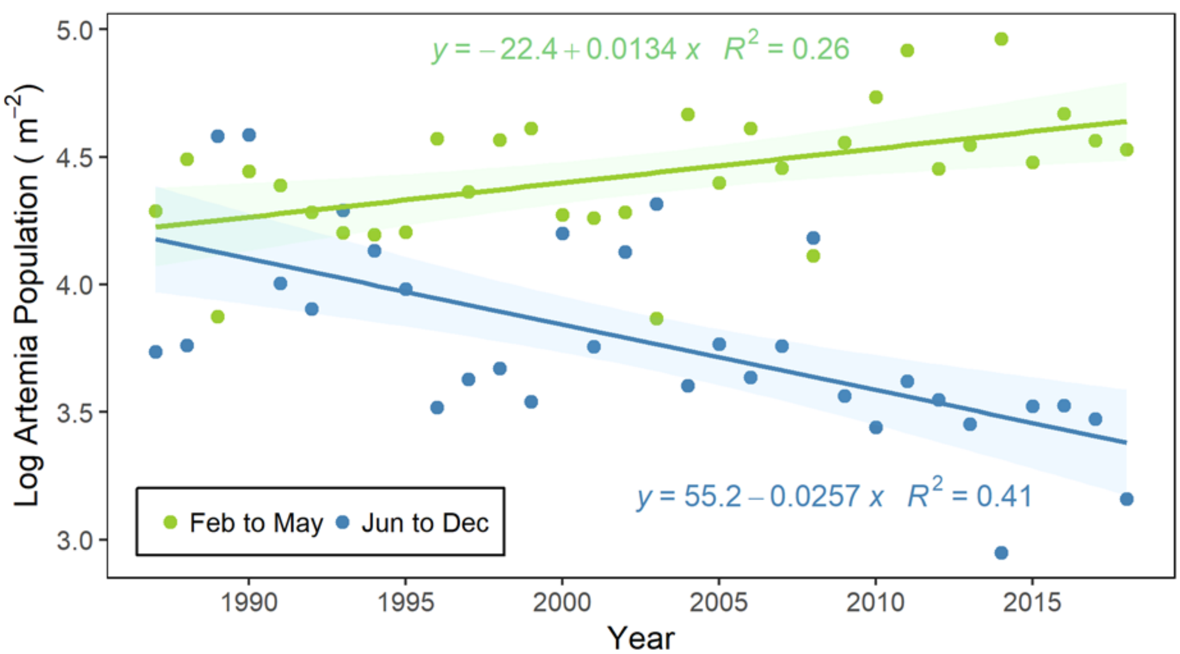

Figure 3.2-39. Comparison of Mean Lake-wide Instar Artemia Population (per m2) between Earlier and Later Months

Analysis of Long Term Trends

Salinity and Mono Lake Elevation

Salinity of Mono Lake is tightly associated with lake elevation across all monitoring stations but one, and relationships are much stronger for salinity measured at shallower depths (Table 3.2-16).

Table 3.2-16. Relationships between Salinity and Lake Elevation for 3 Different Depth Classes

	Depth		
Station	1 to 10m	11 to 20m	21 to 38m
2	-0.93	-0.88	-0.63
3	-0.90	-0.88	-0.66
4	-0.91	-0.88	-0.64
5	-0.89	-0.89	-0.68
6	-0.93	-0.88	-0.66
7	-0.93	-0.88	-0.70
8	-0.89	-0.89	-0.68
10	-0.88	-0.85	-0.79
12	-0.87	-0.87	-0.65

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.

A relationship between salinity and lake elevation at Station 6 is presented in Figure 3.2-40 to Figure 3.2-42 for each depth category. These figures show a very strong linear trend but also deviations from the trend line which are colored orange in each figure. Further analysis revealed that these deviated points were mainly occurring after 2008 and corresponded to seasonal low and high spikes (Figure 3.2-43). Snowmelt driven runoff decreases salinity in the hypolimnion in summer and with a combination of evaporation and lake mixing salinity in the hypolimnion increases through fall and winter during monomixis but during meromixis salinity remains relative low throughout the year. An annual range of seasonal salinity fluctuations widened considerably in recent years (Figure 3.2-44). This trend was much more pronounced for the depth between 1 and 10 m and found consistently across the lake. In 2018, a range of salinity exceeded 15 g/L at all stations. At Station 6 salinity started at 96.6 g/L in February reaching the lowest level in September at 80.9 g/L, resulting in an inter-annual range of 15.7 g/L, compared to 4.8 g/L observed in 1995. The second meromixis started in 1995 and the lowest epilimnetic salinity was observed in March at 91.3 g/L in March while the highest epilimnetic salinity was observed in December at 86.5 g/L. Smaller salinity range was expected in 1995 compared to 2018 because of smaller input into Mono Lake (210% of Normal in 1995 compared to 308% of Normal). Between 1995 and 1996, Mono Lake input from Rush and Lee Vining Creeks averaged 200% of Normal; yet, hypolimnetic salinity range was 4.8, 2.1, 1.0, and 3 g/L during the same period, averaging 2.7 g/L. These ranges were much smaller than what had been observed since 2008.

Starting around 2008 as well, there appears to be a shift in the salinity-lake elevation relationship. Figure 3.2-45 clearly demonstrate this change. For instance, at lake levels near 6,377.5 feet, salinity levels were around 85 g/L during the earlier period, while at the same lake level salinity levels ranged between 88 g/L and 97 g/L during the later period. The opposite trend was also found for lower salinity. At lake levels near 6,382 feet, salinity as low as 74.3 g/L was observed in 2008 while at the same lake elevation salinity clustered around 80 g/L. Salinity appears to be higher for lower lake elevation or elevations below approximately 6,382 feet but lower for higher lake elevation or elevation above approximately 6,382 feet, resulting in a much steeper slope between salinity and lake elevation since 2008.

It is not clear what is driving higher seasonal salinity range and changing relationship between salinity and lake elevation. One possibility is a lack of data for higher lake elevation levels (above 6,385 feet); such that observed changes may be an artifact of the existing data range. As a result, it is difficult to predict future salinity levels.

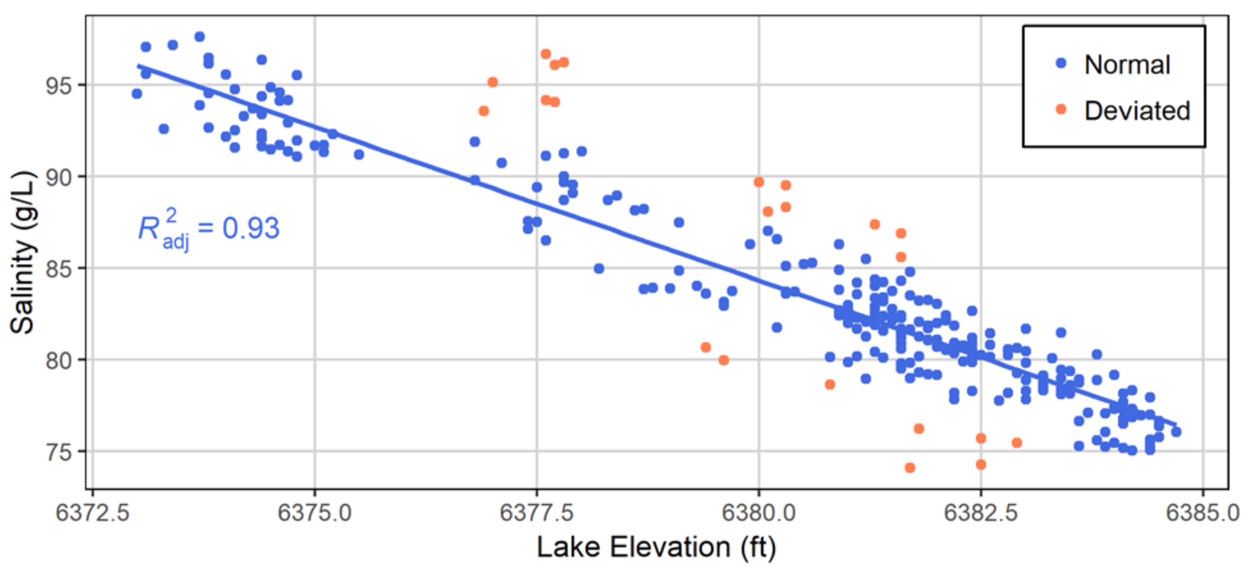

Figure 3.2-40. Relationship between Mono Lake Elevation and Salinity Averaged between 1 and 10 m of Depths at Station 6

Blue line indicates a best fit line based on blue colored points only

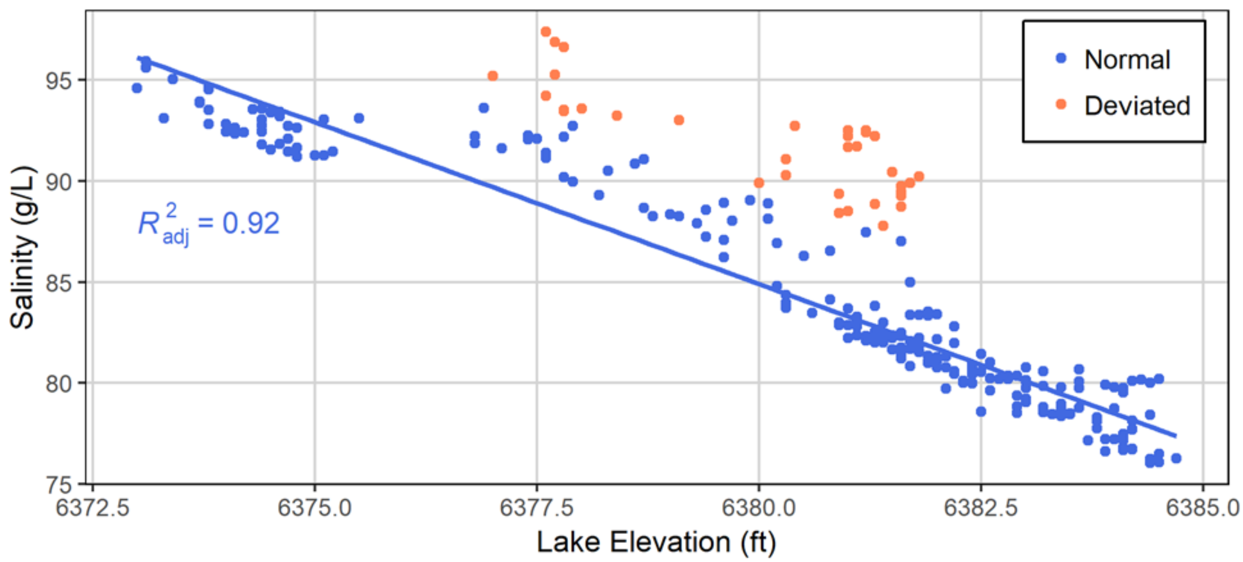

Figure 3.2-41. Relationship between Mono Lake Elevation and Salinity Averaged between 11 and 20 m of Depths at Station 6

Blue line indicates a best fit line based on blue colored points only

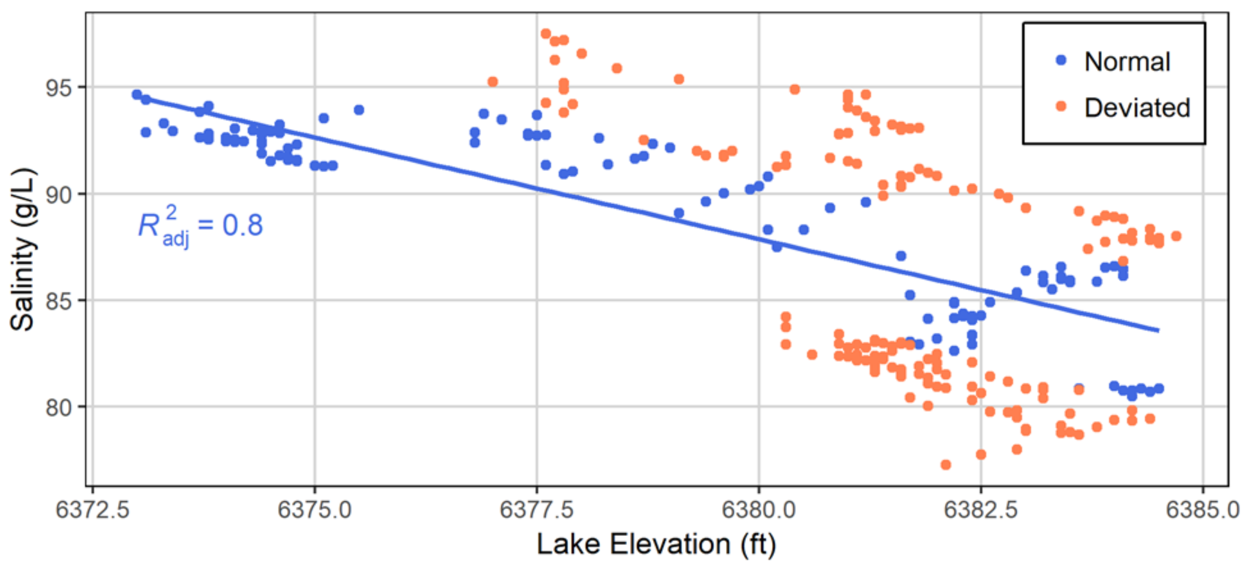

Figure 3.2-42. Relationship between Mono Lake Elevation and Salinity Averaged between 21 and 38 m of Depths at Station 6

Blue line indicates a best fit line based on blue colored points only

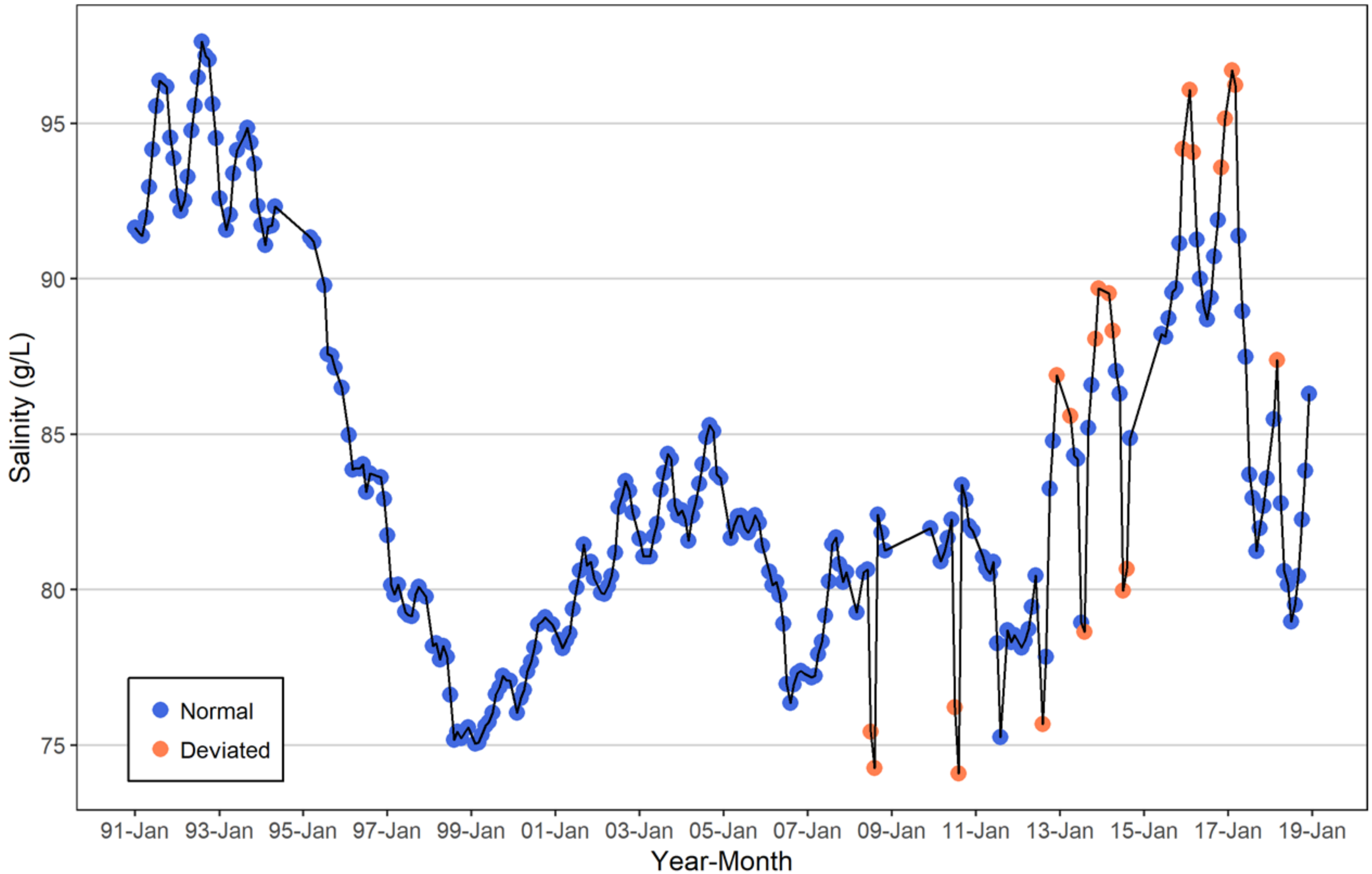

Figure 3.2-43. Time Series Plot based on Monthly Salinity Averaged between 1 and 10 m of Depths at Station 6

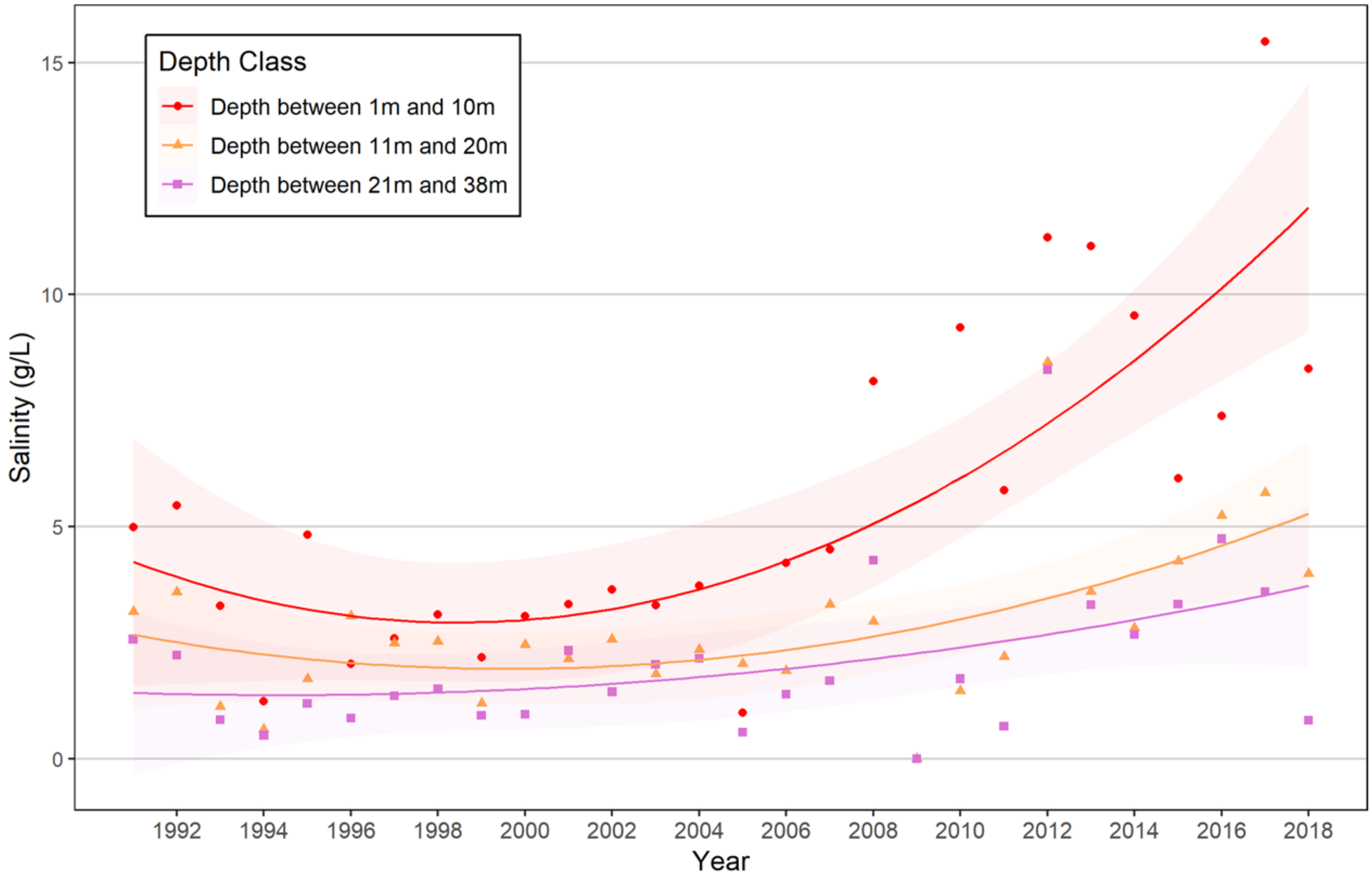

Figure 3.2-44. Inter-annual Range of Monthly Salinity Readings at Station 6

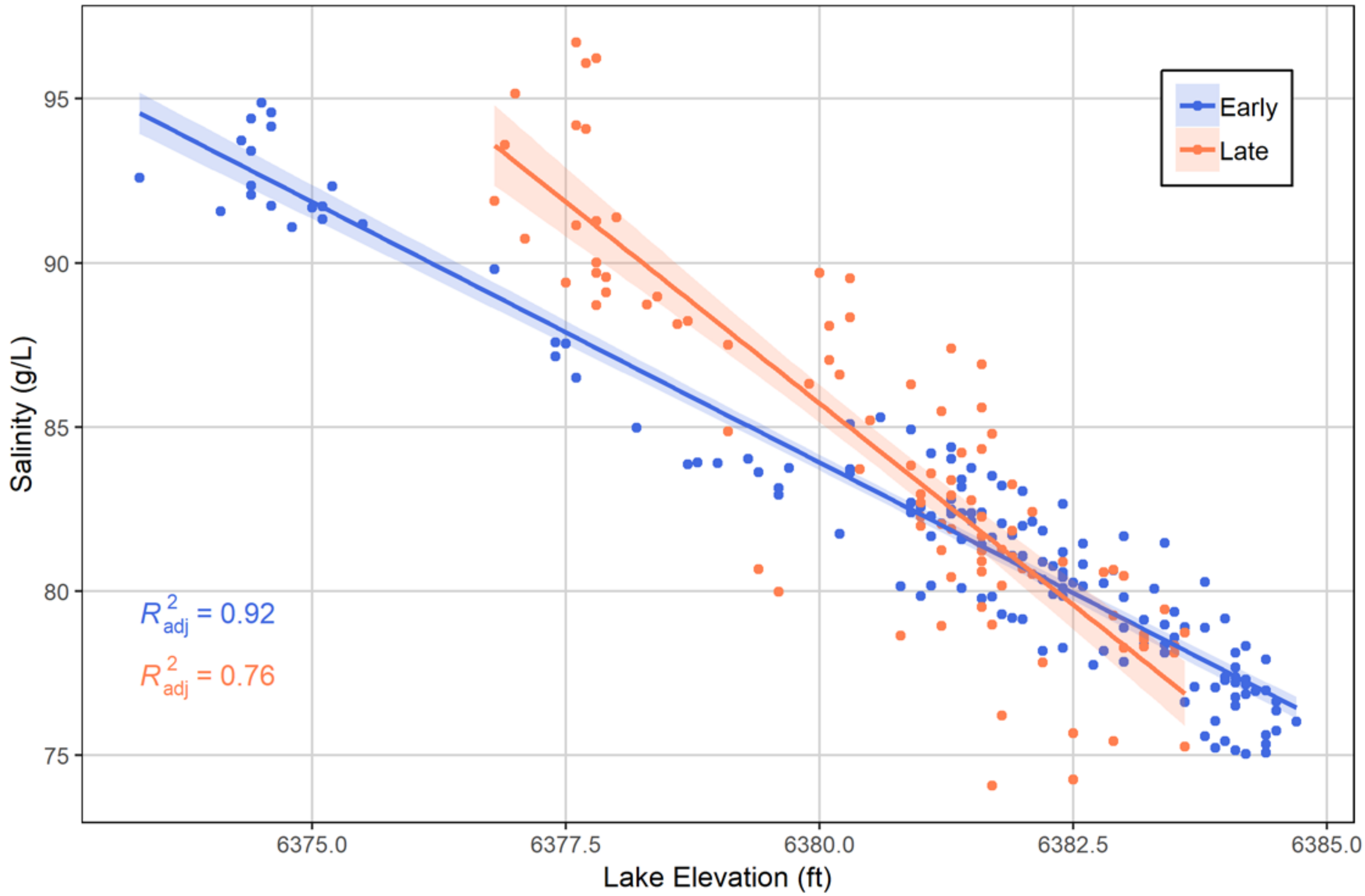

Figure 3.2-45. Difference in slopes between 2 periods of monitoring years: Earlier (1991-2008) and Later (2009-2018)

Average salinity values between 1 and 10 m were used.

Artemia Population Peak

Lake-wide mean *Artemia* population peaked in 1989, 2004, 2009, and 2013 and showed a declining trend with an average decline of approximately 500 m^{-2} per year (Figure 3.2-35). According to this relationship, the *Artemia* population would be approximately 22,860 m^{-2} if the current meromixis breaks in 2020. A predicted peak would be indistinguishable from any other monomictic years which range from 7,676 m^{-2} to 27,639 m^{-2}. In spite of a declining trend of peaks, *Artemia* abundance during peak years was significantly different from that during non-peak years ($P = 0.0099$) as peak years averaged 30,102 m^{-2} compared to 17,206 m^{-2} during non-peak years (Table 3.2-17). The magnitude of population peak, however, is greatly influenced by the duration of meromixis; as a result, the longer the current meromixis lasts the higher the *Artemia* population peak would be when the meromixis breaks. The population peak, therefore, may be much higher than what predicted by a simple linear regression. This section examined the effect of meromixis on the *Artemia* population in Mono Lake.

Ammonium (NH_4)

Ammonium recorded at the deepest monitoring depths (28 and 35 m) shows a similar trend as the *Artemia* population peaks. Peak monthly accumulation prior to the *Artemia* population peak during the second meromixis was 1,131 µM in A 2001 (Figure 3.2-46) with the average rate of accumulation being 124 µM/year, and for successive peaks ammonium accumulation dropped from 107 µM in 2007 to 52.4 µM in 2012. Decline of *Artemia* population peaks during the same period indicates the importance of nutrient build up which appeared to be proportional to duration of meromixis. The maximum accumulation during the current meromixis is 102 µM as of December. This value is one magnitude smaller than the peak during the second meromixis; however, with continuation of meromixis throughout 2019, the maximum accumulation should increase.

When meromixis broke down, accumulated ammonium became available throughout the water column, and a nutrient boost above 10 m of depth was apparent in 2004 but only slightly in 2009 and 2013 (Figure 3.2-47). Fluctuation in ammonium availability above 10 m, however, does not follow the clear pattern of hypolimnetic ammonium accumulation as more ammonium was available in 2016, a monomictic year which did not follow meromixis, than in 2004 and 2009. A lower amount of epilimnetic ammonium availability during the third and fourth meromixis may explain reduced *Artemia* peaks following the meromixis, and ammonium accumulation as of December in 2018 suggests a population peak following the current meromixis could be at least as high as that observed in 2007.

Table 3.2-17. Artemia Population Summary during Meromixis and Monomixis

		Peak				
Meromixis	Duration	Year	Artemia abundance	Average *Artemia* between peaks	Reduction following a peak	NH_4 accumulation during meromixis*
1983-1987	5	1989	36,359		45%	NA
				16,576		
1995-2002	8	2004	32,044		44%	1,131
				17,514		
2005-2007	3	2009	25,970		43%	107
				17,529		
2011	1	2013	26,033		48%	52
				11,822		
2017-						102
Average			30,102	15,860	45%	

* Maximum monthly hypolimnetic NH_4 reading during a meromixis period recorded at depth of 28 m at Station 6.
A unit of NH4 measurement is μM.
Artemia abundance is express as m^{-3}.

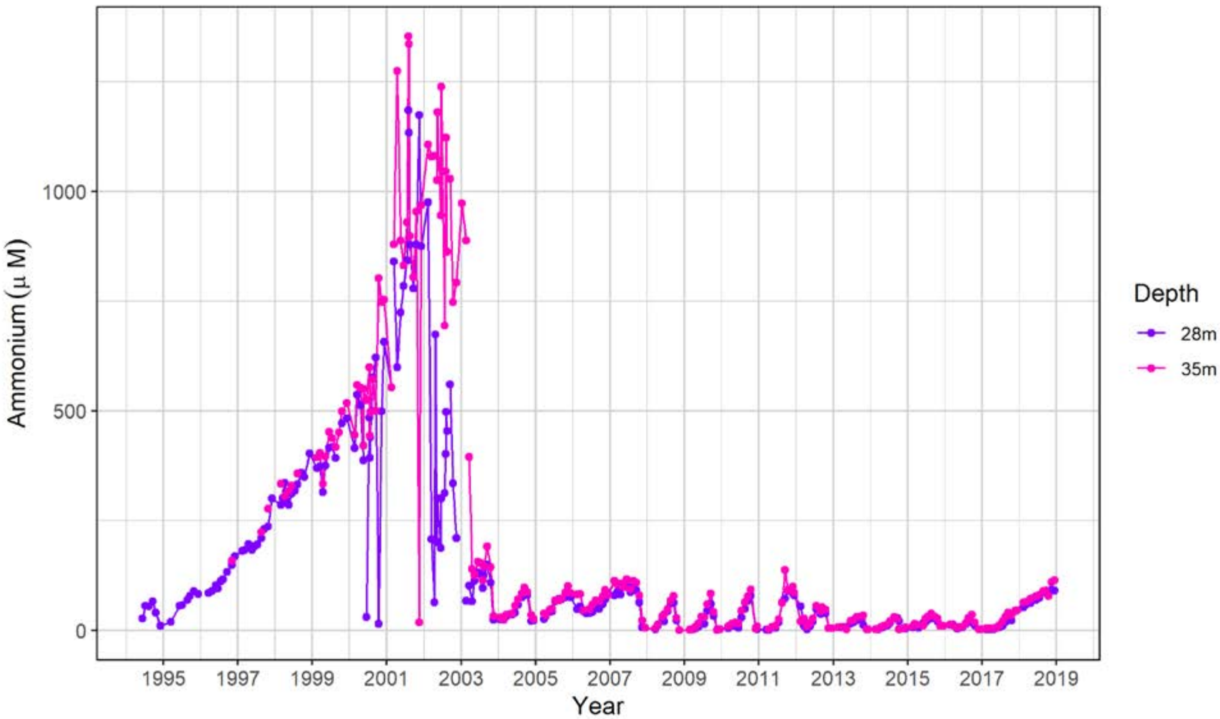

Figure 3.2-46. Ammonium Accumulation at 28 and 35 m of Depths at Station 6

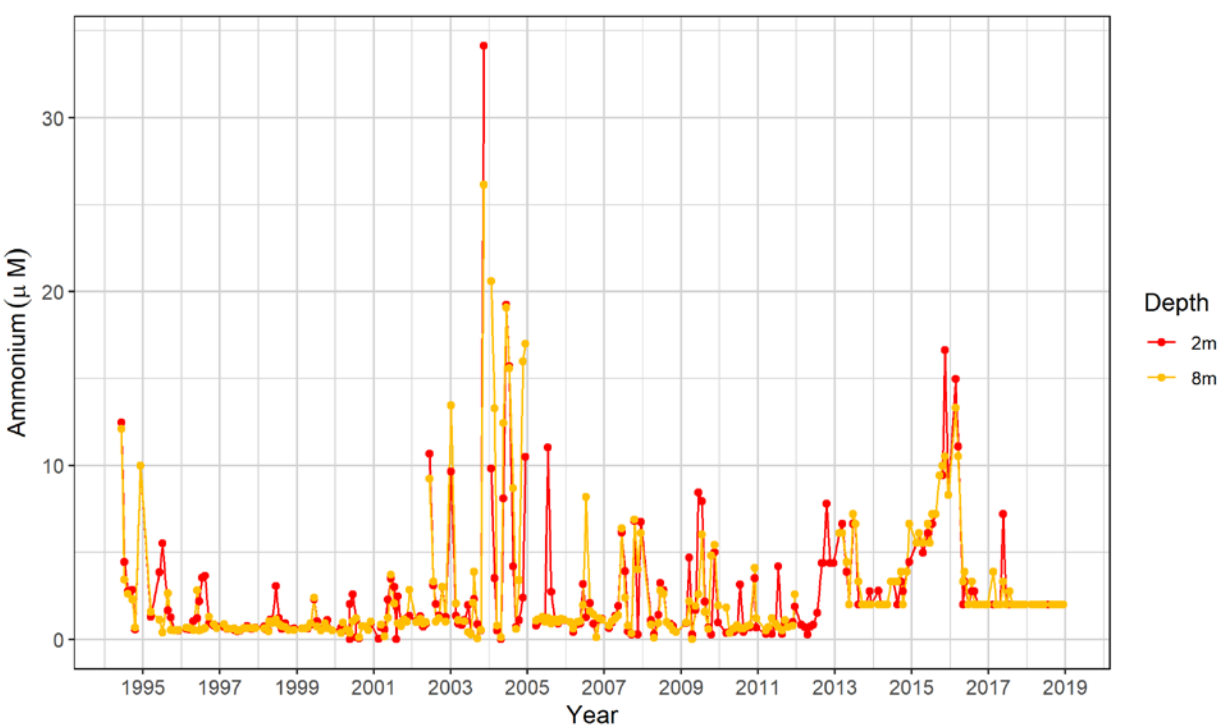

Figure 3.2-47. Ammonium Accumulation at 2 and 8 m of Depths at Station 6

<u>Mono Lake Input</u>

Rush and Lee Vining Creeks constitute the majority of freshwater input into Mono Lake since export restrictions first began in the late 1980s. The second meromixis was by far the longest recorded meromixis lasting from 1995 to 2002. A majority of increased freshwater input occurred between 1995 and 1999 during which a total of 717,670 AF of water discharged into Mono Lake, the highest 5 year total on record (Table 3.2-18). Mono Basin runoff total during the first meromixis was higher than during the second meromixis (179,139 AF between 1982 and 1986 compared to 164,880 AF between 1995 and 1999); however, due to export from Mono Basin, inflow to Mono Lake was larger during the second meromixis than the first. As a result the lake level rose by 10.3 feet during the second meromixis compared to 6.2 feet during the first. Based solely on freshwater influx the second meromixis should have produced a much higher *Artemia* peak than the first meromixis.

The rise in the lake level in 2017 was comparable to what was observed during the third meromixis (2005 to 2007), 3.9 feet compared to 4.0 feet. The Mono Lake input between October 1, 2017, and September 30, 2018, was 97,022 AF, just above 100% of Normal since 1982. This was enough to maintain the lake level at where it had started in January and also to maintain the meromixis which had started in 2017. With the projected April 1 snowpack close to 150%, the meromixis most likely will persist into 2020; thus, the current meromixis would last three or more years. The third meromixis lasted three years. Subsequently, the *Artemia* peak following the current meromixis could have a potential to achieve an *Artemia* peak closer to the third meromixis in terms of magnitude or higher. A longer meromixis is achieved with higher sustained inflow to Mono Lake; the longer the period of sustained high flow, the longer the duration of meromixis.

As mentioned previously, a longer meromixis results in a greater accumulation of ammonium, which in turn results in a higher *Artemia* population peak. The current meromixis, therefore, could reverse the trend of declining population peaks.

Table 3.2-18. Mono Lake Input during Meromixis and Monomixis

Meromixis	Total Input (10^3 AF)	Input responsible to form meromixis			Average Input (10^3 AF) for all other years	Lake Elevation Change (ft)
		Year	Total (10^3 AF)	Average (10^3 AF)		
1983-1987	535	1983-1987	510	128		6.2
					61	
1995-2002	971	1995-2002	718	144		10.3
					83	
2005-2007	375	2005-2007	313	156		4.0
					73	
2011	162	2011	162	162		1.9
					59	
2017-	334	2017	236	236		3.9
Average				165	69	

<u>Salinity</u>

With a large influx of freshwater, epilimnetic salinity declines. During the second meromixis, the salinity gradient slowly developed with the onset of meromixis peaking at 15.84 g/L in August 1998 and disappearing in 2003, a year before the *Artemia* population peak (Figure 3.2-48, Figure 3.2-49). During the third meromixis, however, the magnitude of the salinity gradient was much smaller peaking at 6.0 g/L in July and August 2006 and the duration was much shorter as well. A smaller gradient may be attributable to the lower salinity environment at the beginning of the meromixis, the low 80's in 2005 comparted to over 90 g/L in 1995. Since the third meromixis, inter-annual ranges of salinity had increased considerably; as the result, the meromixis in 2011 failed to create a salinity gradient which was distinguishable from monomictic years. Higher inter-annual ranges are accentuated by rapidly climbing salinity between 2012 and 2016, the driest 5 year period on record, after the short meromixis in 2011. In 2017, the salinity gradient exceeded 20 g/L. Influx of freshwater appears to be capable of lowering salinity below 80 g/L at least briefly regardless of the total amount of runoff; however, the overall rising trend remains unchanged unless consecutive years of higher runoff happens. Due to almost record input in 2017 and close to 100% of Normal input in 2018, maximum salinity values have started to decrease from 98 g/L in February, 2017, to 93 g/L in December, 2018. The current trend should continue throughout 2019. Chemocline during the second meromixis was strong and lasted for 8 years, resulting in greater accumulation of ammonium and subsequently a higher *Artemia* population peak in 2003. The current chemocline is much stronger than during the second meromixis but more variable, and the duration remains to be seen. If the current meromixis persists beyond next year, *Artemia* population peak could become higher than the second meromixis.

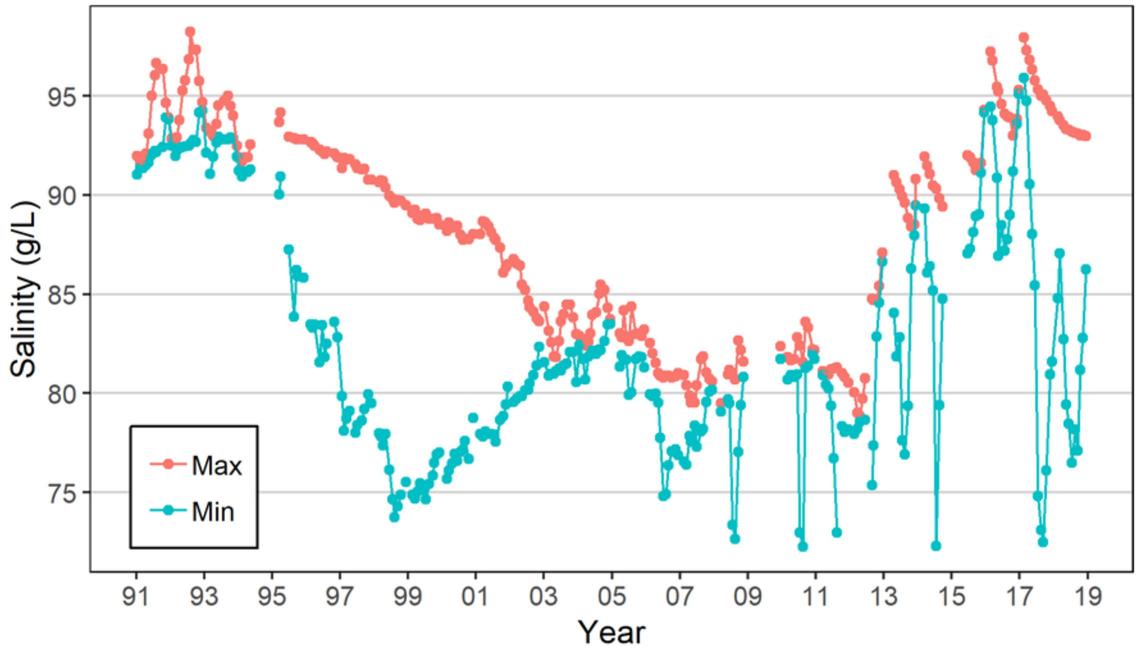

Figure 3.2-48. Maximum (red) and Minimum (blue) Salinity through Water Column at Station 6

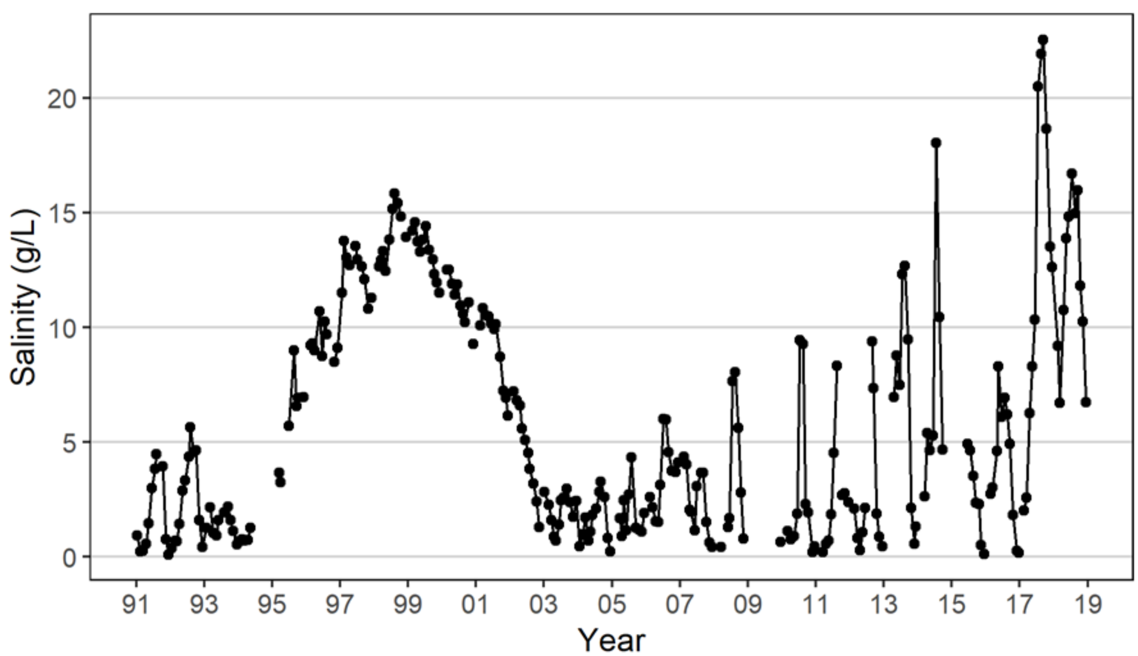

Figure 3.2-49. Salinity Range through Water Column at Station 6

A Temporal Shift in Monthly *Artemia* Abundance

There has been a clear temporal shift in peak abundance of instars and adults as monthly peaks are occurring earlier in the year (Figure 3.2-36, Figure 3.2-37), which are reflected on a strong linear negative trend of centroid days (calculated center of abundance of adults) in respect to monitoring years (Figure 3.2-34). There appear to be three distinct periods of instar and adult abundance patterns; 1) later season occurrence between 1987 and 1994, 2) transition between 1995 and 2003, and 3) earlier season occurrence since 2004. The first period coincides with the breakdown of the first recorded meromixis and subsequent monomixis, and monthly peaks tend to occur in June except 1987 and 1988 for instars and mostly in July or later for adults. High adult abundance (over 100,000 m^{-2}) is maintained into fall. The transition period coincides with the second meromixis which lasted from 1995 through 2003. During this period monthly peaks have shifted earlier for instars while monthly peaks remain in June for adults even though high abundance in fall is no longer recorded. The third period features two short lived meromictic events and current meromictic event. Peak monthly instar abundance tends to occur mostly in April but as early as in March during some years while peak monthly adult abundance tends to occur in June and as early as in May. This trend, however, appears to be reversed slightly in recent years as a half of monitoring years show peak monthly abundance in July since 2010. In 2015 and 2016, a peak monthly abundance is observed in June; however, each peak is smaller but broader such that July monthly abundance is almost as high as that of June and July in the case of 2016.

Chlorophyll *a*

Increasing food abundance in earlier months (spring to early summer) could facilitate higher growth rates of *Artemia*. Annual fluctuations of chlorophyll during spring months show a positive trend at deeper depths throughout the year and at shallower depths in late spring to summer (Figure 3.2-50, Table 3.2-19). There also appears a cyclic pattern of chlorophyll levels as lower levels are found during meromictic years while higher levels are found during monomictic years. Data prior to 1995 is not available for the analysis; thus, it is not possible to assess whether a positive trend has existed including data prior to 1995. Chlorophyll levels should have been higher during the monomixis in the early 90’s; and this coincided with earlier monthly population peaks between 1992 and 1995. The positive trend, therefore, may be the artifact of duration of the data; however, it appears to support that higher abundance of food sources leads to early *Artemia* population peaks.

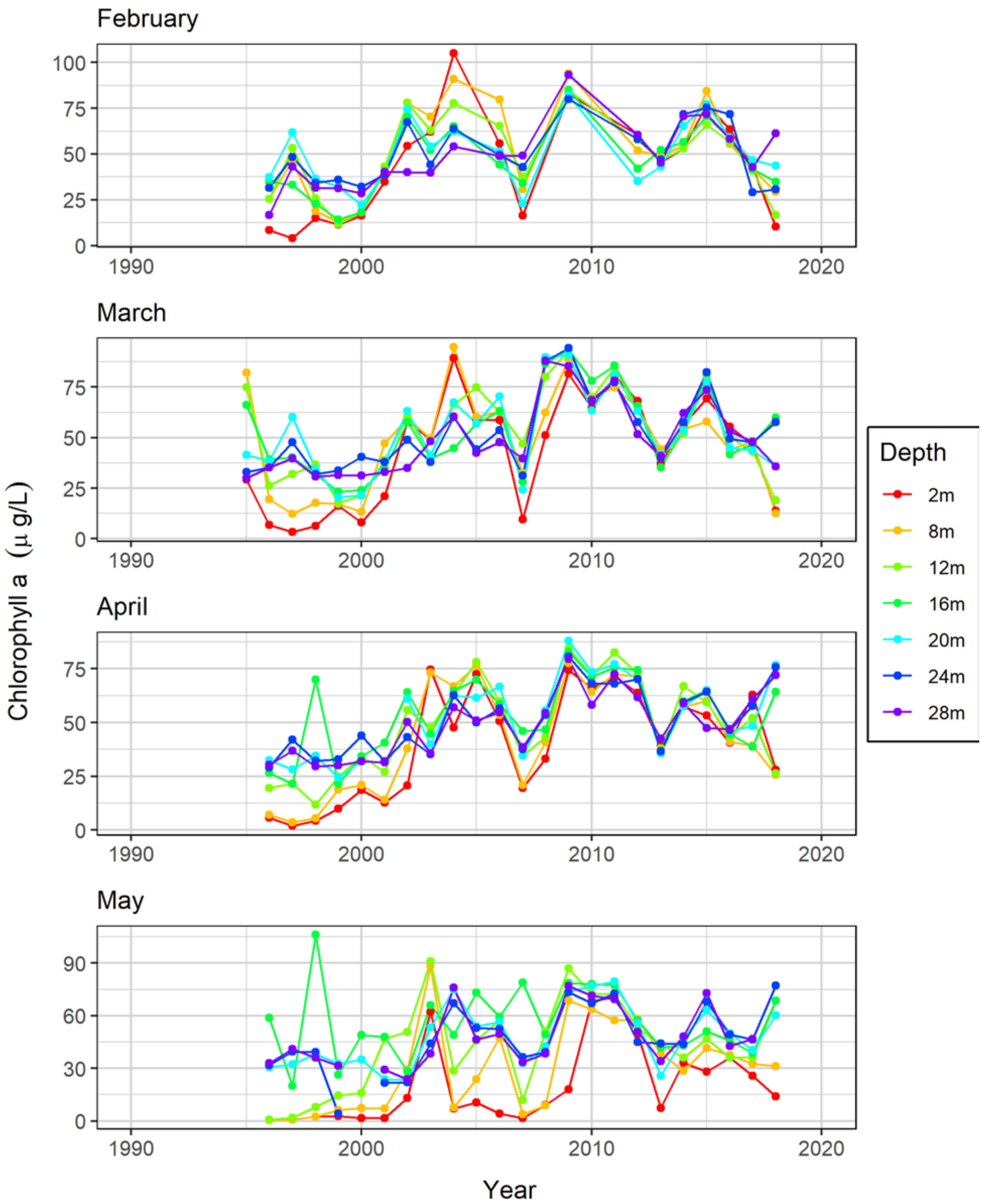

Figure 3.2-50. Chlorophyll a Level over Time at All Depths between February and May

Table 3.2-19. Correlation Coefficients (r) between Monthly Chlorophyll a Values and Monitoring Years

	Depth						
Month	2m	8m	12m	16m	20m	24m	28m
Feb	0.39	0.27	0.19	0.42	0.27	0.35	0.67 **
Mar	0.48 *	0.21	0.24	0.39	0.29	0.51 *	0.51 *
Apr	0.59 **	0.47 *	0.46 *	0.39	0.55 **	0.66 ***	0.69 ***
May	0.47 *	0.49 *	0.48 *	0.05	0.44 *	0.58 **	0.44 *
Jun	0.69 ***	0.69 ***	0.69 ***	0.45 *	0.38	0.57 **	0.67 ***
Jul	0.45 *	0.53 **	0.78 ***	0.63 ***	0.69 ***	0.60 **	0.67 ***
Aug	0.47 *	0.57 **	0.70 ***	0.52 **	0.66 ***	0.74 ***	0.69 ***
Sep	0.54 **	0.53 **	0.79 ***	0.62 ***	0.31	0.62 ***	0.46 *
Oct	0.32	0.34	0.46 *	0.49 *	0.57 **	0.66 ***	0.67 ***
Nov	0.09	0.21	0.41	0.55 *	0.47 *	0.59 **	0.65 **
Dec	0.52 *	0.56 **	0.69 ***	0.60 **	0.67 ***	0.65 ***	0.72 ***

* indicates significance at 0.05, ** indicates significance at 0.01, and *** indicates significance at <0.01.

<u>Temperature</u>

Following the obligate period of dormancy, warmer water temperature is found to lead to shorter time required for hatching (Dana et al. 1988). Hypolimnetic water temperature remains relatively high during meromixis which reduces convection across the chemocline, resulting relatively warm and stable water temperature condition in hypolimnion. This is evident clearly during the second meromixis (1995-2002) and the current meromixis (2017 to present), but only somewhat evident during the third meromixis (2005-2007) (Figure 3.2-51). The trend becomes more apparent when hypolimnetic water temperature (<20 m) is averaged over February and March (Figure 3.2-52). The monomictic period between 1991 and 1995 (meromixis has not started in spring of 1995 yet) show lower spring hypolimnetic water temperature than the subsequent meromixis, averaging 2.6°C compared to 4.1°C during the meromixis. Spring hypolimnetic water temperature starts dropping as meromixis weakens, and this is evident between 2002 and 2005, in 2008, and somewhat between 2014 and 2016. The hypolimnetic water is slightly warmer during monomixis since 2003, 2.6°C compared to 2.4°C between 1991 and 1995. During the last monomictic period (2013-2017[2]), however, spring hypolimnetic water temperature averaged 3.2°C; thus, spring hypolimnetic water temperature appears warmer compared to earlier 90s, consistent with the trend found for *Artemia* instar monthly abundance. Warmer hypolimnetic water appears to support earlier hatching of cysts, hence, earlier peak of instar abundance;

[2] The current meromixis did not start until the summer of 2017.

however, the epilimnetic water temperature does not appear to explain the reversing of the peak monthly abundance.

Ambient temperature strongly affects water temperature especially at shallower depths. For instance between 0 and 3 m of depth the relationship between water and air temperature yields a coefficient of determination of 0.91 (Figure 3.2-53). For hypolimnetic water, it takes approximately 6 months to show any meaningful statistical relationship (Figure 3.2-54). Summer ambient temperature influences winter hypolimnetic water temperature or winter ambient temperature influences summer hypolimnetic water temperature. These relationships are much stronger only during monomixis when the lake becomes isothermal in winter. A combination of warmer summer and winter can lead to a gradual increase in hypolimnetic water temperature, which in turn can lead to earlier hatching.

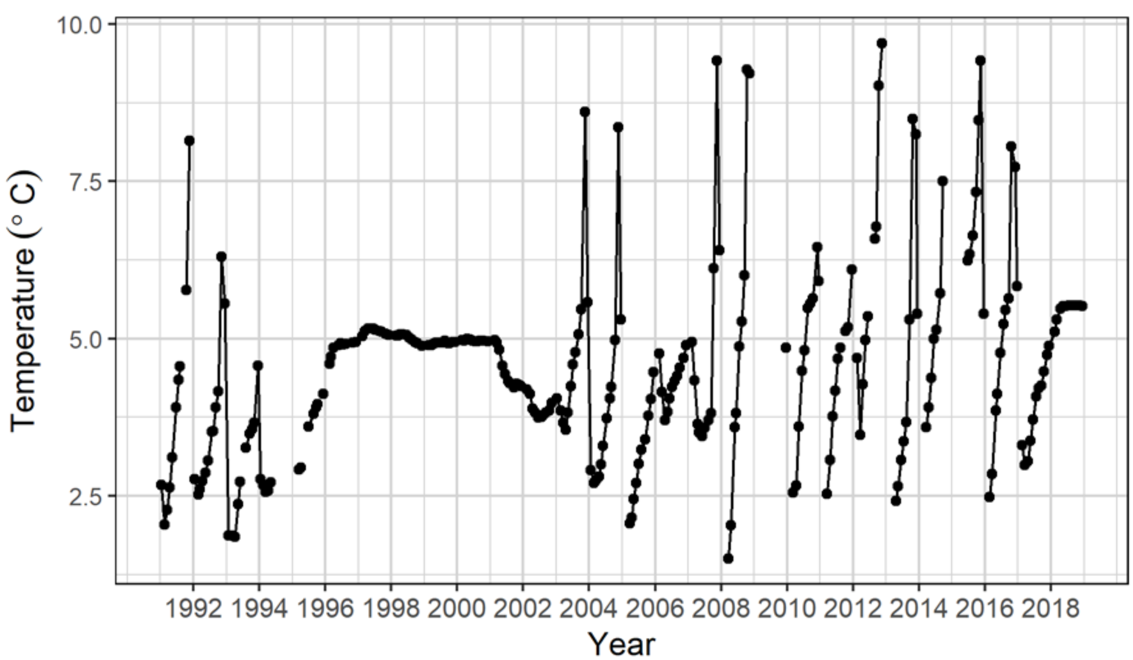

Figure 3.2-51. Average Water Temperature between 30 and 40 m of Depths at Station 6

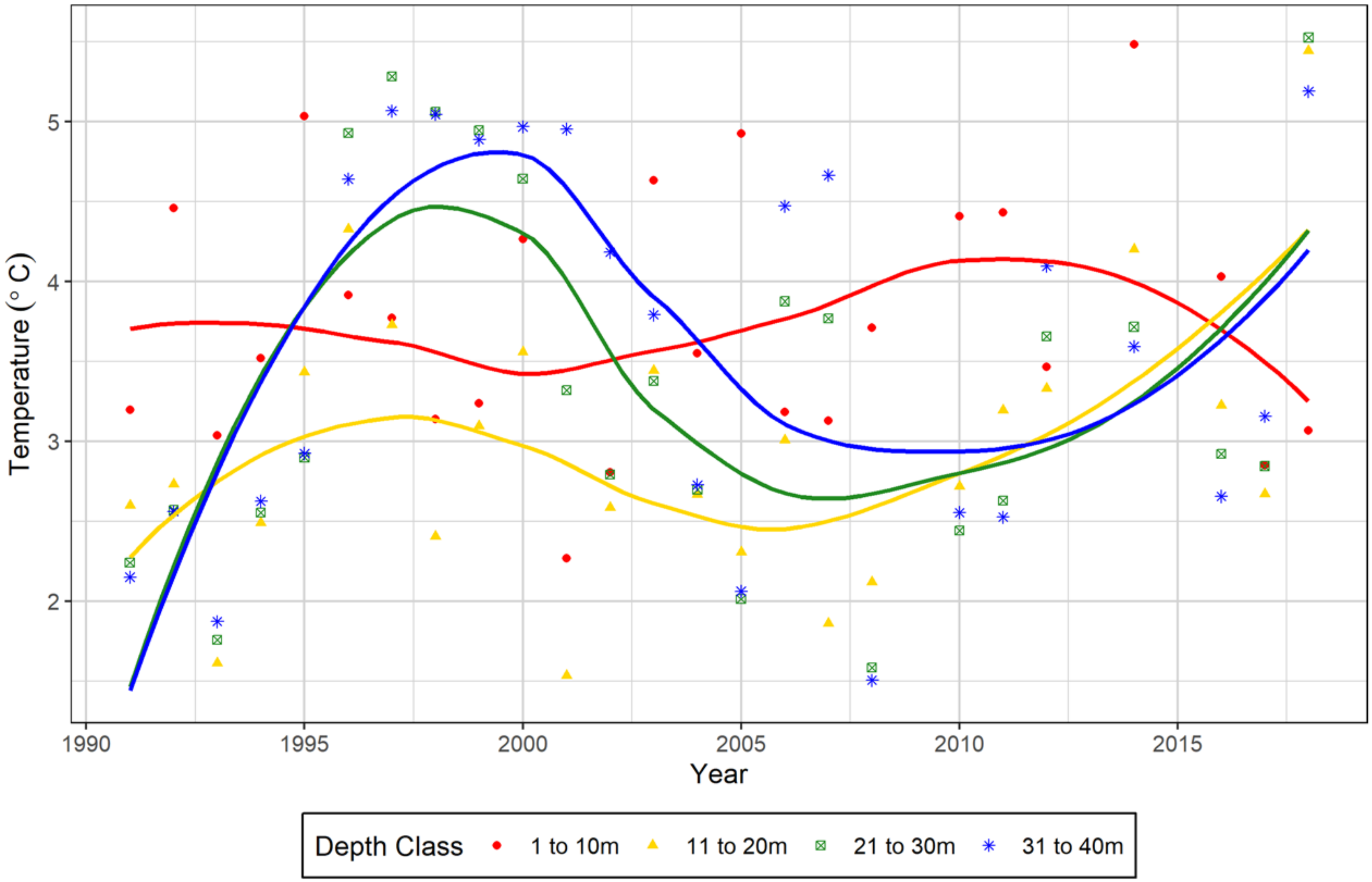

Figure 3.2-52. Average Water Temperature at 4 Depth Classes in Spring at Station 6

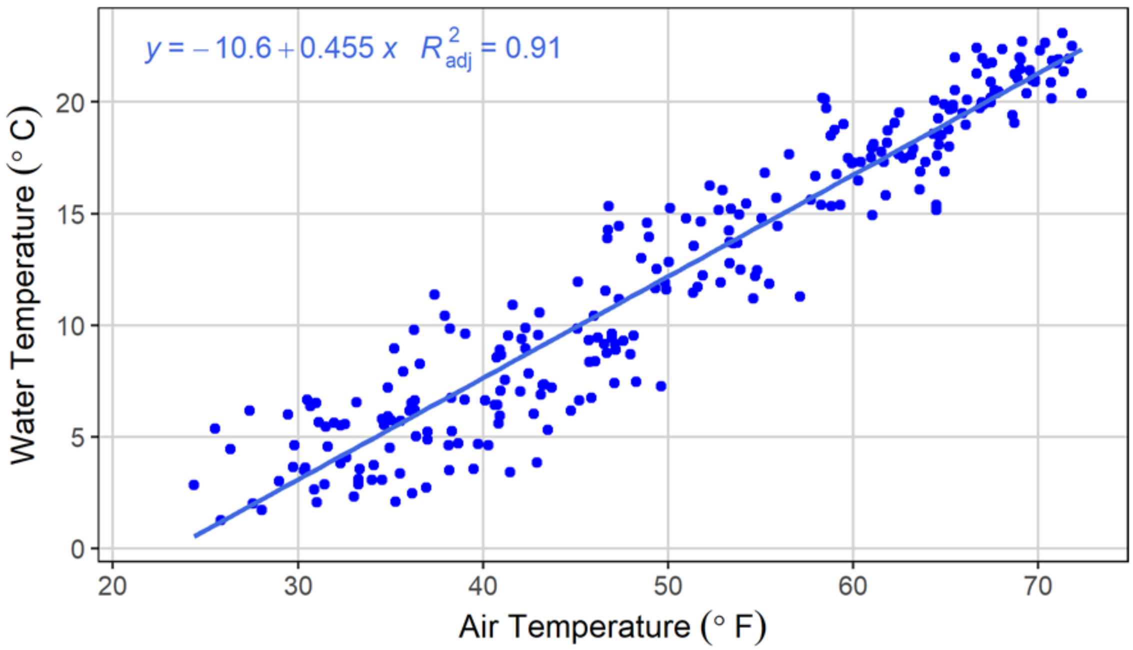

Figure 3.2-53. Relationship between Water and Air Temperature at Depths between 0 and 3 m at Station 6

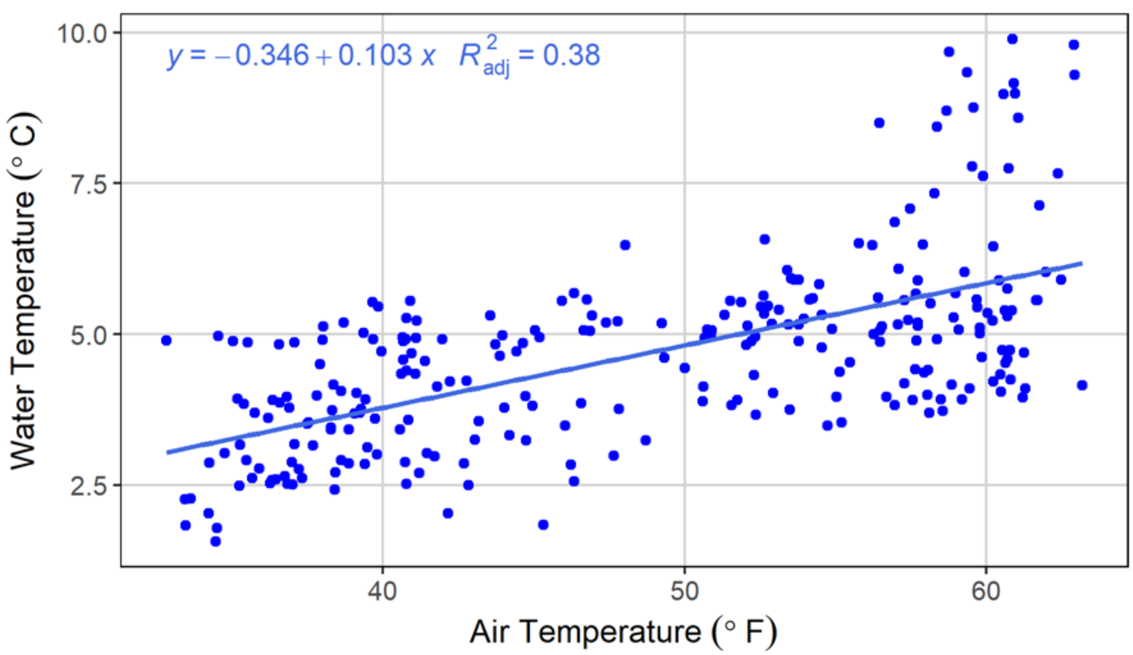

Figure 3.2-54. . Relationship between Water and Air Temperature at Depths between 20 and 40 m at Station 6

There is a lag of 6 months for air temperature.

3.2.4 Limnology Discussion

2018 Condition

The 2018 monitoring year marked the 2nd year of the 5th meromictic event since the beginning of the program. Mono Lake rose by 4.0 feet from 6377.2 feet in January to 6381.2 feet in September in 2017, and almost remained static in 2018 (6381.1 feet in December). The chemocline established persisted at the depths between 8 and 15 m with the salinity gradient peaking at 16.8 g/L in July. The gradient decreased to 6.8 g/L at the end of the year. As a result, the hypolimnion remained mostly anoxic, and ammonium was deprived from the epilimnion and accumulated in the hypolimnion. The *Artemia* population decreased from 15,158 m^{-2} in 2017 to 12,120 m^{-2} in 2018. The value was higher than the record low of 7,676 m^{-2} in 2015, but still remained far below the long term average of 18,951 m^{-2}. Due to an average influx of freshwater and decreased *Artemia* population, clarity of the lake was not as good as in 2017. For the third year in row a centroid (the calculated center of abundance of adults) remained above 220 days reversing the long term declining trend. Monthly instar abundance peaked in April.

The 2018 *Artemia* population was lower than the average between peaks but falls within the value expected from the last population peak in 2013 (26,033 m^{-2}). The *Artemia* population tends to drop consistently during years following the population peak by the average of 45%, 30,102 m^{-2} during peaks compared to 16,570 m^{-2} for years immediately following the peak and tends to hover around this nonpeak average value during monomixis (Table 3.2-19). The *Artemia* population had declined in 2015 recording the lowest abundance on record; however, it has rebounded since then, showing resiliency of the *Artemia* population in Mono Lake. Historically the *Artemia* population also has demonstrated resiliency. The *Artemia* population has rebounded in spite of the lake level declining to the lowest level of the past century at 6,371.6 feet in December 1981 as Mono Lake input has started to increase. Salinity in the beginning of 2017 was the highest since 1991 despite of the lake level being almost 4 feet higher than during the early 1990s. It has been demonstrated that salinity affects the survival, growth, reproduction, and cyst hatching of *Artemia* (Starrett and Perry 1985, Dana and Lenz 1986). Five years of drought between 2012 and 2016, the worst five year period on record, has resulted in the lake level declining from 6,383.6 feet in April 2012 to 6,376.8 feet in October 2016; consequently salinity between 1 and 10 m of depth increased from 75.7 g/L in August 2012 to 96.7 g/L in February 2017 (Table 3.2.20). Epilimnetic salinity declined to 86.3 g/L in December, 2018. Increasing salinity most likely contributed to lower *Artemia* abundance especially under the condition that salinity was higher than expected given lake levels during this period as discussed previously. With the second largest input into Mono Lake from tributaries, salinity decreased 80.9 g/L in September and remained at 83.7 g/L in December 2018. The *Artemia* population responded positively to declining salinity by increasing from 7,676 m^{-2} in 2015 to 15,158 m^{-2} in 2017, but did not respond to continuous improvement in salinity during 2018. Mono Lake remained stratified at the end of 2017 and throughout 2018; consequently, hypolimnetic nutrients did not become available in epilimnion. An amount of runoff determines lake level and salinity, and higher inputs helped the *Artemia* population to rebound in 2017, but a lack of nutrients in hypolimnion due to the meromixis may have contributed to lower Artemia population in 2018.

Peak monthly instar and adult *Artemia* population abundance occurred in April and July, respectively, which is consistent with the long-term trend for instar peak even though the adult peak is toward the later range of the months. Hypolimnetic spring (February and March) water temperature in 2018 was 5.4°C in 2018, higher than the average of 3.9 C found during the second meromixis between 1996 and 2002. Lower chlorophyll a levels throughout water columns in spring due to the meromictic condition. Warmer water temperature may have favored earlier instar peak in 2018, but lower food availability may have resulted in more mixed responses in timing and magnitude of adult population peak. Individual water parameters are not adequate to explain timing and magnitude of adult and instar population peaks.

Future Condition

Future limnological condition of Mono Lake will largely depend on future runoff conditions. A lack of prolonged meromixis leads to smaller *Artemia* peaks and lower abundance during subsequent monomixis. Since the end of the third meromixis (1995-2002), the longest duration of wet period is 2 years (2005 to 2006) which resulted in 3 years of meromixis. Without a prolonged period of meromixis, ammonium accumulation remains magnitudes smaller than the accumulation level between 1995 and 2002.

A lack of sustained high freshwater input or extremely large freshwater input will also result in higher salinity. When the lake level dropped by 6.8 feet salinity increased by 19 g/L from 78.7 g/L to 97.5 g/L in hypolimnion at the lake level of 6,377 feet. Between 1995 and 1999, the 200% of Normal Mono Lake Input from Rush and Lee Vining Creeks, dropped hypolimnetic salinity by 15.7 g/L, from 91.3g/L to 75.6 g/L. The same decline was observed after 2018 with 308% of Normal Input. The relationship between salinity and lake elevation, however, appear to be changing as salinity tends to be higher when the lake level is below 6,380 feet but lower when the lake level is above 6,380 feet. It is evident from Figure 3.2-21 that a prolonged wet period could have lasting effects on salinity. After the four consecutive years of above Normal Mono Basin runoff, the salinity declined from 93.6 g/L in April, 1995, to 82.7 g/L in April, 1999. This wet period had much lasting effect on salinity because the input from Rush and Lee Vining was maintained at or above 80,000 AF until the next wet years (2005 and 2006).

A prolonged wet period is necessary to lower the hypolimnetic salinity by more than 10 g/L, and once it is low the input above 80,000 AF may be sufficient to maintain salinity at that level. Prolonged drought could have an opposite effect as seen between 2012 and 2016 during which salinity increased from 78.7 g/L to 97.5 g/L. The adverse effect of drought on salinity could become more severe as drier and warmer climate is forecasted for much of California in future (Ficklin et al. 2013). The *Artemia* population in Mono Lake appears to survive and thrive in the salinity levels during monitoring years. However, further decline in the lake level could result in much higher salinity, which could approach the tolerance level (Dana and Lenz 1986).

The estimated salinity level at 6,392 feet ranges between 66 g/L and 72 g/L depending on the depth. It is not clear whether the *Artemia* population will increase beyond what has been recorded since 1987. As discussed, the *Artemia* population is strongly influenced by strength and duration of meromixis. Lower salinity certainly will result in a weaker salinity gradient or chemocline, such that Mono Lake could become holomictic much more easily than the current state. Without a strong and long lasting chemocline, ammonium accumulation would be lower, which would result in a lower *Artemia* population peak. A higher Mono Lake elevation, therefore, may have very limited impact on the lake's *Artemia* population; however, lower salinity associated with a higher Mono Lake level could lead to "*invasions by predators or competitors of the brine shrimp, which could reduce productivity of the brine shrimp population*" (Jones and Stokes Associates, 1994). At the same time, more diverse invertebrate fauna could lead to increased food sources for shorebirds and waterfowl populations.

3.2.5 Limnology Monitoring Program Evaluation

Background and Methods

The current limnology monitoring program was started in 1998 as a part of the Plan although some limnological work on Mono Lake had been conducted since the late 1970s. *Artemia* adult population statistics date back to 1979 while 9-m integrated data for ammonium and chlorophyll *a* is available as far back as1987 and other water parameter data exist since either 1991 or 1994. It has been well-documented that the lake mixing regime greatly influences *Artemia* population dynamics and water parameters. A wealth of data has accumulated over the years and has led to a better understanding of Mono Lake limnological processes. The Periodic Overview Report evaluated trends and provided recommendations for changes to the Waterfowl Program. An analysis was conducted using all available data to evaluate the limnological program to determine if changes could be made to reduce cost, while maintaining the ability to assess the long term health of the waterfowl habitat on Mono Lake. The results of this analysis suggest that both temporal and spatial reductions in monitoring effort could be made, and thus a proposed modification to the limnological monitoring program is presented.

In 2011, Dr. Brian White, the Waterfowl Program Director at that time, evaluated available data and suggested that the monitoring program could be reduced. Dr. White presented proposed reductions to the SWRCB, but these changes were met with opposition from the Parties and no changes were implemented.

For this report, a program analysis independent of that conducted in 2011 by Dr. White was conducted. The results of this analysis suggest that the current monitoring program can be reduced by:

1. Reducing a number of stations to monitor water parameters and *Artemia* population (spatial reduction) and/or
2. Reducing a number to visitations to the designated stations (temporal reduction).

Currently, conductivity and temperature are monitored at Stations 2 through 8, 10 and 12 (9 stations in total), 9-m integrated samples for ammonium and chlorophyll *a* are taken at Stations 1, 2, 5 through 8, and 11 (7 stations in total), and the *Artemia* population is sampled at all 12 stations. At Station 6 ammonium and chlorophyll *a* samples are taken from 8 (2, 8, 12, 16, 20, 24, 28, and 35 m) and 7 (2, 8, 12, 16, 20, 24, and 28 m) different depths, respectively, and dissolved oxygen is recorded at every meter or less to the depth of around 38 m. Conductivity, temperature, 9-m integrated and *Artemia* population monitoring can be reduced in both time and space while dissolved oxygen and depth profiles of ammonium and chlorophyll *a* can be only reduced in time since these parameters are only monitored at one station.

All parameters of interest are currently sampled at Station 6, and this station will continue to be monitored in the future; thus, it is recognized that Station 6 should be used as the basis of spatial comparison. The adequacy of Station 6 to represent the entire lake was tested by correlating water parameters (temperature, conductivity, and 9-m integrated ammonium and chlorophyll *a*) to the remaining stations. For conductivity and temperature, the depth profiles for each monitoring month at Station 6 were correlated to the depth profiles for the corresponding months at the other eight stations. Monthly values based on 9-m integrated ammonium and chlorophyll at Station 6 were correlated to those recorded at the remaining six stations.

Annual statistics (mean, median, peak and centroid) are indices calculated based on the lake-wide averages of adult *Artemia* population between May and November. Adult *Artemia* population counts were converted to annual statistics at each station, and the stations statistics were correlated to the lake-wide annual statistics in order to determine the representativeness of station. Once representative stations were selected, annual statistics were calculated based on these stations and compared to the lake-wide annual statistics based on all 12 stations.

Artemia population samples are further processed in the lab to estimate the population for each station. *Artemia* individuals are classified into instars, juvenile, female, and male. At 7 stations, instars are further classified into one of seven stages. The instar population count provides insight into how many cysts or/and nauplii have hatched but it appears to be poor predictor of the adult population statistics. Monthly *Artemia* instar population counts were first correlated against the monthly adult population at each station in order to quantify the relationship between instar and adult numbers.

Results

Table 3.2-20 shows the correlation coefficient between Station 6 and the 11 other stations for temperature and conductivity at 5 different depth classes. All stations show very high degree of similarities to Station 6, and correlation coefficients ranged from 0.93 to 0.99. This is particularly true for conductivity with correlation coefficients being mostly around 0.99. The same result was found for ammonium and chlorophyll *a* ranging between 0.91 and 0.98 (Table 3.2-21). Very little variations in water chemical/physical properties occurred across the lake compared to Station 6; consequently, Station 6 can represent the entire lake for water chemical/physical properties.

Correlation coefficients between the *Artemia* lake-wide annual statistics and station annual statistics are presented in Table 3.2-22. For the lake-wide mean, Station 9 shows very strong correlation followed by Stations 6 and 4 while the same three stations show strong correlations with the lake-wide median even though correlations are weaker compared to the lake-wide mean. The lake-wide peaks show strong correlations with Station 10 followed by Stations 9 and 4. The lake-wide centroid shows strong correlations with centroid calculated at Stations 6 and 10, and 6 other stations show correlation coefficients above 0.9.

Table 3.2-20. Correlation Coefficients between Station 6 and Other Stations for Temperature and Conductivity

Temperature											
	Stations										
Depth	1	2	3	4	5	7	8	9	10	11	12
1 to 5m	0.9916	0.9965	0.9974	0.9971	0.9980	0.9982	0.9974	0.9978	0.9966	0.9970	0.9966
6 to 10m	0.9911	0.9961	0.9964	0.9957	0.9975	0.9972	0.9940	0.9908	0.9924	0.8698	0.9937
11 to 20m		0.9749	0.9897	0.9902	0.9507	0.9934	0.9303		0.9809		0.9829
21 to 30m		0.9440	0.9837	0.9952		0.9810	0.9888		0.9643		0.9699
31 to 38m				0.9887							0.9487
Conductivity											
	Stations										
Depth	1	2	3	4	5	7	8	9	10	11	12
1 to 5m	0.9955	0.9967	0.9971	0.9960	0.9976	0.9977	0.9957	0.9957	0.9953	0.9978	0.9932
6 to 10m	0.9903	0.9956	0.9961	0.9958	0.9958	0.9977	0.9953	0.9773	0.9952	0.9664	0.9913
11 to 20m		0.9968	0.9992	0.9991	0.9927	0.9994	0.9868		0.9978		0.9965
21 to 30m		0.6881	0.9952	0.9995		0.9939	0.9958		0.9799		0.9967
31 to 38m				0.9994							0.9967

Table 3.2-21. Correlation Coefficients between Station 6 and Other Stations for Ammonium and Chlorophyll a

	Stations					
	1	2	5	7	8	11
NH4	0.9127	0.9507	0.9430	0.9554	0.9612	0.9237
Chla	0.9666	0.9665	0.9823	0.9716	0.9780	0.9630

Because median and peak have been rarely mentioned in past, a focus should be mean and centroid in order to determine representativeness of stations for *Artemia* annual statistics. Stations 6 and 9 show strong correlations with the lake-wide mean with correlation coefficients of 0.95 and 0.91, respectively. Stations 2, 4, 8, 10, and 12 show correlations above 0.8. For centroid, Stations 6 and 10 show strong correlations with the lake-wide centroid with correlation coefficients of 0.94 for both. Six other stations (2. 3, 4, 7, 8, 9) show correlations above 0.9. Based on strength of correlations for mean and centroid, eight stations (2, 4, 6, 7, 8, 9, 10, and 12) appear to represent the lake-wide mean and centroid adequately. Annual statistics based on these eight stations show very strong correlation to the lake-wide statistics (Table 3.2-23). Coefficients of determination (r^2) for mean and centroid were 0.97 and 0.98, respectively. These eight stations are adequate to represent the *Artemia* population of Mono Lake.

Relationships between *Artemia* instar abundance and adult abundance were much weaker than what seen between stations and annual statistics for adult abundance (Figure 3.2-55, Figure 3.2-56). Correlation coefficients ranged from -0.50 to 0.60, and even all stations were used the relationship did not improve. Maximum monthly instar abundance showed the strongest correlation at r= 0.51 with maximum monthly adult abundance, which only translated into at most 25% of the variation in adult population explained by the instar abundance. Abundance of instars is not a good indicator of adult abundance of the same year. Eliminating February and March would not reduce our ability to understand adult population dynamics; however, March sampling should be retained in order to continue monitoring a temporal shift in monthly instar peaks. A monthly peak usually occurs in April but has occurred in March in past.

Table 3.2-22. Correlation Coefficients between Lake-wide and Station based statistics for Adult Artemia Population

		Station											
Lake-wide	Station	1	2	3	4	5	6	7	8	9	10	11	12
Mean	Apr.to.Nov	0.38	0.84	0.76	0.88	0.77	0.91	0.89	0.86	0.96	0.82	0.64	0.85
	Mean_Stat	0.35	0.85	0.73	0.87	0.78	0.91	0.89	0.86	0.95	0.81	0.63	0.86
Median	Apr.to.Nov	0.45	0.69	0.71	0.83	0.76	0.81	0.80	0.73	0.86	0.60	0.50	0.71
	Mean_Stat	0.42	0.70	0.66	0.80	0.77	0.81	0.77	0.73	0.85	0.59	0.48	0.70
	Median_Stat	0.47	0.79	0.65	0.81	0.78	0.71	0.84	0.76	0.87	0.81	0.66	0.78
Peak	Apr.to.Nov	0.31	0.79	0.65	0.78	0.63	0.81	0.81	0.82	0.88	0.92	0.75	0.87
	Peak_Mo	0.23	0.74	0.58	0.92	0.65	0.83	0.75	0.87	0.86	0.86	0.51	0.69
	Peak_Stat	0.30	0.73	0.63	0.80	0.55	0.83	0.74	0.87	0.87	0.85	0.57	0.72
Centroid	Centroid	0.84	0.90	0.91	0.91	0.89	0.94	0.91	0.92	0.90	0.94	0.81	0.88

Table 3.2-23. Simple Linear Correlation and Regression between the Lake-wide Annual Statistics and Statistics based on 8 Stations

	r	p	y intercept	slope	r^2
Mean	0.98	0.0000	2296.0	0.89	0.97
Median	0.93	0.0000	2634.4	0.84	0.86
Peak	0.98	0.0000	7983.3	0.82	0.96
Centroid	0.99	0.0000	17.1	0.92	0.98

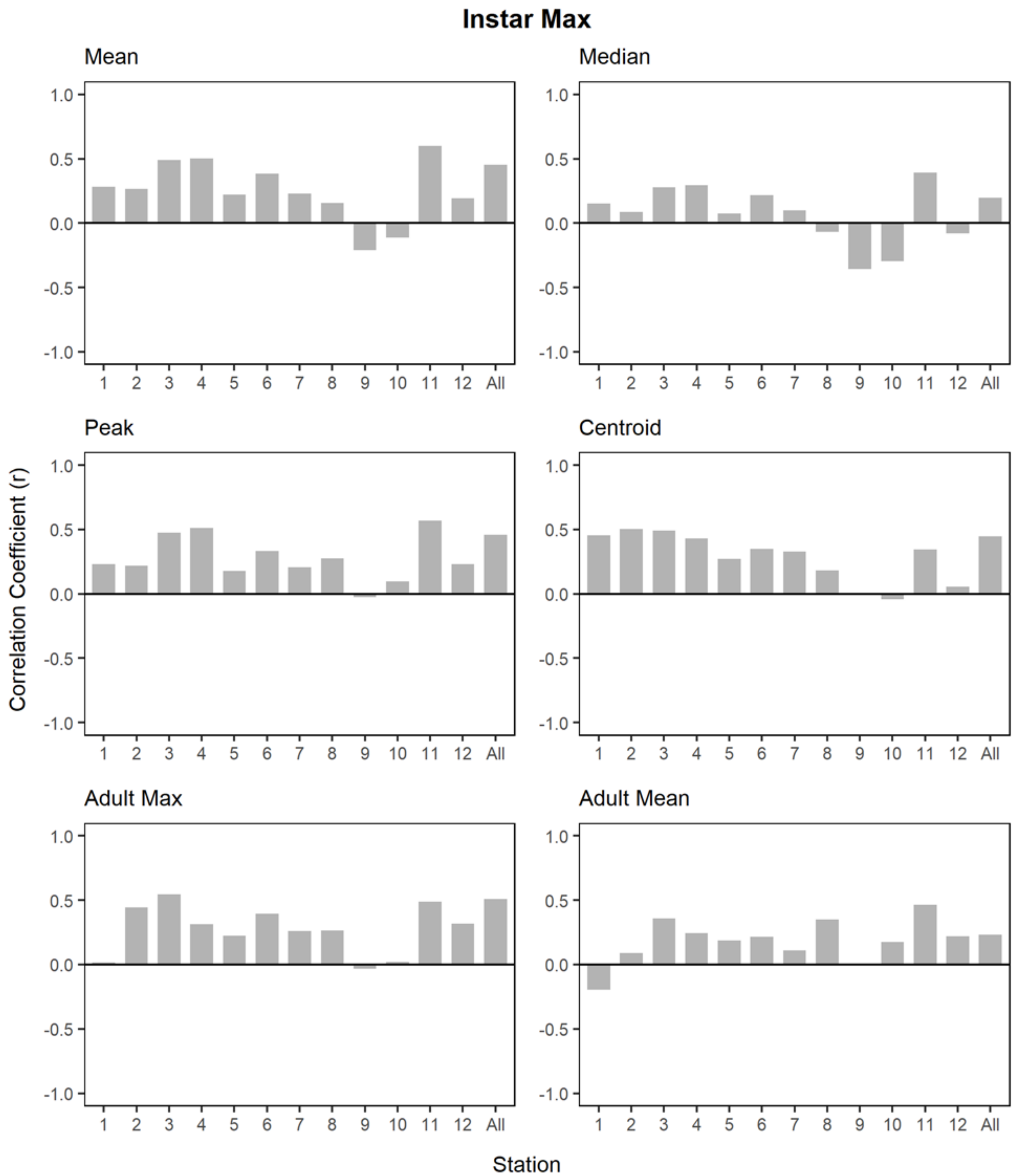

Figure 3.2-55. Correlation Coefficients between Maximum Monthly Instar Abundance and Annual Statistics/Adult Monthly Abundance

Max indicates maximum monthly abundance.
Mean indicates mean monthly abundance between June and September.

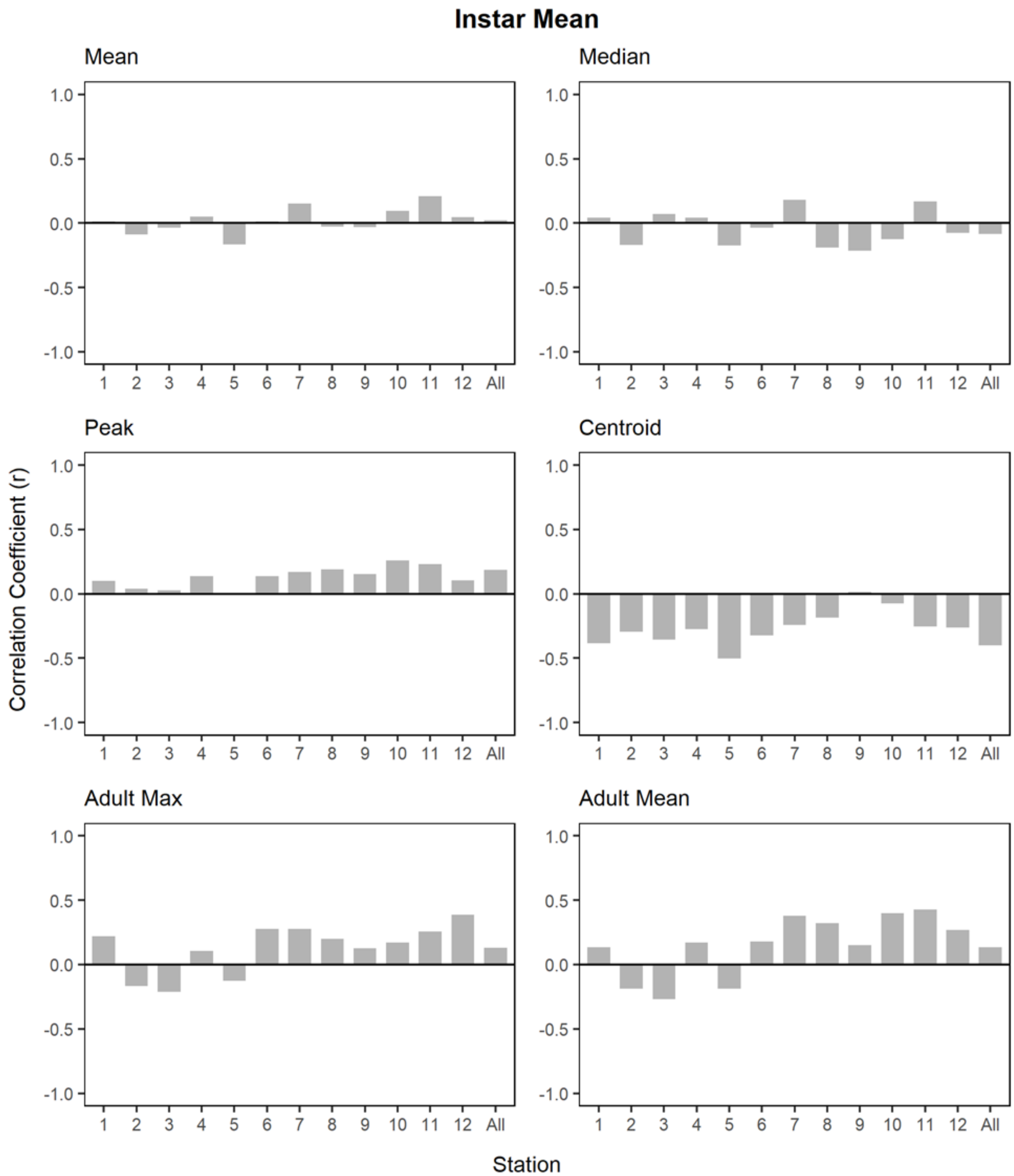

Figure 3.2-56. Correlation Coefficients between Mean Instar Abundance between February and April and Annual Statistics/Adult Monthly Abundance

Max indicates maximum monthly abundance.

Mean indicates mean monthly abundance between June and September.

Conclusion

For water chemical/physical properties, Station 6 alone could represent the all other stations. For *Artemia* population abundance, at least eight stations are necessary to adequately represent the entire 12 stations. Lake-wide annual statistics provide invaluable insight into adult *Artemia* population dynamics as the statistics date back to 1987. It is essential to continue to provide lake-wide annual statistics in future. The above presented analysis, however, indicates that lake-wide annual statistics can be reliably calculated using eight stations (2, 4, 6, 7, 8, 9, 10, and 12) instead of 12 stations. The regression equation can be used to convert annual statistics based on eight stations to the lake-wide annual statics in future. *Artemia* population statistics are calculated using months between May and November; thus, a number of months during which monitoring is conducted can be reduced from 11 (February to December) to 7 (May to November). A temporal shift in monthly peaks for both adult and instar *Artemia* has been observed, and a monthly peak of instar has occurred in as early as March. In order to continue monitoring these two trends, *Artemia* sampling should conducted between March and November. For February and December water chemical/physical properties should be monitored at Station 6 in order to detect a long term trend of hypolimnetic water temperature and salinity, accumulation of nutrients in hypolimnion, abundance of food source throughout water column, and also to see whether the lake becomes holomictic. Coldest water temperature is normally observed in February and during monomictic years the lake normally becomes holomictic in December.

In addition, instar analysis should discontinue. Jellison and Rose (2011) have discussed previously that zooplankton populations can exhibit a high degree of spatial and temporal variability. For this same reason, instar analysis may not yield accurate estimate of *Artemia* demography, especially for instar stages 1 through 7. Sampled *Artemia* individuals should be only classified into instar which includes stages 1 through 7, juvenile which includes stages 8 through 11, and adult male and female. Females still should be further classified accordingly to its ovigery state.

Further, *Artemia* fecundity should be conducted at all eight stations. Fecundity sampling has been conducted at seven stations (1, 2, 5, 6, 7, 8, and 11); but the new proposal would eliminate three of seven stations (1, 5, and 11), leaving only four stations. Conducting at all eight stations will add one more station and 10 more females to the overall sample size. Because of high variability within and between stations in fecundity parameters, modifying a list of stations would not affect continuity of data but improve estimates of fecundity parameters by increasing sample sizes.

Summary of Specific Recommendations for Limnology Monitoring Program

Based on the analysis above, LADWP makes the following recommendations for future limnology monitoring presented below:

1. Conduct *Artemia* sampling at Stations 2, 4, 6, 7, 8. 9, 10, and 12 monthly from March through November.
2. Continue monitoring a depth profile of dissolved oxygen, ammonium, and chlorophyll *a* at Station 6 monthly from March through November.
3. Conduct CTD (conductivity and temperature), Secchi depth, and 9-m integrated sampling for ammonium and chlorophyll *a* at Stations 2, 4, 6, 7, 8. 9, 10, and 12 from March through November.
4. Conduct CTD (conductivity and temperature), dissolved oxygen, ammonium, chlorophyll *a*, and Secchi depth sampling at Station 6 in February and December.
5. Discontinue *Artemia* instar analysis in the future.
6. Conduct *Artemia* fecundity analysis at Stations 2, 4, 6, 7, 8. 9, 10, and 12 from June through October.
7. Continue all other monitoring not mentioned above e.g. Meteorological.

3.3 Vegetation Status in Lake-Fringing Wetlands

3.3.1 Lake-fringing Wetland Monitoring Methodologies

The SWRCB determined that aerial imagery and subsequent mapping studies performed at five year intervals at Mono Lake would not be sufficient to evaluate rapid shoreline changes that may occur, and would be of limited value for use in adaptive management of ongoing restoration activities (SWRCB 1998). Annual aerial photographs are therefore a requirement of Order 98-05 in order to document waterfowl habitat conditions and provide more complete information to assess shoreline changes at Mono Lake.

Annual aerial photography is conducted at the three waterfowl survey areas - Mono Lake, Bridgeport Reservoir and Crowley Reservoir. Shoreline subareas were established at each waterfowl survey areas for use in evaluating the spatial distribution of waterfowl, and photos were taken of each subarea. At Mono Lake, the 15 shoreline subareas (Figure 3.3-1) followed those established in Jehl (2002), except for minor adjustments made in order to provide the observer with obvious landmarks that are easily seen during aerial waterfowl surveys. Bridgeport Reservoir has three shoreline survey areas (Figure 3.3-2) and Crowley Reservoir seven (Figure 3.3-3). In 2018, still photos of lake-fringing habitats were taken from a helicopter on October 17, by Deborah House, LADWP Watershed Resources Specialist.

3.3.2 Lake-fringing Wetland Photo Compilation

The annual photographs of waterfowl habitats at Mono Lake, Bridgeport Reservoir and Crowley Reservoir were reviewed and compiled. Representative photos from each shoreline subarea were selected. The annual photos, combined with field notes, were used to evaluate and subjectively describe shoreline conditions in 2018.

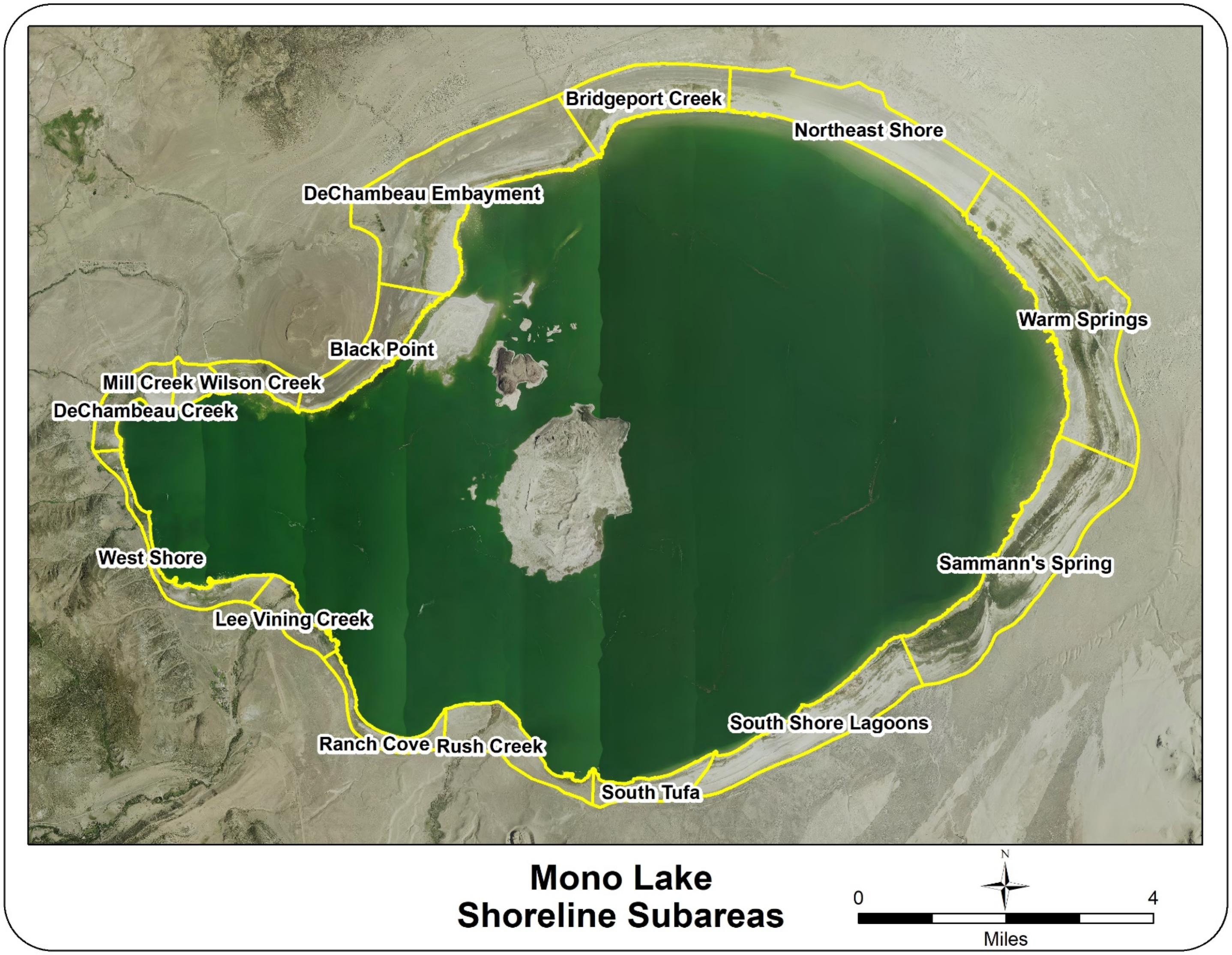

Figure 3.3-1. Mono Lake shoreline subareas

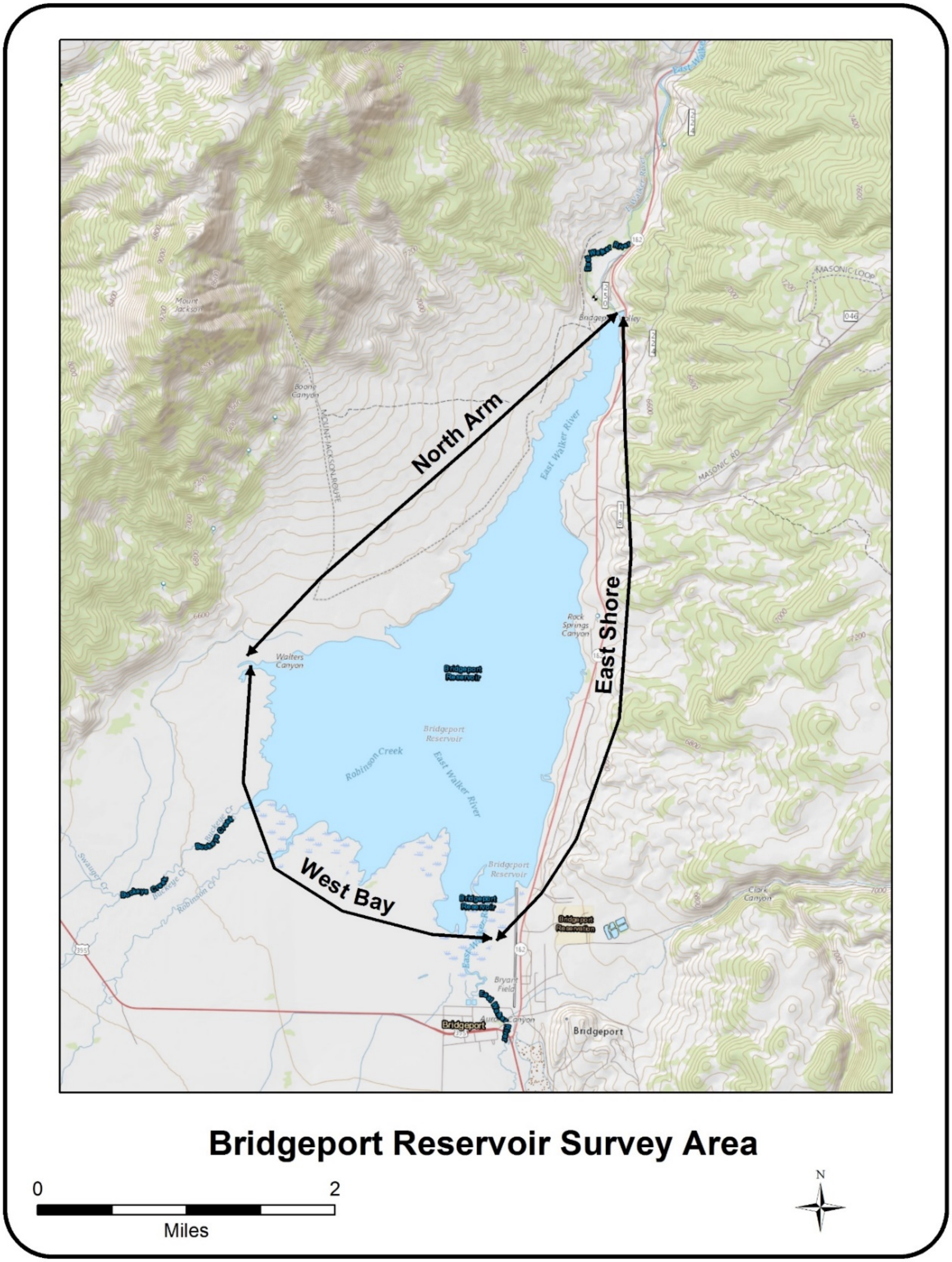

Figure 3.3-2. Bridgeport Reservoir Shoreline Subareas

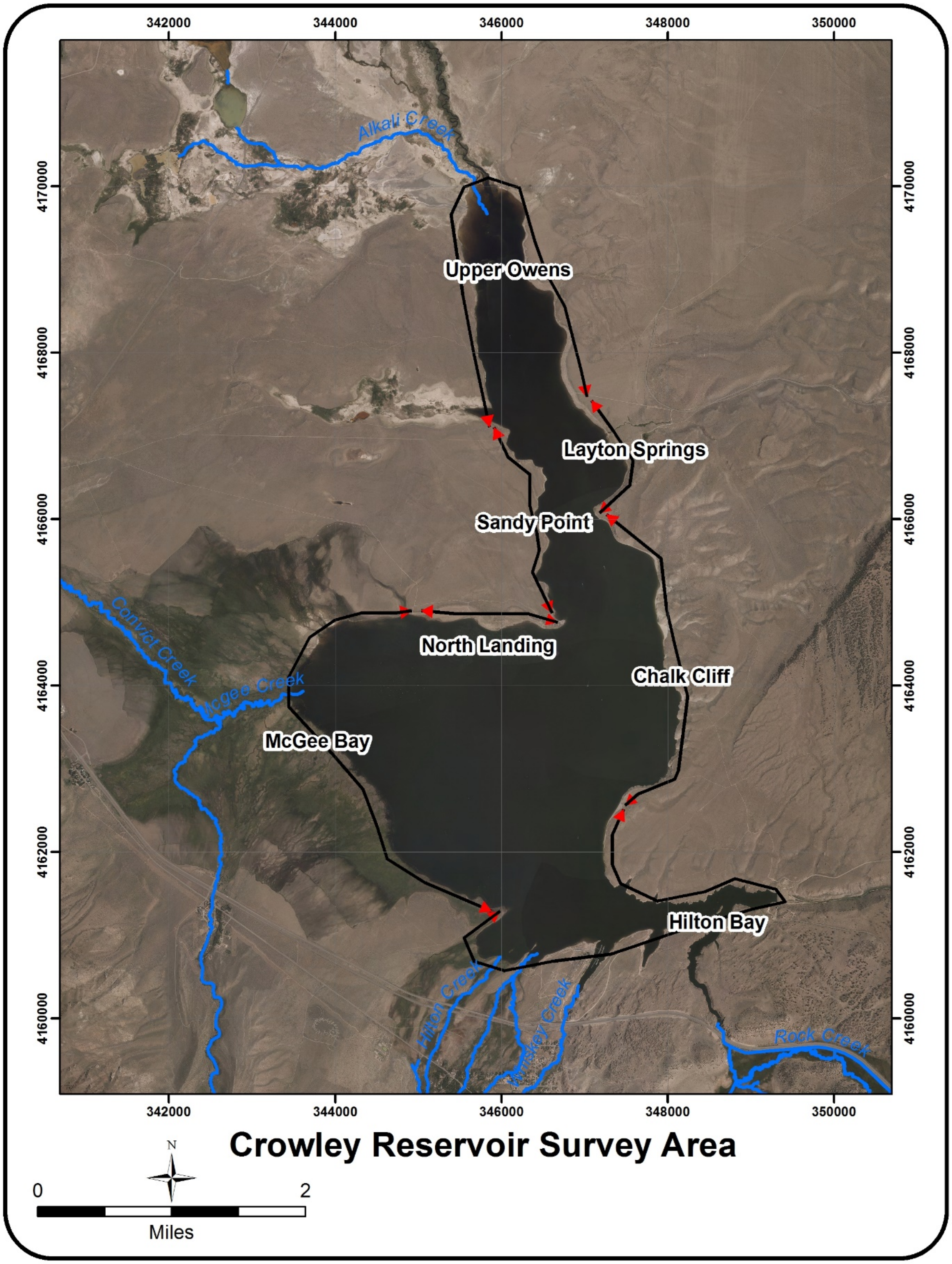

Figure 3.3-3. Crowley Reservoir Shoreline Subareas

3.3.3 Lake-fringing Wetland Survey Area Conditions

Mono Lake Shoreline Subareas

<u>Black Point (BLPO)</u>

Black Point (BLPO) is a volcanic hill on the northwest shore of Mono Lake. The shoreline at BLPO is composed of fairly dry, loose volcanic soils. Alkali and wet meadow vegetation occurs in scattered patches, primarily upgradient of the shoreline and bathymetry studies indicate a gradual offshore slope in this area (LADWP 2018). At lower lake elevations, this shoreline area can be quite dry with notable increases in barren lake bed. At the higher lake levels observed, brackish ponds develop along the shoreline, and the alkaline wet meadow becomes more lush. Although there are no mapped springs in this subarea, the ponds that develop here during periods of higher lake elevation have been used by waterfowl. It is therefore suspected that the ponds are brackish and unmapped springs occur in this area as indicated by LADWP (1987). In 2018, the Black Point shoreline area was dry and brackish ponds were absent (Figure 3.3-4, Figure 3.3-5).

Figure 3.3-4. Black Point Shoreline Area in 2018, Looking Northeast

Figure 3.3-5. Black Point Shoreline Area, Looking Northwest

In Fall of 2018, Brackish Ponds Were Absent in this Area.

Bridgeport Creek (BRCR)

This shoreline area is at the terminus of the Bridgeport Creek (BRCR) drainage, however there is 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 Bridgeport Creek are alkaline wet meadow, with 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. Ria is present at the outlet of each spring, and is likely to be more extensive than mapped. 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 have disappeared. The bathymetry indicates a gradual offshore slope in this area, and there is a shallow shelf just offshore (LADWP 2018). In 2018, only small brackish ponds existed away from the immediate shoreline, with limited to no direct drainage of spring water to the lake (Figure 3.3-6, Figure 3.3-7).

Figure 3.3-6. Bridgeport Creek Shoreline Area, Looking Northwest

In Fall of 2018, brackish ponds were absent.

Figure 3.3-7. Bridgeport Creek Shoreline Area, Looking Northeast

In fall of 2018, brackish ponds were absent.

<u>DeChambeau Creek (DECR)</u>

DeChambeau Creek (DECR) lies along the northwest shore of Mono Lake. The flows in DeChambeau Creek are intermittent, and do not consistently reach the lakeshore. The DECR area has abundant shoreline freshwater resources due to the numerous springs.

The freshwater springs at DeChambeau Creek support wet meadow, mudflats, and riparian scrub. During periods of declining lake levels, wet meadow vegetation has been observed to expand in this area due to the abundance of freshwater spring flow which supports the expansion of wetland vegetation onto newly exposed mudflats. During periods of subsequent increasing lake elevations, the wet meadow vegetation, mudflats, and playa become inundated, leaving little exposed shoreline. The drop in lake elevation after 2011 resulted in erosional headcutting along several of the spring channels, and increased spring channel depths near the lake shore. Increases in barren lake bed area with declining lake elevation are much less substantial along west shore sites such as this than is seen in other areas of the lake. An area of ria is expected to occur at the outflow of each spring, although the extent of ria offshore is expected to vary with spring flow. The bathymetry indicates a gradual offshore slope only near the shore in this area, and a moderately-rapid increase in water depth with increasing water depth from shore quickly follows (LADWP 2018).

In 2018, the increased lake elevation resulted in an inundation of shoreline meadow habitats and mudflats (Figure 3.3-8, Figure 3.3-9). Little to no mudflat habitat existed along much of the length of this shoreline area in the vicinity of the Mono Lake County Park. By July, shoreline meadow vegetation was appearing salt-stressed due to the inundation, and some die-off was occurring. In 2018 beaver activity in the form of a small dam was seen along a spring channel east of the viewing platform.

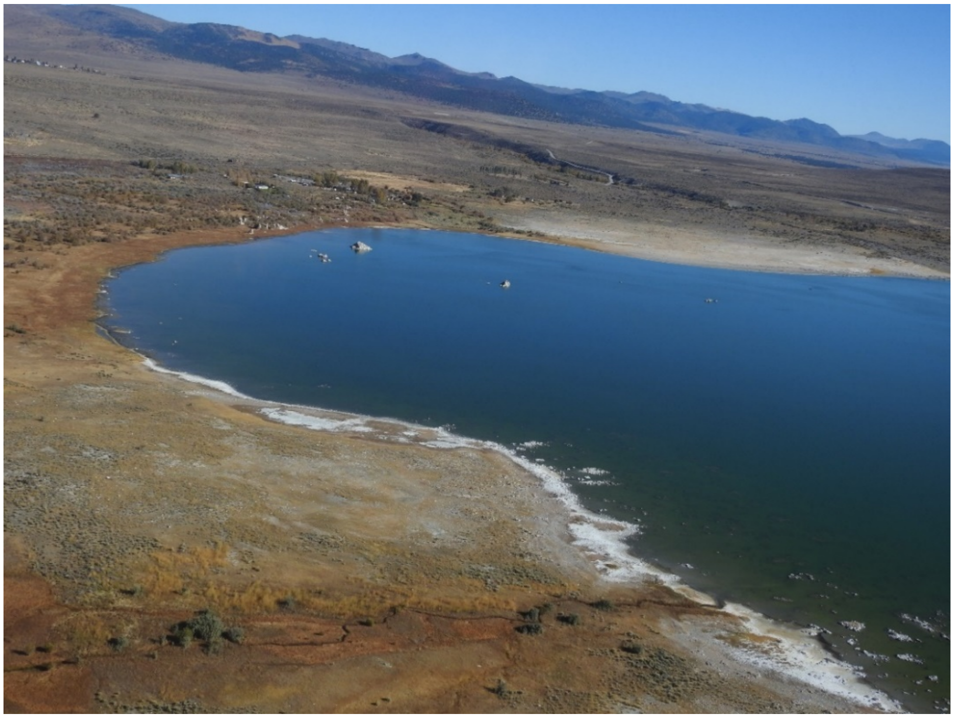

Figure 3.3-8. The DeChambeau Creek Area, Looking Northeast

The increase in lake elevation resulted in inundation of the shoreline vegetation, and near absence of mudflats in fall 2018.

Figure 3.3-9. The DeChambeau Creek Area, Looking Northwest

Danburg Beach is in the foreground.

<u>DeChambeau Embayment (DEEM)</u>

The DeChambeau Embayment (DEEM) area lies just east of the historic DeChambeau Ranch, and the DeChambeau and County Restoration ponds. Historically, Wilson Creek discharged into this area, and the area may have also been influenced by irrigation of the DeChambeau Ranch. Vegetation, dominated by alkali and wet meadow, is primarily confined to the inland portions of the embayment. 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 wetland resources in DeChambeau embayment include alkaline wet meadow, small amounts of marsh, and several small brackish ponds. This portion of the lake is relatively shallow, and experiences rapid increases in the acreage of barren lake bed with decreasing lake levels.

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. Tufa blocks litter the entire subarea, and are most often visible in the southern portion of this area due to the topography. At the higher lake elevations observed, the tufa blocks become partially to completely submerged and the shallow nearshore areas expand. A land bridge with an offshore island had formed by 2015. At more extreme low lake levels, such as those observed in 2016, the geographic extent of the tufa 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 2018, the water level in this area appeared slightly lower than in 2017, as small amounts of barren lakebed were exposed on shore (Figure 3.3-10). Small, isolated brackish ponds were present along the shoreline in 2018 (Figure 3.3-11).

Figure 3.3-10. DeChambeau Creek Embayment, Western Extent

The western extent of this shoreline area, looking southwest. The outflow of Perseverance Spring is in the foreground.

Figure 3.3-11. DeChambeau Creek Embayment, Eastern Extent

The eastern extent of this shoreline area, looking west.

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 1996). 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 wet meadows. 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 from salt water stress. During periods of descending lake elevations, freshwater ponds form behind littoral bars and the entire delta becomes flooded due to extensive channeling. At the 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.

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 lake-ward.

In 2018, water inundated the southern portion of the delta, and multiple braided channels existed nearshore (Figure 3.3-12). In the northern portion of the delta, a sand bar created a fresh water pond near shore, but retained outflow to the lake. The floodplain was well-vegetated with herbaceous wetland vegetation in 2018.

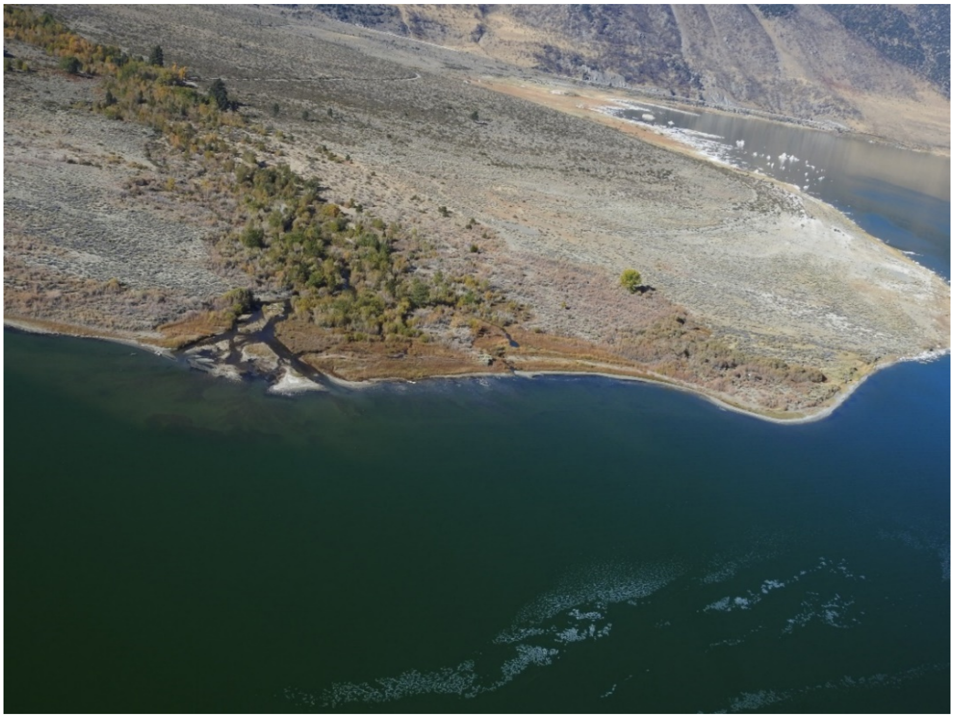

Figure 3.3-12. Lee Vining Creek Delta

In 2018, there was sheet flow of water across the southern portion of the delta, and an onshore freshwater pond in the northern portion

<u>Mill Creek (MICR)</u>

Mill Creek (MICR), Mono Lake's third largest tributary originates in Lundy Canyon. Historically, water diversions for hydropower have affected Mill Creek riparian vegetation.

Freshwater ponds, streams, ria and riparian shrubs are the main waterfowl resources at Mill Creek. Both the flows from Mill Creek and Wilson Creek enter Mill Creek bay in this subarea, thus an area of ria is expected to extend well beyond the mapped boundary. While no springs have been identified in this area, freshwater often enters the lake at several points in the delta due to seepage through the loose volcanic soils. There has also been a tendency for freshwater ponds to form on shore behind littoral bars. By 2012, beaver activity was noted in the delta, and over the years, several dams have been built amongst the willows leading to additional freshwater ponds near shore (Figure 3.3-13).

Previous bathymetry studies have indicated the creek mouth constitutes the only shallow areas in the Mill Creek delta area. Field observations indicate that the high creek flows experienced in the summer of 2017 and 2018 have created a deep channel at the mouth of the creek. In 2018, a large 5-foot tall headcut was present in the western portion of the delta. Additional headcutting was occurring where the main channel enters the lake that was about 2 feet deep in the summer. Upstream of this headcut, the channel formed a deep glide up to 6 feet deep that supported high densities of brine shrimp.

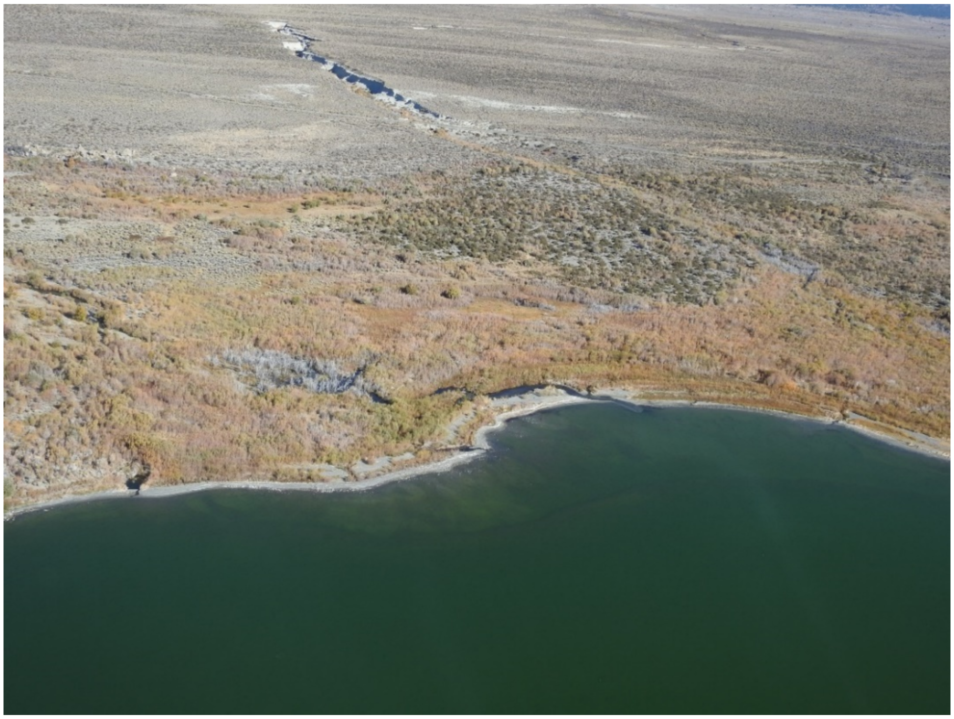

Figure 3.3-13. Mill Creek Delta

Water from Mill Creek entered the lake at multiple points. Relatively large beaver ponds are just left of center in the photo.

Northeast Shore (NESH)

In the Northeast Shore (NESH) area, extensive areas of barren playa dominate at most lake elevations as the groundwater is too saline to support 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 3.3-14). 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. In contrast to the perennial nature of these historic ponds, the ponds observed along the northeast shore have only been observed to persist a single season. Bathymetry studies indicates a very gradual sloped shoreline in this subarea. In 2018, the Northeast Shore area consisted primarily of dry playa, as is typical (Figure 3.3-15).

Figure 3.3-14. An Unnamed Spring Along Northeast Shore

Figure 3.3-15 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. The shoreline 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 2018 Ranch Cove showed no onshore ponds or direct spring input (Figure 3.3-16)

Figure 3.3-16. Ranch Cove Shoreline Area, Looking Southwest.

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 of water diversion of Rush Creek waters for irrigation dating back to the 1860s. Water diversion by LADWP 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. Incision of floodplains drained shallow groundwater tables and left former side channels stranded above the newly incised main stream channel (SWRCB 1994). Under Decision 1631, LADWP was required to develop a stream restoration plan and undertake projects to rehabilitate Rush Creek (LADWP 1996). 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 mapped boundary.

The decline in lake elevation as compared to 2017 resulted in some changes to the Rush Creek delta. In 2018 channel incision of up to 3 feet deep was noted in the delta. A narrow sand bar, not present in 2017 had formed at the mouth of the creek (Figure 3.3-17). Small backwater areas, and a small freshwater pond formed along the eastern bank of the delta. Just upstream of the delta, a long glide was present and attracted many of the waterfowl seen in the Rush Creek delta area.

Figure 3.3-17. Rush Creek Delta

Features of the delta in 2018 include a narrow sandbar at the mouth, several small backwater areas, and a long glide on the mainstem just upstream of the delta

<u>Simons Spring (SASP)</u>

The Simons Spring subarea (SASP) includes the southeastern portion of the lakeshore. Located centrally in the subarea is the Simons Spring faultline, 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 faultline. 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 faultline. 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. When the lake has subsequently decreased, 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. In summer of 2015, headcutting commenced along the westernmost spring channels with the continued decline in lake elevation. This resulted in a drying of the exposed playa in the westernmost part of this subarea. Terminal and Abalos spring at the faultline did not experience headcutting, and mudflats remained, and supported most of the bird activity in this area.

Just east of the Simons Spring faultline, 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 faultline reach shore, and an increasing shallow slope to the east.

In 2018, increases in lake elevation in 2017 and 2018 resulted in the inundation of wetland vegetation that had colonized down-gradient areas during recent lake level declines (Figure 3.3-18). Brine fly were noticeably abundant along the entire length of the

Simons Spring shoreline area, and especially in areas where flooding had inundated shoreline vegetation. By mid-June, lake-fringing vegetation was showing signs of salt-stress and death. A narrow littoral bar formed along much of the length of shore both west and east of the faultine (Figure 3.3-18, Figure 3.3-19), creating onshore ponds through late summer and fall. The wetland vegetation around several of the springs have been heavily grazed by wild horses that have colonized the area.

Figure 3.3-18. Simon's Spring, West of the Faultline

The increase in lake elevation since 2017 resulted in the inundation of wetland vegetation. A narrow littoral bar formed and shoreline ponds were present along much of the length of this area in 2018.

Figure 3.3-19. Simon's Spring, East of the Faultline

Shoreline ponds were present along much of the length of the shoreline in this area in 2018.

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, and 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 faultline. This pond has been ephemeral, and its presence 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 as 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, 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.

In 2018, the brackish lagoon at the western extent of the subarea was flooded, however open water habitat was limited due to vegetation encroachment (Figure 3.3-20). Vegetation encroachment has also impacted waterfowl habitat at Sand Flat Spring and very little open water was present in either the upper or lower ponds (Figure 3.3-21). In the Goose Springs area of South Shore Lagoons, the formation of a narrow littoral bar formed an extensive brackish pond on shore (Figure 3.3-22). Multiple ponds were present in the Goose Springs area including fresh water ponds at the spring heads, a large open freshwater pond downstream of the springheads, and a large brackish pond immediately on shore (Figure 3.3-23).

Figure 3.3-20. South Shore Lagoons, West

The brackish lagoon at the western extent of the subarea was flooded again in 2018; however, the open water was limited due to vegetation encroachment

Figure 3.3-21. Sand Flat Spring

There was very little open water in either the upper or lower pond due to vegetation encroachment.

Figure 3.3-22. Overview of the Goose Springs Area

In 2018, the formation of a narrow littoral bar formed an extensive brackish pond on shore.

Figure 3.3-23. Goose Springs

In 2018, multiple ponds were present including fresh water ponds at the spring heads, a large open freshwater pond downstream of the springheads, and a large brackish pond immediately on shore.

South Tufa (SOTU)

The South Tufa area (SOTU) is the primary visitor access point to the Mono Lake shoreline and includes a large display of tufa towers. The western portion of the survey area, just east of the tufa towers 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 creating 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 pond has persisted from summer through fall. In periods of lower lake elevation the brackish pond was present in summer, but had dried by fall.

During the summer of 2018, the lake elevation was such that there was very little exposed beach on the west half (Figure 3.3-24), and a brackish shoreline pond covering the length of the eastern portion. By fall, the brackish pond in the eastern portion had dried considerably (Figure 3.3-25).

Figure 3.3-24. South Tufa

The western portion of the South Tufa shoreline area had very little exposed beach due to the rise in lake elevation since 2017.

Figure 3.3-25. South Tufa, Eastern Extent

By fall, the extensive brackish lagoon present mid-summer had dried considerably leaving a mostly dry beach.

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 that is 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. 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, 2012 (LADWP 2018). Due in part to their ephemeral nature, vegetation development was not observed in these nearshore brackish ponds. In the summer of 2014, shoreline subsidence of approximately one foot was seen in the vicinity of the north pond. From 2014-2016, several new springs appeared in the expanse of exposed playa. Since 2014, some drying of the wetlands has been noted.

In 2018, extensive flooding and numerous brackish ponds existed in the Warm Springs area (Figure 3.3-26). The north pond (Figure 3.3-27) retains open water and has not experienced vegetation encroachment as has been observed in the fresh water marshes at Mono Lake. In summer, adult brine fly appeared to be very abundant along the shoreline of Warm Springs. Additionally, brine fly were present in high numbers on the water surface of Mono Lake in summer. A wild horse herd of approximately 300 individuals with colts was present in early June at Warm Springs. Well-developed livestock trails now exists in the area, with feeding concentrated near springs.

Figure 3.3-26. Overview of Warm Springs

The outflow channel of Pebble Spring in the foreground feeds a permanent brackish pond nearshore.

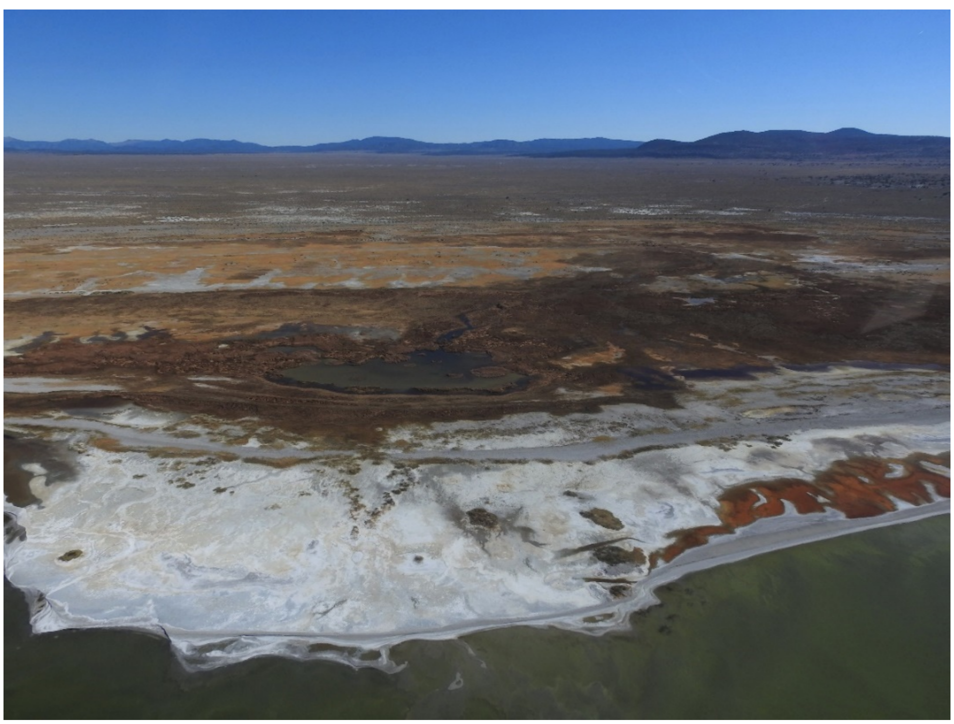

Figure 3.3-27. Warm Springs, North Pond, Looking East.

<u>West Shore (WESH)</u>

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, the majority of the 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 which becomes inundated at higher lake elevations. Significant changes have not been noted in the configuration of this shoreline subarea with lake elevation changes. The lake level increase since 2017 has reduced barren playa in the area (Figure 3.3-28).

Figure 3.3-28. Overview of the West Shore, Looking North/Northwest

<u>Wilson Creek (WICR)</u>

Wilson Creek is along the northwest shore. 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. Due to the shoreline configuration and presence of large tufa towers, this subarea 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 would support 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 2018, mudflats areas were limited in the bay due to the increase in lake elevation since 2017 (Figure 3.3-29). During summer surveys, waterfowl use was concentrated in and around the spring channel on the west side of the bay (Figure 3.3-30).

Figure 3.3-29. Wilson Creek Bay, as Viewed From the Northeast

The outflow of two springs, Black Point Seep and Scoria Tufa enter the bay from the northeast. Limited mudflats were present in 2018.

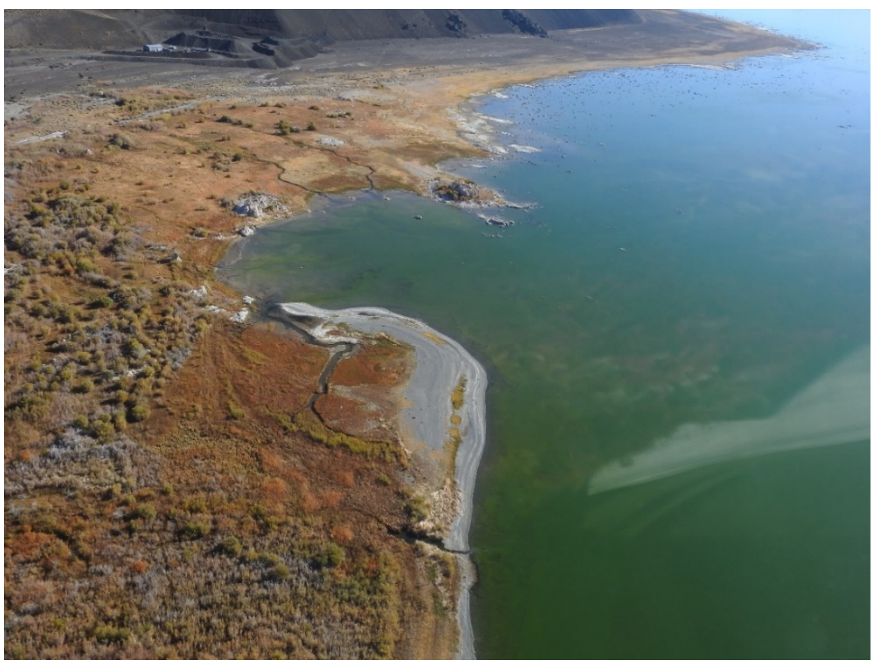

Figure 3.3-30. Wilson Creek Bay, as Viewed From the West

Waterfowl activity in 2018 was concentrated around the spring channel entering the bay from the west.

Bridgeport Reservoir Shoreline Subareas

All three shoreline segments at Bridgeport Reservoir: North Arm, West Bay, and East Shore are shown in Figure 3.3-31. 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 2018, elevated reservoir levels resulted in the creation of shallow feeding areas near the deltas of the East Walker River, Robinson and Buckeye Creeks.

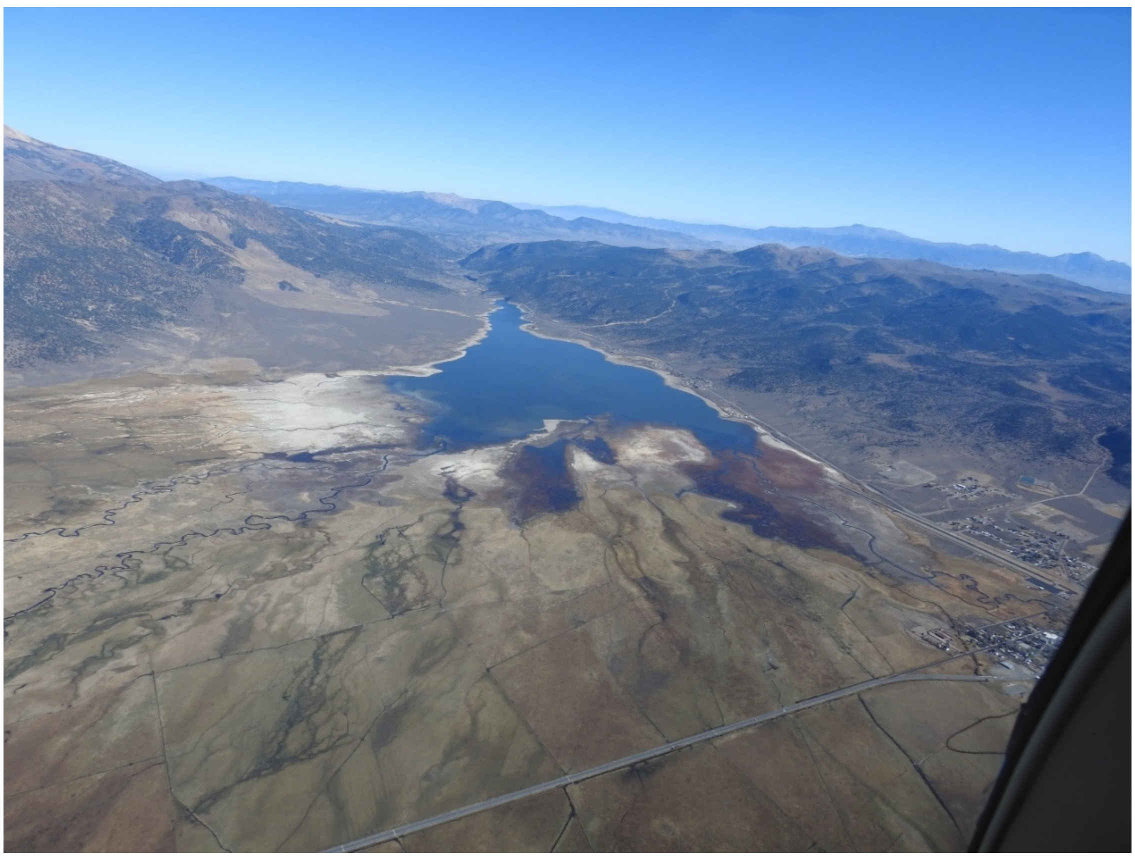

Figure 3.3-31. Bridgeport Reservoir, Looking North

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 3.3-32).

Figure 3.3-32. Chalk Cliffs

Hilton Bay (HIBA)

Hilton Bay includes Big Hilton Bay to the north and Little Hilton Bay to the south (Figure 3.3-33). 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 3.3-33. Hilton Bay

<u>Layton Springs (LASP)</u>

The Layton Springs shoreline area is bordered by upland vegetation and a large area of sandy beach (Figure 3.3-34). Layton Springs provides fresh water input at the southern border of this lakeshore segment.

Figure 3.3-34. Layton Springs

<u>McGee Bay (MCBA)</u>

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 (Figure 3.3-35. Vast mudflats occur along the west shore of Crowley Reservoir, receiving inflow from springs and subsurface flow from up-gradient irrigation (Figure 3.3-36).

Figure 3.3-35. The Outflow Area of McGee and Convict Creeks

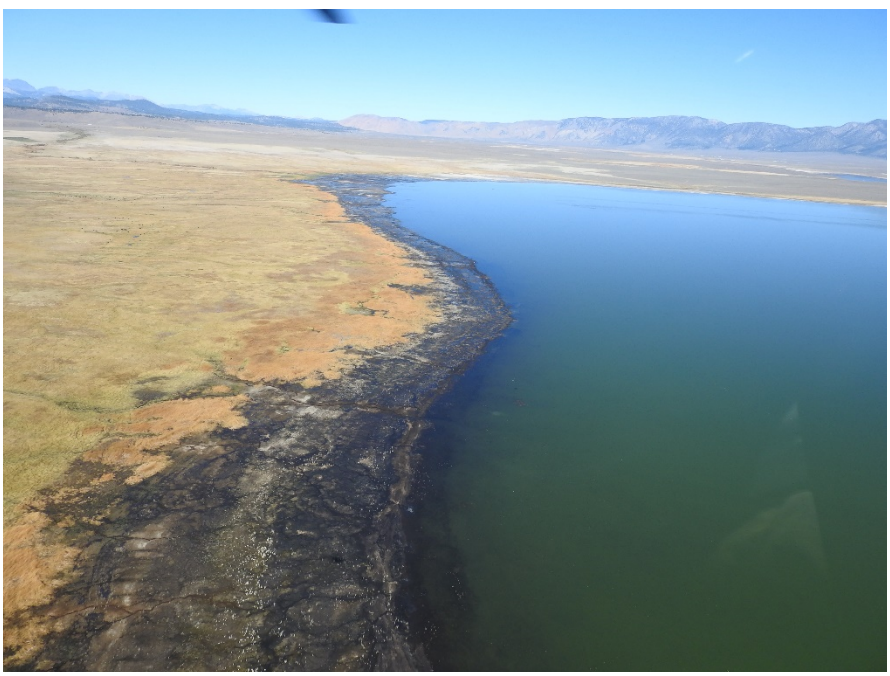

Figure 3.3-36. Mudflat Habitat in the McGee Creek Shoreline Area

North Landing (NOLA)

The North Landing area is influenced by subsurface flows and supports meadow, wet meadow and mudflat habitats (Figure 3.3-37).

Figure 3.3-37. North Landing

<u>Sandy Point (SAPO)</u>

Most of the length of Sandy Point area is bordered by cliffs or upland vegetation (Figure 3.3-38). Small areas of meadow habitat occur in this area, and limited freshwater input occurs at Green Banks Bay.

Figure 3.3-38. Sandy Point

<u>Upper Owens River (UPOW)</u>

The Upper Owens River receives direct flow from the Owens River, the largest source of fresh water to Crowley Reservoir. This subarea includes large areas of exposed mudflats and reservoir bottom (Figure 3.3-39).

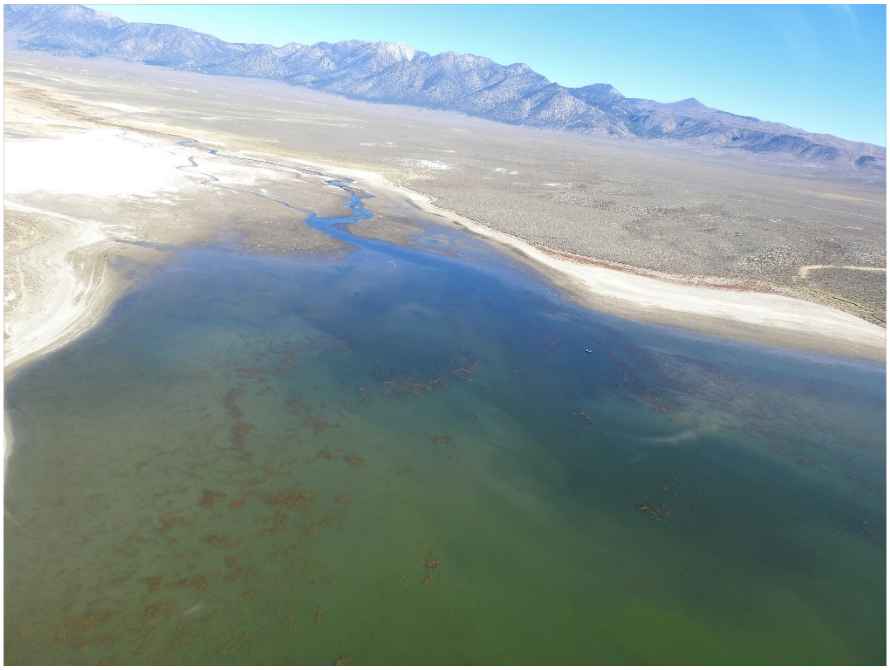

Figure 3.3-39. Upper Owens Delta

3.3.4 Lake-fringing Wetland Condition Discussion

At Mono Lake, the acreage of lake-fringing ponds has been a function of lake elevation (LADWP 2018). During previous periods of lake elevation increase, the acreage of lake-fringing ponds has increased. As a result of the 2012-2016 drought, Mono Lake dropped almost 7 feet. Lake level increases in response to the extreme wet runoff year of 2016-2017 were evident starting in February 2017. The increase in lake elevation observed since early 2017 has risen the lake level from its most recent low of 6376.8 feet in October 2016 to a maximum level of 6381.8 feet in June 2018. As compared to conditions at the end of the drought in 2016, this recent increase in lake level has not resulted in a significant increase in the number of open water lake-fringing ponds. The most significant change the recent increase in lake elevation has had is to restore the connectivity of existing ponds with the water line and spring outflow areas of Mono Lake. In areas of continuous fresh water flow such as DeChambeau Creek and Simon's Spring, wetland vegetation expands lakeward with declining lake levels, and mudflats form. Lake level recovery then often inundates the newly colonized wetland vegetation and submerges mudflats. Inundated wetland vegetation in these areas may support large numbers of alkali flies, and create shallow foraging areas with cover for waterbirds. The increased connectivity of shoreline ponds with the shoreline and spring outflow areas results in improved habitat quality for waterfowl. Although the increased lake elevation observed from 2017-2018 resulted in improved habitat conditions, the highest recent elevation observed of 6381.8 feet did not result in more shoreline ponds, which are an important waterfowl habitat component.

3.4 Waterfowl Population Surveys

Overview of Waterfowl Population Surveys

Waterfowl population surveys have been conducted annually from 2002-2018 at three sites in Mono County including Mono Lake, Bridgeport Reservoir, and Crowley Reservoir (Figure 3.4-1). Situated just east of the town of Lee Vining, Mono Lake is almost centrally located in Mono County. Bridgeport Reservoir is approximately 22 miles northwest of Mono Lake near the town of Bridgeport. Crowley Reservoir is approximately 31 miles southeast of Mono Lake, and 12 miles southeast of the town of Mammoth Lakes.

Waterfowl monitoring at Mono Lake has been more intensive, including summer ground surveys for breeding waterfowl as well as fall aerial surveys. At Bridgeport and Crowley Reservoirs, only fall aerial surveys are conducted. The monitoring of waterfowl populations at Mono Lake was to continue through one complete wet/dry cycle after the targeted lake elevation of 6,392 foot elevation had been reached. At the time of development of the Plan, LADWP anticipated monitoring annually until 2014 (LADWP 1996).

Summer Ground Surveys

Summer ground surveys were conducted only in the Mono Basin along shoreline of Mono Lake and at the DeChambeau and County Pond complexes. Although summer use was believed to be small as compared to the fall migratory population, limited information has been available regarding summer waterfowl populations at Mono Lake. Summer ground surveys of Mono Lake were conducted at sites as specified in the Plan. The Plan provided no specific guidance regarding the objectives of summer monitoring, however Drewien et al. (1996) recommended summer counts to record numbers and species composition of waterfowl and other waterbirds. The implied intent of summer surveys was to fill in gaps in knowledge regarding summer use by waterfowl.

Fall Aerial Surveys

Fall aerial waterfowl surveys were conducted at all three survey areas including Mono Lake and two nearby lakes - Bridgeport Reservoir and Crowley Reservoir. The primary value of Mono Lake to waterbirds is as a migratory stopover, and use by waterfowl is expected to be highest during the fall migratory period. In order to evaluate whether population changes observed at Mono Lake are mirrored at other Eastern Sierra water bodies or are specific to Mono Lake, Bridgeport and Crowley Reservoirs have also been surveyed annually to provide reference data for comparison.

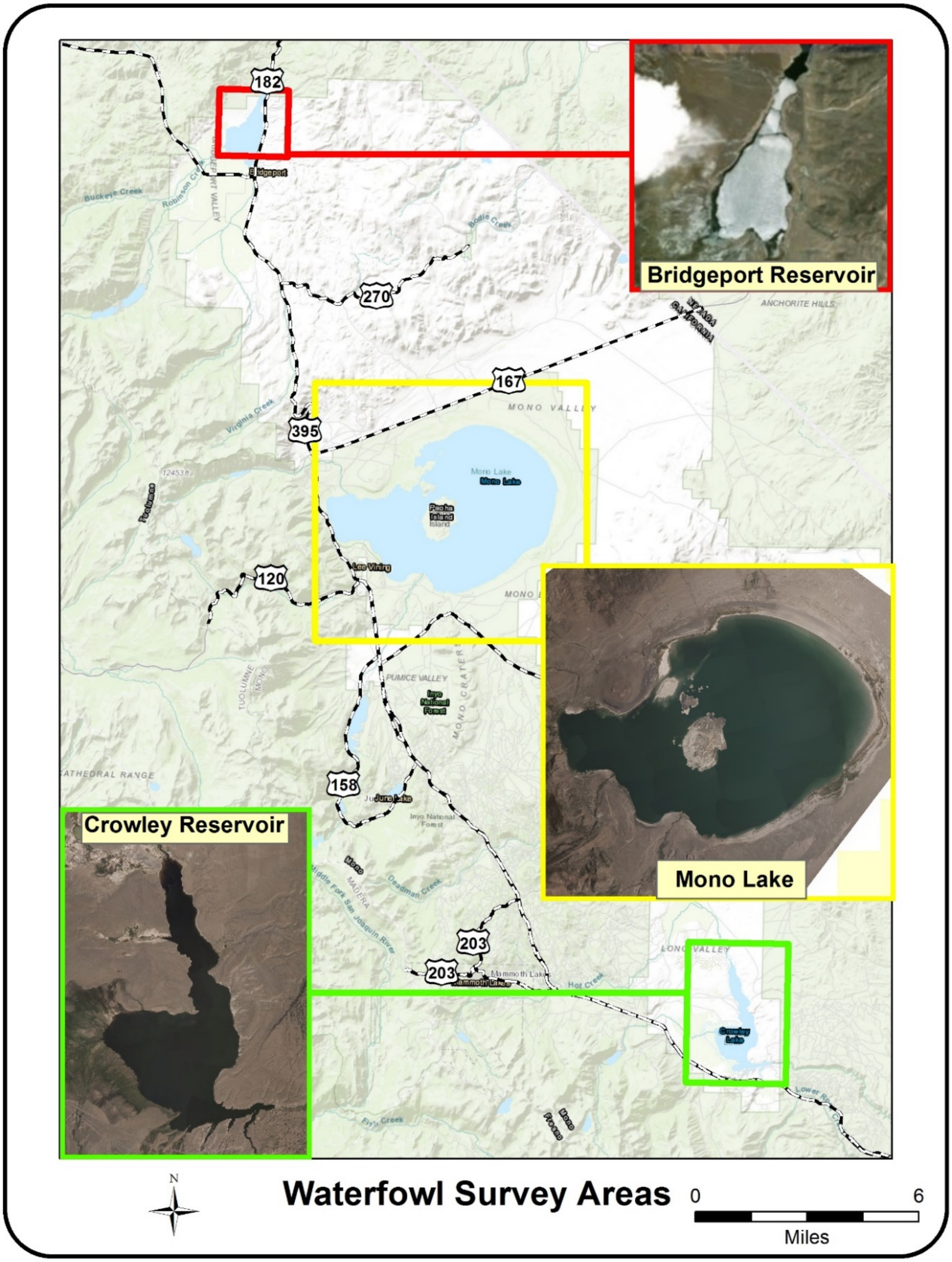

Figure 3.4-1. Overview of Waterfowl Survey Areas

3.4.1 Waterfowl Population Monitoring Methodologies

Summer Ground Surveys

Mono Lake Shoreline Surveys

Each year, from 2002 to 2018, summer ground counts were conducted along the shoreline of Mono Lake to record summer waterfowl use, assess the breeding population, document the number of broods, and record habitat use. The following is a summary of the ground count methodology detailed in Mono Lake Waterfowl Population Monitoring 2016 Annual Report (LADWP 2017). All surveys were conducted by Deborah House.

Nine shoreline subareas and approximately 14 miles of shoreline was surveyed annually (Figure 3.4-2). The following shoreline subareas were surveyed: South Tufa, South Shore Lagoons, Simons Spring, Warm Springs, Wilson Creek, Mill Creek, DeChambeau Creek Delta, lower Rush Creek and Rush Creek Delta, and lower Lee Vining Creek and delta.

Three summer ground-count surveys were conducted annually at each of the shoreline subareas. Surveys were conducted at three-week intervals beginning in early June. The summer ground count survey dates for 2018 are found in (Table 3.4-1). 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 4-5 day period. The order in which the various sites 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 3.4-1. 2018 Summer Waterfowl Survey Number and Dates by Subarea

	2018 Survey Number and Date		
Subarea	**Survey 1**	**Survey 2**	**Survey 3**
RUCR	4-Jun	25-Jun	17-Jul
SOTU	7-Jun	25-Jun	17-Jul
SSLA	4-Jun	27-Jun	16-Jul
DECR	4-Jun	26-Jun	20-Jul
MICR	5-Jun	26-Jun	20-Jul
WICR	8-Jun	26-Jun	20-Jul
LVCR	5-Jun	25-Jun	17-Jul
DEPO	5-Jun	25-Jun	20-Jul
COPO	5-Jun	25-Jun	20-Jul
SASP	6-Jun	28-Jun	18-Jul
WASP	7-Jun	26-Jun	19-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 frequently scanned 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.

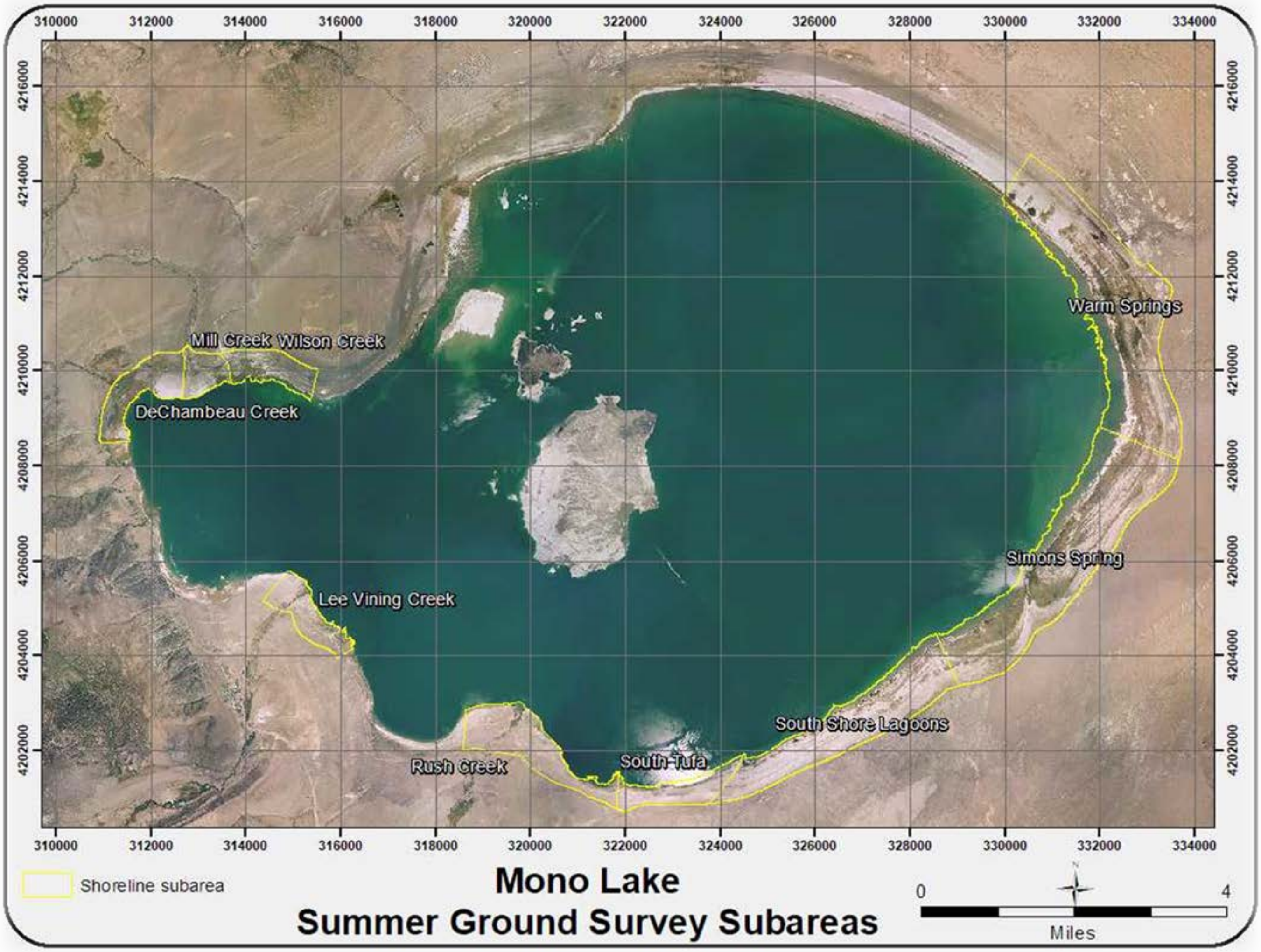

Figure 3.4-2. Summer Ground Count Shoreline Subareas - 2002-2018

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.

Restoration Ponds

From 2002-2018, summer ground counts were also conducted at the DeChambeau and County Pond complexes north of the lake.

The DeChambeau Ponds are a complex of five artificial ponds of varying size (Figure 3.4-3). There are two water sources currently supplying water to the DeChambeau Ponds. Most of the water is from Wilson Creek and is delivered to the DeChambeau ponds via an underground pipe and has averaged 1-2 cfs recently (N. Carle, pers. com.). The underground piping flows water from pond 1 to pond 5. The second source is water from a hot spring adjacent to DEPO_4. The hot spring water was formerly 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.). Since the development of the leak, hot spring water has only been capable of being delivered to DEPO_4.

Figure 3.4-3. DeChambeau Ponds

Aerial Image taken in 2014 shows pond names.

The two County Ponds lie in a natural basin and former lagoon which dried as the lake level dropped below 6,405 feet in the 1950's. The County Pond complex consists of two ponds – County Pond East (COPO_E) and County Pond West (COPO_W) (Figure 3.4-4). Water is delivered to the County Ponds via a pipe from the DeChambeau Ponds. A diverter box exists at the County Ponds to allow some control over water releases to the individual ponds. According to the U.S. Forest Service, County Pond West has been difficult to dry out, thus has been subject to cattail overgrowth. In 2018, the County Ponds were dry throughout summer and most of fall due to a problem with water delivery due to ageing infrastructure.

Figure 3.4-4. County Ponds

Aerial Image taken in 2014 shows pond names.

Fall Aerial Surveys

From 2002-2018, aerial surveys were conducted annually during the fall waterfowl migratory period at three lakes in Mono County: Mono Lake, Bridgeport Reservoir, and Crowley Reservoir. Each year, six surveys were conducted biweekly, with the first survey conducted the first week of September, and the final survey occurring in the mid-November. In all cases, surveys of all three waterbodies were completed in a single flight by 1200 hours (local time) on the day of the survey. Survey dates for 2018 are provided as Table 3.4-2. Each of the three study sites were divided into shoreline and/or open-water segment areas in order to document the spatial distribution of waterfowl.

Table 3.4-2. Fall 2018 Aerial Survey Dates

Survey Number	Date
Survey 1	5-Sep
Survey 2	17-Sep
Survey 3	5-Oct
Survey 4	16-Oct
Survey 5	31-Oct
Survey 6	14-Nov

Aerial surveys were conducted using a high-winged four-passenger aircraft at a speed of approximately 130 kilometers per hour, and at a height of approximately 60 meters above ground. Two observers other than the pilot were present on all six flights and including Deborah House and Chris Allen of LADWP.

Ground verification counts were conducted whenever flight conditions (e.g., lighting, background water color, etc.) did not allow the positive identification of a significant percentage of the waterfowl encountered, or to confirm the species or number of individuals present. During a ground validation count, the total number of waterfowl present in an area was recorded first, followed by a count of the number of individuals of each species present

<u>Mono Lake Shoreline</u>

Fall aerial surveys were conducted at Mono Lake in order to effectively survey both the shoreline and open water areas and be able to complete the surveys in less than two hours. Most dabbling ducks and geese can be found in close proximity to the shoreline. Ruddy Duck, which is one of the two most abundant species, however, can occur in large numbers well offshore. Completing the surveys within this short of a time period limits the chance of double-counting birds due to local movements, and effectively records the total birds present on a single day.

The areas surveyed at Mono Lake were the shoreline and off-shore open water areas of Mono Lake. All areas were surveyed during each flight. The shoreline was divided into 15 shoreline segments. Shoreline segment boundaries for Mono Lake followed those established in Jehl (2002), except for minor adjustments made in order to provide the observer with obvious landmarks that are easily seen during aerial surveys. A sampling grid was established in 2002 to survey open-water areas of Mono Lake during aerial flights. The grid consisted 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 (LADWP 2018)(Figure 3.4-5).

Perimeter surveys were conducted over water while maintaining a distance of approximately 500-800 feet from the shoreline. When conducting aerial surveys, the perimeter flight was conducted first, and in a counterclockwise direction, starting in the Ranch Cove area.

Cross-lake transects were flown immediately afterward, starting with the southernmost transect and working northwards. When conducting cross-lake transect counts, observers sat on opposite sides of the plane and counted Ruddy Ducks, other waterfowl, and phalaropes occurring on the open water. In order to increase detection of waterfowl on the open water, observers sat on opposite sides of the aircraft during cross-lake transect surveys. Although the flight path of the aircraft along the latitudinal transects effectively alternated the observer's hemisphere of observation in a North-South fashion due to the aircraft's opposite headings on successive transects, the one nautical mile spacing between the transects worked in conjunction with the limited detection distance of the waterfowl (<< 0.5 nautical mile) to effectively prevent double-counting of birds on two adjacent transects.

During aerial surveys, the beginning and ending points for each subsection were determined using landscape features, or, when over open water, by using a stopwatch, since the survey aircraft's airspeed was carefully controlled and uniform, and the approximate length of each subsection was known.

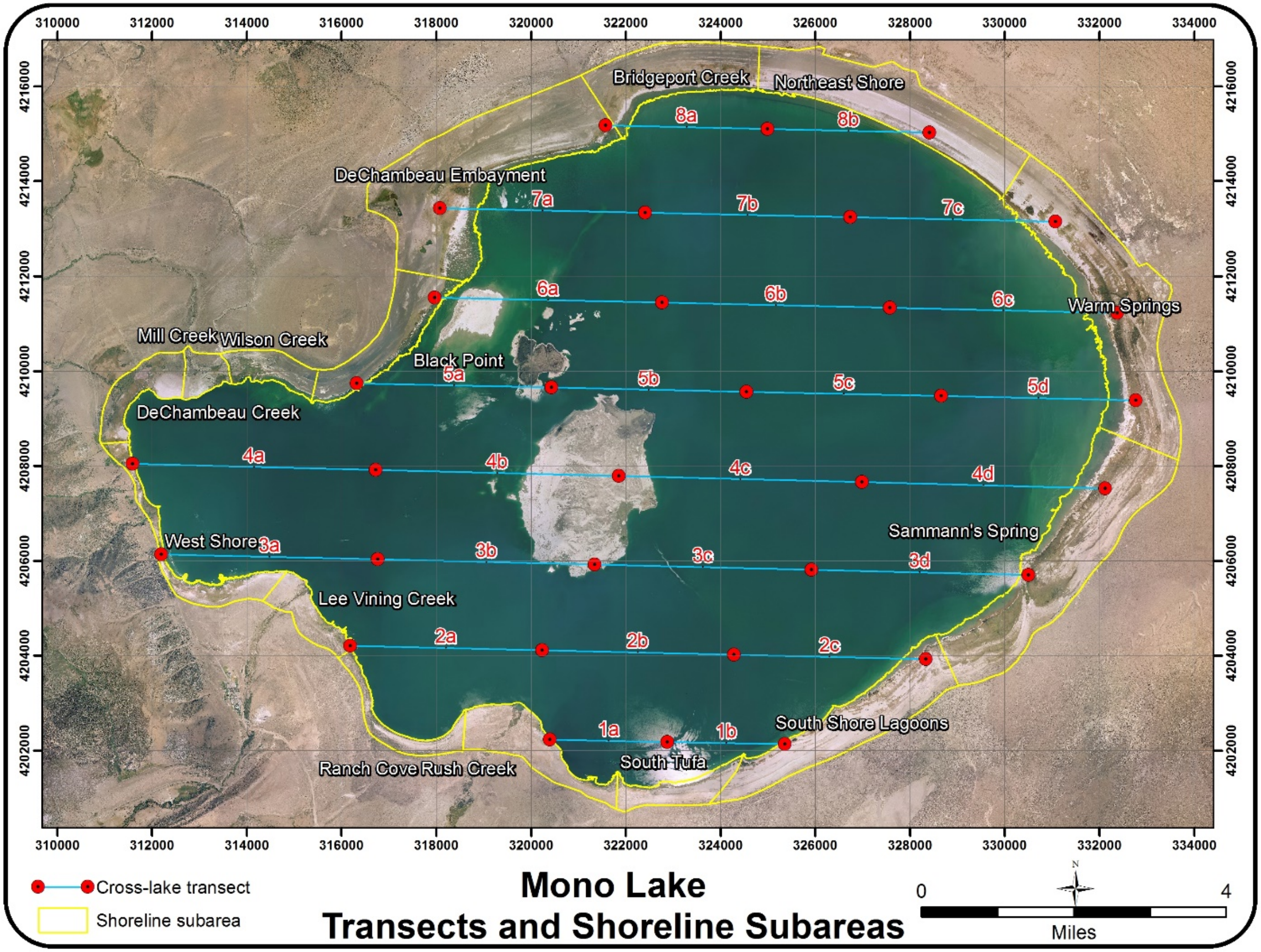

Figure 3.4-5. Mono Lake Shoreline Subareas and Cross-lake Transects

<u>Restoration Ponds</u>

DeChambeau and County Restoration Pond complexes were also surveyed during the aerial flights. Waterfowl observations were recorded by pond.

<u>Bridgeport Reservoir</u>

Bridgeport Reservoir is located in Bridgeport Valley, at an elevation of 6,460 feet. Bridgeport Reservoir is a small reservoir with a surface area of approximately 7.4 square miles and a storage capacity of 42,600 acre-feet. In September 2018, Bridgeport Reservoir held 14,940 acre-feet. The reservoir is rather shallow with a mean depth of 15 feet and a maximum depth of 43 feet (Horne 2003). Bridgeport Reservoir captures flows from Buckeye Creek, Robinson Creek, and the East Walker River to be used for agricultural purposes in Nevada. Irrigated pastures border the south and southwestern portion of the reservoir, while Great Basin scrub is dominant along the north arm and east shore.

Bridgeport Reservoir is eutrophic and experiences summer blooms of blue-green algae. Four colonial forms of cyanobacteria have been found to be common: *Aphanizomenon*, *Anabaena*, *Microcystis*, and *Gloeotrichia* (Horne 2003). In shallow areas near the deltas, submergent aquatic vegetation is abundant.

Although Bridgeport is a small reservoir, ground access to areas where waterfowl concentrate is limited. At Bridgeport Reservoir, all shoreline areas were surveyed during aerial flights, with additional passes over open water areas as needed, based on waterfowl distribution.

Crowley Reservoir

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 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. 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). The storage capacity of Crowley Reservoir is 183,465 acre-feet. In September 2018, Crowley Reservoir held 120,857 acre-feet.

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).

At Crowley Reservoir, all seven shoreline areas were surveyed during each flight with additional passes over open water areas as needed, based on waterfowl distribution. Ground access is good at most locations of Crowley, but limited in the area of highest waterfowl use in the McGee Bay area.

3.4.2 Waterfowl Data Summary and Analysis

Summer Ground Surveys

Summer Waterfowl Community

Summer waterfowl numbers were totaled over the three surveys for all species. Waterfowl species were classified as breeding or nonbreeding based on whether a territorial pair, nest, or brood had been observed over the length of the study. The mean number of detections for 2018 was calculated for breeding waterfowl species and compared to the long-term 2002-2018 value.

Breeding Population Size and Composition

The annual breeding population size was estimated by calculating seasonal species means for all breeding waterfowl species (LADWP 2018). Peak summer waterfowl numbers have been seen during the first survey conducted the first week of June, with the presence of late migrants and males of breeding pairs. Waterfowl numbers decline through the summer to their lowest value by the third survey as males depart after breeding. The third survey has been composed largely of females with their broods of various ages, small numbers of juveniles, and occasional small groups of transient males. Trends in total breeding waterfowl community size were evaluated using simple linear regression (Sigma Plot 13.0).

The breeding waterfowl community composition was evaluated by calculating the 2002-2018 mean plus standard error for each breeding species. The long-term trend in the size of the breeding population was evaluated using simple linear regression.

Variables influencing breeding waterfowl populations
Hydrologic, limnologic, and weather parameters were examined to determine their influence on waterfowl breeding population size.

Habitat use data were summarized by breeding species, using the modeled and mapped vegetation types described in LADWP 2018.

<u>Waterfowl Brood Parameters</u>

In order to provide an index of waterfowl productivity at Mono Lake, the total number of broods was determined. 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 broods difficult.

<u>Breeding Waterfowl Spatial Distribution</u>

The spatial distribution of breeding waterfowl was evaluated by comparing the total broods observed per shoreline subarea with the long-term averages by shoreline subarea.

<u>Restoration Ponds</u>

Waterfowl numbers for each pond were summed. The 2018 waterfowl use and total brood results were compared to the long-term mean from the period 2002-2018.

Fall Surveys

Fall Waterfowl Population Size and Species Composition

Three indices were developed to evaluate the fall use of the survey areas: total waterfowl counts, peak counts, and an estimate of population size or the number of waterfowl using Mono Lake each fall. Total waterfowl counts involved summing waterfowl totals over the six surveys for a year to provide a yearly total. The total waterfowl counts can be interpreted as an index of the number of waterfowl using each survey area, assuming a short turnover time (< the average time between surveys, or ~14 days), and that new individuals are encountered during each survey. This method is likely to overestimate use, but is a simple index in the absence of information regarding stopover periods. The peak counts within any one year was also compiled to represent the maximum number of waterfowl that might be expected on any one day at Mono Lake and to allow for comparison to early waterfowl data. Thirdly, a population estimator was used to estimate the total number of waterfowl using Mono Lake each fall (LADWP 2018).

Species totals per survey area were summarized by survey. The results of 2018 aerial surveys were compared to the long-term 2002-2018 average.

Long-Term Trends and Variables Influencing Fall Waterfowl Populations

Trends in total fall waterfowl numbers of the eight most abundant species were evaluated with linear regression. Waterfowl counts were log-transformed prior to analysis. Linear regression was used to evaluate the response of fall waterfowl populations to environmental conditions. The parameters examined include lake elevation and *Artemia* biomass and fecundity. Lake elevation was used since lake elevation management is the primary means of waterfowl habitat restoration. Lake elevation in September was used as an indicator of current lake shore condition. In addition, the lake elevation in September of the previous year was also tested since changes in lake elevation have been shown to influence lake shore conditions also. *Artemia* was analyzed as an indicator of food resource abundance at Mono Lake. The biomass and *Artemia* fecundity (mean number of cysts) were tested. Two time periods were evaluated: September (peak use by Northern Shoveler) and October (peak Ruddy Duck use). Only significant findings are presented.

Spatial distribution

The number of waterfowl detected during the shoreline perimeter flights and the cross-lake transects were summed. The annual and total mean proportion of waterfowl detected in each of the shoreline subareas was calculated.

Comparison with Reference Data

Surveys of Bridgeport Reservoir and Crowley Reservoir are being conducted to provide a set of reference data with which to evaluate trends observed at Mono Lake. In order to evaluate the relative use of these three areas as a fall waterfowl site, annual mean waterfowl populations from 2003-2018 were compared. The indices of annual peak waterfowl numbers and annual total waterfowl population were evaluated for correlations between the three survey areas. Trends in total numbers of the two key species at Mono Lake, Northern Shoveler and Ruddy Duck, were also examined to determine if correlations existed between the three survey areas.

Restoration Ponds

Waterfowl were summed by species across the three annual surveys. Mean annual waterfowl use was calculated for 2002-2018.

Waterfowl Population Monitoring Program Evaluation

The Periodic Overview Report included an analysis of the waterfowl survey data to determine if the monitoring program could be streamlined, yet provide indices to the response of the waterfowl population to restoration. The 2018 data were incorporated into the analysis of summer breeding waterfowl population data, brood data, and total fall waterfowl counts, and the conclusions and recommendations put forth in the Periodic Overview Report were reevaluated.

The mean waterfowl breeding population size and total broods are useful indices to evaluate long-term trends in the response of breeding waterfowl populations to restoration at Mono Lake. Pearson correlation and linear regression methods were used to evaluate the relationship between brood numbers and breeding waterfowl totals from each of the three summer surveys. If results from any individual survey or surveys are highly predictive of population size or total broods, options to streamline monitoring efforts are evaluated and discussed.

The proposal set forth in the Periodic Overview Report regarding fall monitoring was also reevaluated. The objective of this exercise is to determine if an efficient ground-based monitoring program for fall could be developed that would provide acceptable indices. Aerial surveys are very efficient and complete, however a ground-based monitoring program that could be completed in one day would be more sustainable. The key shoreline waterfowl areas of Mill Creek, Wilson Creek, and Simons Spring were included in the analysis. Pearson correlation was also used to evaluate the relationship between total annual waterfowl, and the number of Northern Shoveler and Ruddy Duck in key fall shoreline areas vs. lakewide values for the time period 2002-2018.

3.4.3 Waterfowl Population Survey Results

3.4.3.1 Summer Ground Counts - Mono Lake Shoreline

Summer waterfowl community

In 2018, 985 waterfowl were tallied over the three summer shoreline surveys (Table 3.4-3) with the highest counts observed at the end of June, and lowest in mid-July, after the departure of breeding males. Gadwall, Mallard and Canada Goose were the most abundant species, and these species as well as Cinnamon Teal and Green-winged Teal were seen with broods along the shoreline. Northern Pintail was only observed on the early June survey, and no evidence of breeding along the shoreline of Mono Lake was observed by this species in 2018. Ruddy Duck was not seen breeding in shoreline habitats, however this species breeds regularly at the Restoration Ponds. A Long-tailed Duck over summered at the mouth of Lee Vining Creek in 2018, and was seen by many observers during its stay.

Table 3.4-3. Summer Ground Count Waterfowl Detections in 2018.

	Survey 1	Survey 2	Survey 3	Total
Species	**June 4-8**	**June 25-28**	**July 16-20**	**Detections**
Canada Goose	**36**	**19**	**97**	**152**
Cinnamon Teal	**5**	**18**	**11**	**34**
Gadwall	**182**	**257**	**98**	**537**
Green-winged Teal	**21**	**15**	**5**	**41**
Mallard	**96**	**67**	**48**	**211**
Northern Pintail	**1**	**0**	**0**	**1**
Northern Shoveler	2	0	0	2
Redhead	0	0	1	1
Ruddy Duck	**1**	**0**	**2**	**3**
Long-tailed Duck	1	1	1	3
Total waterfowl by survey	**345**	**377**	**263**	**985**

Breeding population size and composition

The estimated size of the breeding waterfowl population at Mono Lake in 2018 is 326, or approximately 163 pairs. Breeding was confirmed for Canada Goose, Cinnamon Teal, Gadwall, Green-winged Teal and Mallard. The total 2018 breeding population was comparable to the long-term mean of 773. The populations of each of the seven breeding species was generally within the long-term means, although the Mallard population was slightly above average (Figure 3.4-6).

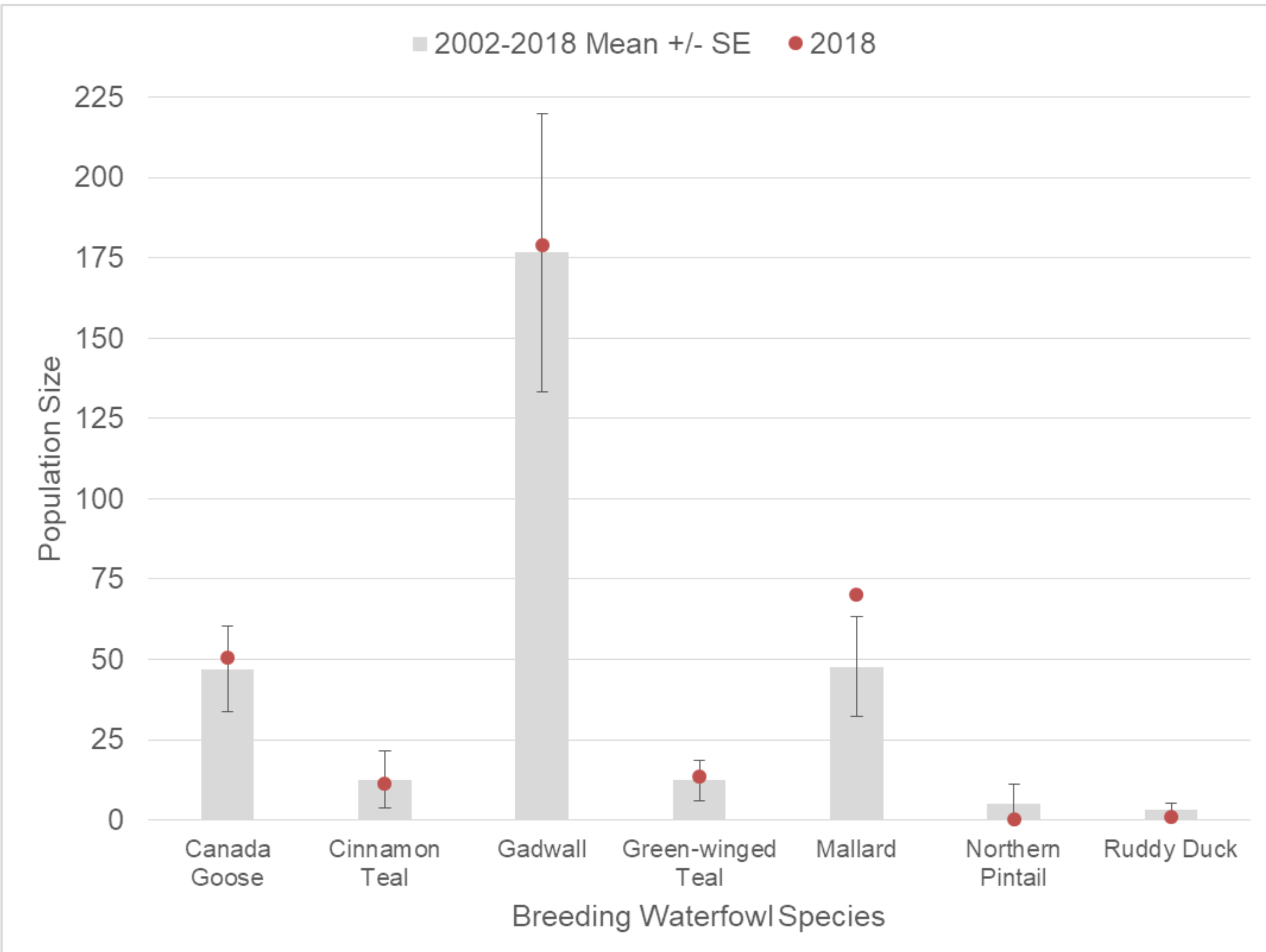

Figure 3.4-6. 2018 Breeding Waterfowl Population Size

The 2002-2018 Long-term Mean is provided as a reference.

At total of 66 waterfowl broods were seen in 2018, including Canada Goose (Table 3.4-4). Gadwall broods were most numerous, comprising almost 60% of all broods (39/66). As many as 16 Canada Goose broods were seen, although this number may be an overestimate as this species is highly mobile, increasing the chance of double-counting family groups.

Table 3.4-4. Waterfowl Broods by Shoreline Area, 2018

Species	DECR	LVCR	MICR	RUCR	SASP	SOTU	SSLA	WASP	WICR	Total broods per species
Canada Goose	2		1		3	3			7	**16**
Cinnamon Teal									1	**1**
Gadwall	1			4			29		5	**39**
Green-winged Teal	1		1						1	**3**
Mallard	1			1	1		4			**7**
Total broods per shoreline area	**5**	**0**	**2**	**5**	**4**	**3**	**33**	**0**	**14**	**66**

The total number of broods along the shoreline of Mono Lake has averaged 45.
6 (range 26-73), exclusive of Canada Goose (Figure 3.4-7). The 50 dabbling duck broods found in 2018 was within the long-term mean.

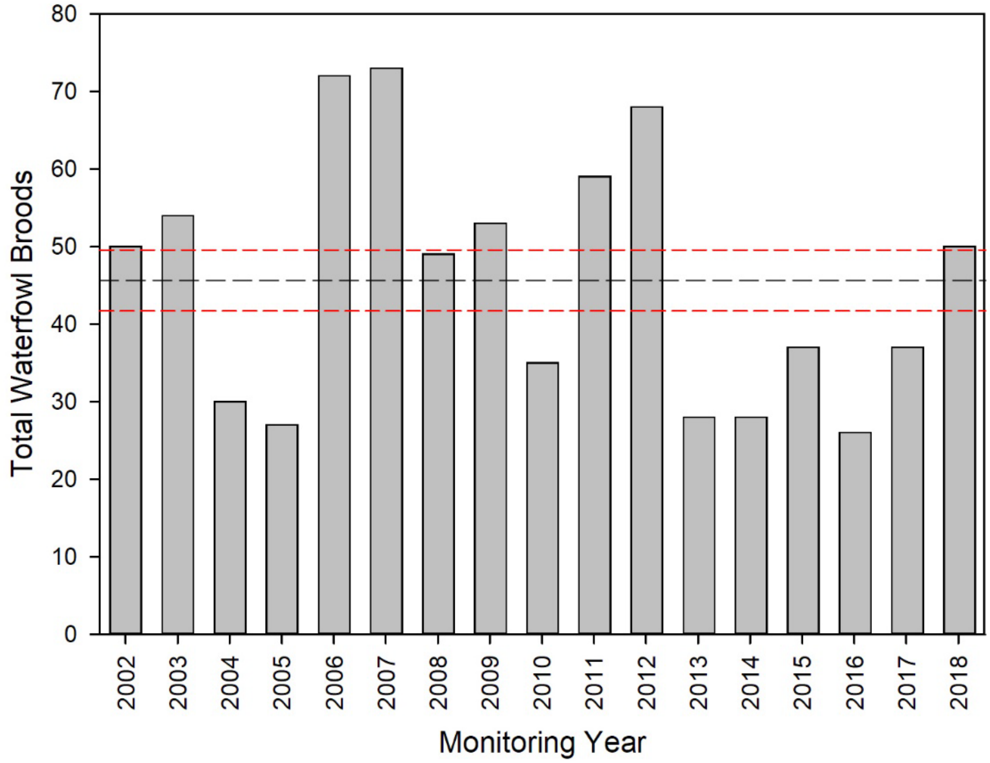

Figure 3.4-7. Total Annual Waterfowl Broods (Excluding Canada Goose) 2002-2018

The dashed reference lines indicate the long-term mean (black) and +/- standard error (SE) (red).

In 2018, breeding activity was highly concentrated around Goose Springs in the South Shore Lagoons shoreline area. Half of all waterfowl broods in 2018 were found in the South Shore Lagoons subarea. The total number of broods in the South Shore Lagoons area in 2018 was significantly above the long-term average for this part of the shoreline (Figure 3.4-8). The use of Wilson Creek, the second most important area for breeding waterfowl, was less than is typical. Wilson Creek supported 21% of the total broods in 2018, however the total number of broods found at Wilson Creek was below the long-term average. Low use was also observed at Mill Creek, Simon's Spring, and Lee Vining Creek. No broods were observed at Lee Vining Creek or Warm Springs.

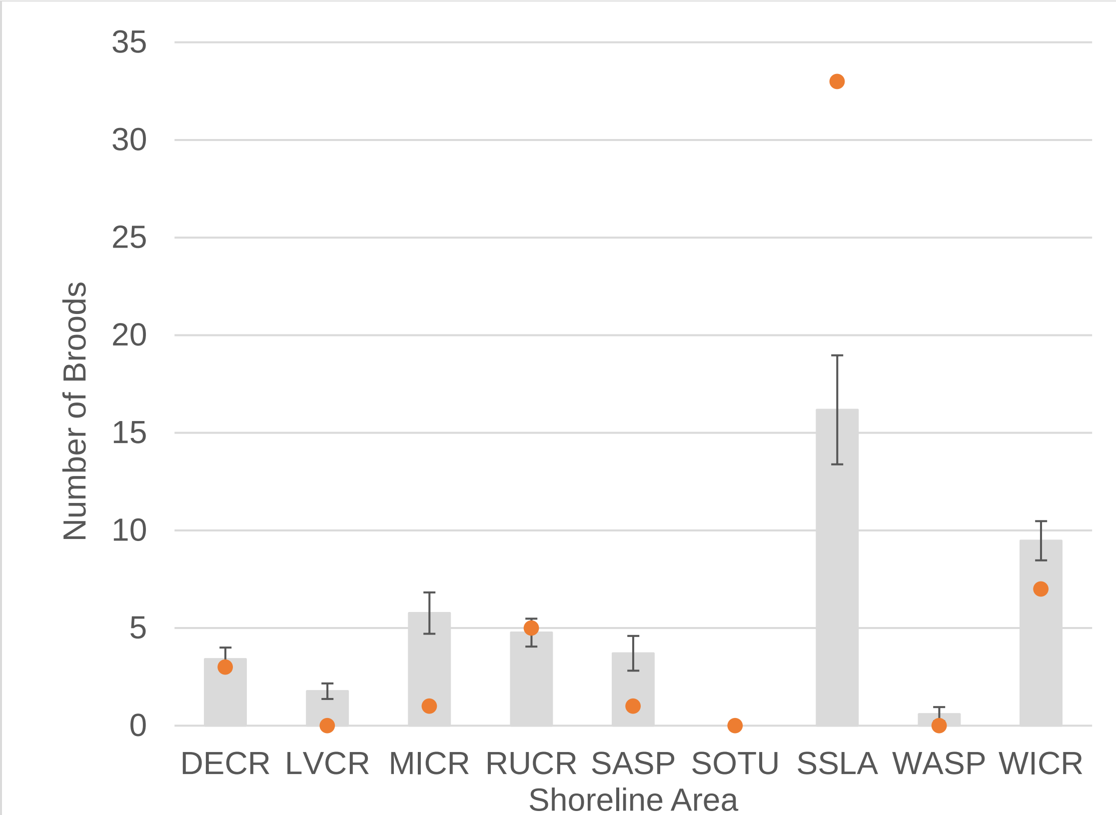

Figure 3.4-8. Brood Numbers at Mono Lake by Shoreline Subarea

The 2018 results are shown relative to the 2002-2018 mean +/- standard error (SE).

Long-term Trends and Variables Influencing Breeding Waterfowl Populations

The largest waterfowl breeding population at Mono Lake (1,666 total detections) was in 2007 when the lake was also at the highest elevation observed of 6,384.5 feet. Breeding populations were at their lowest in 2016 and 2017 when 513 and 434 total breeding waterfowl were observed, respectively. The breeding waterfowl population at Mono Lake has shown a slight downward trend in size since 2002, however this trend is not statistically significant (r^2_{adj} = 0.15, p=0.07) (Figure 3.4-9).

Populations of Gadwall, the most abundant breeding waterfowl species at Mono Lake, have shown a statistically significant decrease in abundance since 2002, although data have been variable (r^2_{adj} = 0.29, p=0.014). Northern Pintail is an uncommon breeding species at Mono Lake, however this species has also shown a decrease in abundance over time (r^2_{adj} = 0.18, p=0.048). Green-winged Teal is the only species whose numbers have shown a significant positive increase since 2002 (r^2_{adj} = 0.21, p=0.036).

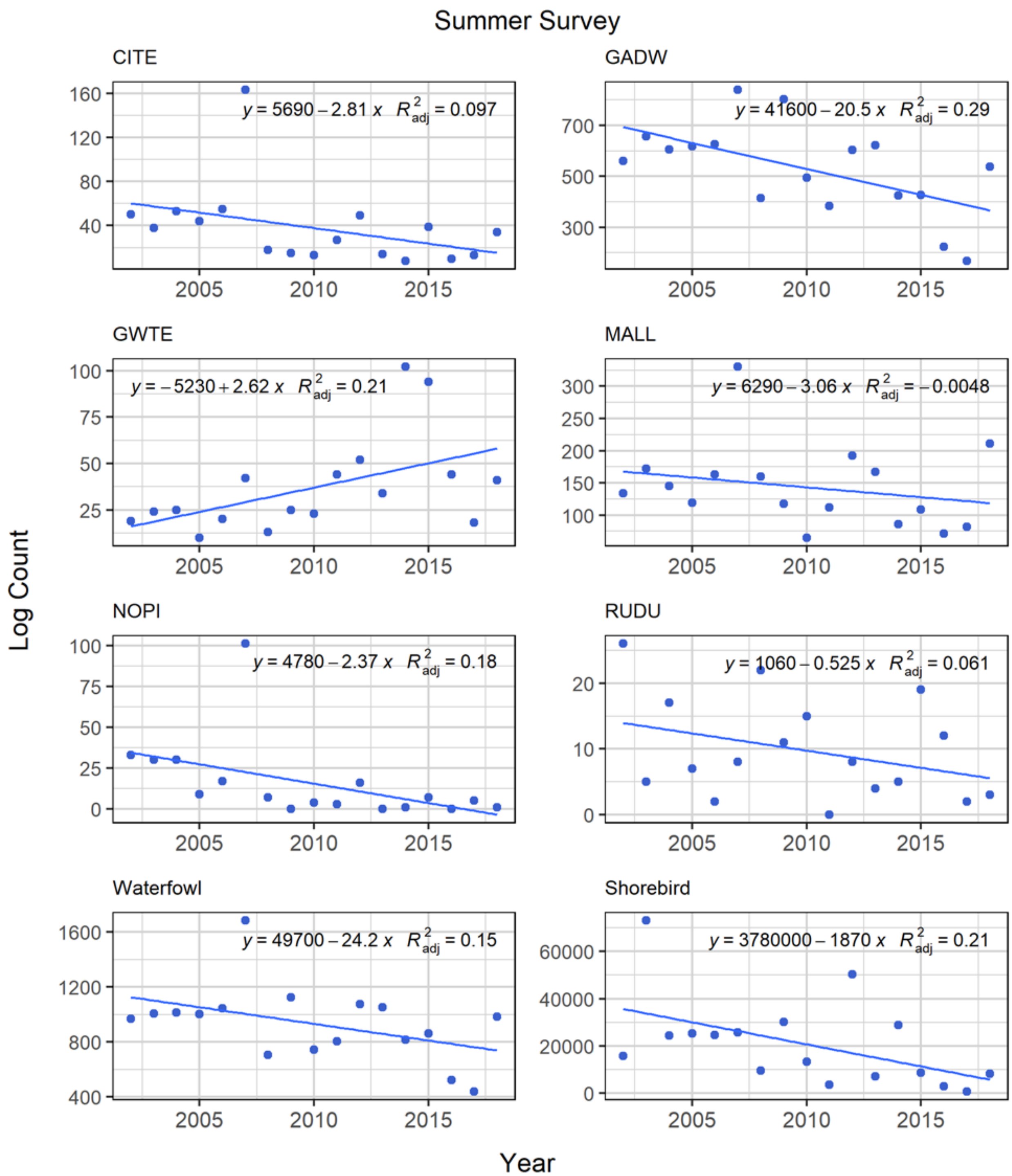

Figure 3.4-9. Trends in Mono Lake Breeding Waterfowl Species, 2002-2018

Table 3.4-5. Best-Fit Model for Total Breeding Waterfowl

Model R^2 adj = 0.57

Variable	*p value*
Lake Elevation (April-June)	0.0033
Monthly *Artemia* biomass – April to June	<0.001

The model that best explains variations in the population size of breeding waterfowl suggests that habitat conditions and food resources in spring are important. The average elevation of Mono Lake April-June of each year combined with the average monthly biomass of *Artemia* has explained 57% of the variability in the waterfowl breeding population size at Mono Lake (Table 3.4-5).

3.4.3.2 Habitat Use

Canada Goose was the only species that regularly used meadow/marsh habitat (Table 3.4-6), and feeding with broods in alkaline wet meadow habitats near or on shore. On-shore water features were the landtype most heavily used by dabbling ducks, with freshwater and brackish ponds receiving the most use. Both freshwater and brackish ponds were used by ducks for feeding and resting. Ria areas were used almost exclusively for feeding. Canada Goose was regularly observed swimming in open water areas offshore, frequently in response to disturbance. Mallard showed the highest proportional use of on shore water features such as fresh and brackish ponds, with relatively less use of ria.

Table 3.4-6. Proportional Habitat use by Breeding Waterfowl Species, 2018

Landtypes		Breeding Waterfowl Species				
Modeled	Mapped	Canada Goose	Cinnamon Teal	Gadwall	Green-winged Teal	Mallard
Meadow Marsh		10%	0%	1%	5%	1%
	Marsh	*0%*	*0%*	*0%*	*0%*	*0%*
	Wet Meadow	*0%*	*0%*	*0%*	*5%*	*1%*
	Alkaline Wet Meadow	*10%*	*0%*	*1%*	*0%*	*0%*
	Dry Meadow/Forb	*0%*	*0%*	*0%*	*0%*	*0%*
Water		13%	65%	71%	32%	86%
	Freshwater Stream	*0%*	*0%*	*5%*	*0%*	*4%*
	Freshwater Pond	*0%*	*6%*	*15%*	*11%*	*15%*
	Brackish Pond	*13%*	*59%*	*51%*	*21%*	*67%*
	Hypersaline Pond	*0%*	*0%*	*0%*	*0%*	*0%*
	Mudflat	*0%*	*0%*	*0%*	*0%*	*0%*
Upland		0%	0%	0%	0%	0%
Ria		34%	35%	25%	53%	10%
Riparian		0%	0%	0%	0%	1%
Barren Lake Bed		0%	0%	0%	0%	2%
Open Water		43%	0%	3%	11%	1%

Summer Ground Counts - Restoration Ponds

In 2018, only 19 waterfowl were recorded at the Restoration Ponds over the three summer survey, this representing the fewest waterfowl observed on summer surveys over the 2002-2018 study period. The pond with the most waterfowl use was DEPO_04 (13 birds over three visits). Only three birds each were observed in COPE_E and DEPO_02. A total of five broods were seen including two Gadwall and Ruddy Duck broods in DEPO_04, and one Ruddy Duck brood in DEPO_02. Both mean waterfowl use and the total number of broods at the Restoration Ponds were below the long-term mean (Figure 3.4-10).

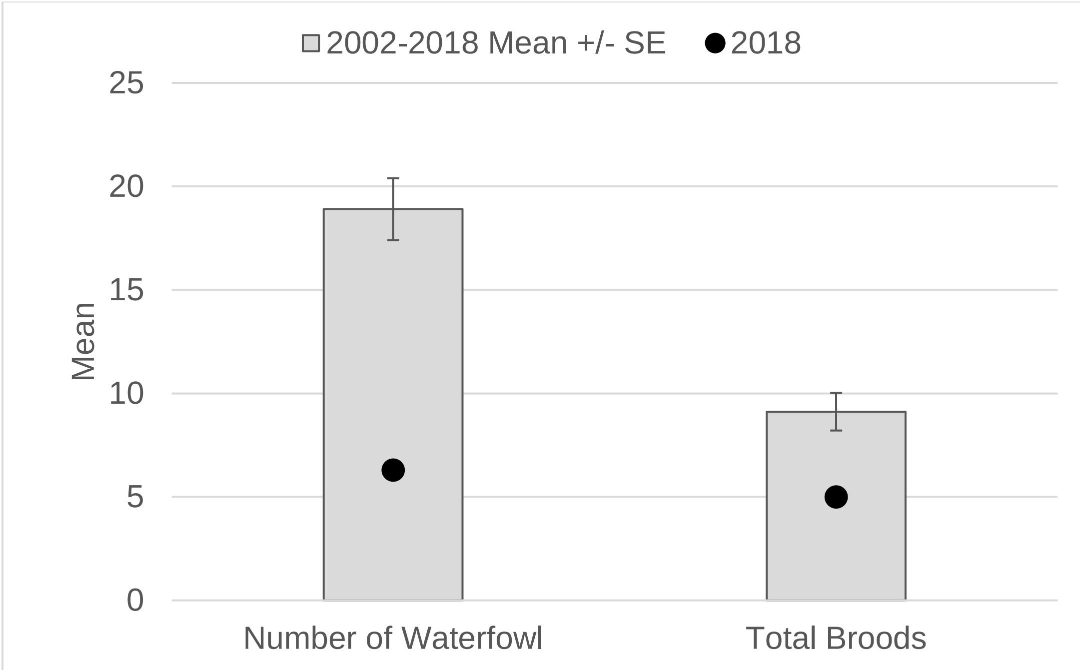

Figure 3.4-10. Total Waterfowl and Broods at the Restoration Ponds, 2018

The 2002-2018 Long-term mean +/- standard error (SE) is shown for reference.

3.4.3.3 Mono Lake Fall Aerial Surveys

Fall Waterfowl Population Size and Species Composition

Fall 2018 waterfowl counts were the lowest since systematic aerial surveys were initiated in 2002. The yearly total number of waterfowl at Mono Lake has averaged 25,434 +/-2,892 SE (Table 3.4-7). The lowest total count of 8,732 was in 2018, and the highest total count of 51,377 in 2004. Peak numbers have averaged 7,941, ranging from a low of 1,826 in 2018 to the highest single day count of 17,844 at the end of September in 2004. The estimated annual fall waterfowl population of Mono Lake, is 9,210 +/- 1,112 SE. Population estimates have ranged from a low of 2,148 in 2018 to a high of 18,590 in 2004.

Table 3.4-7. Mono Lake Yearly Waterfowl Population Indices

Year	Total	Peak	Population Estimate
2002	25,410	7,751	7,571
2003	43,240	9,920	12,868
2004	51,377	10,797	18,590
2005	22,189	7,942	8,263
2006	22,157	6,605	6,943
2007	23,668	9,926	10,080
2008	38,252	13,914	14,017
2009	27,861	7,920	10,906
2010	11,856	3,293	4,760
2011	21,897	5,248	5,635
2012	43,108	17,400	17,400
2013	23,712	8,213	8,557
2014	21,898	8,171	11,075
2015	16,882	8,437	8,654
2016	15,275	4,297	5,644
2017	14,874	3,350	3,460
2018	8,732	1,826	2,148
Mean	**25,434**	**7,941**	**9,210**
Std Err	**2,892**	**947**	**1,112**

Total waterfowl use varies temporally, with numbers highest during the month of September. This early season peak is largely due to the abundance of Northern Shovelers in September. After the end of September, waterfowl numbers at Mono Lake usually decline substantially. Waterfowl numbers in 2018 were well below the long-term mean on all surveys, except mid-November (Figure 3.4-11).

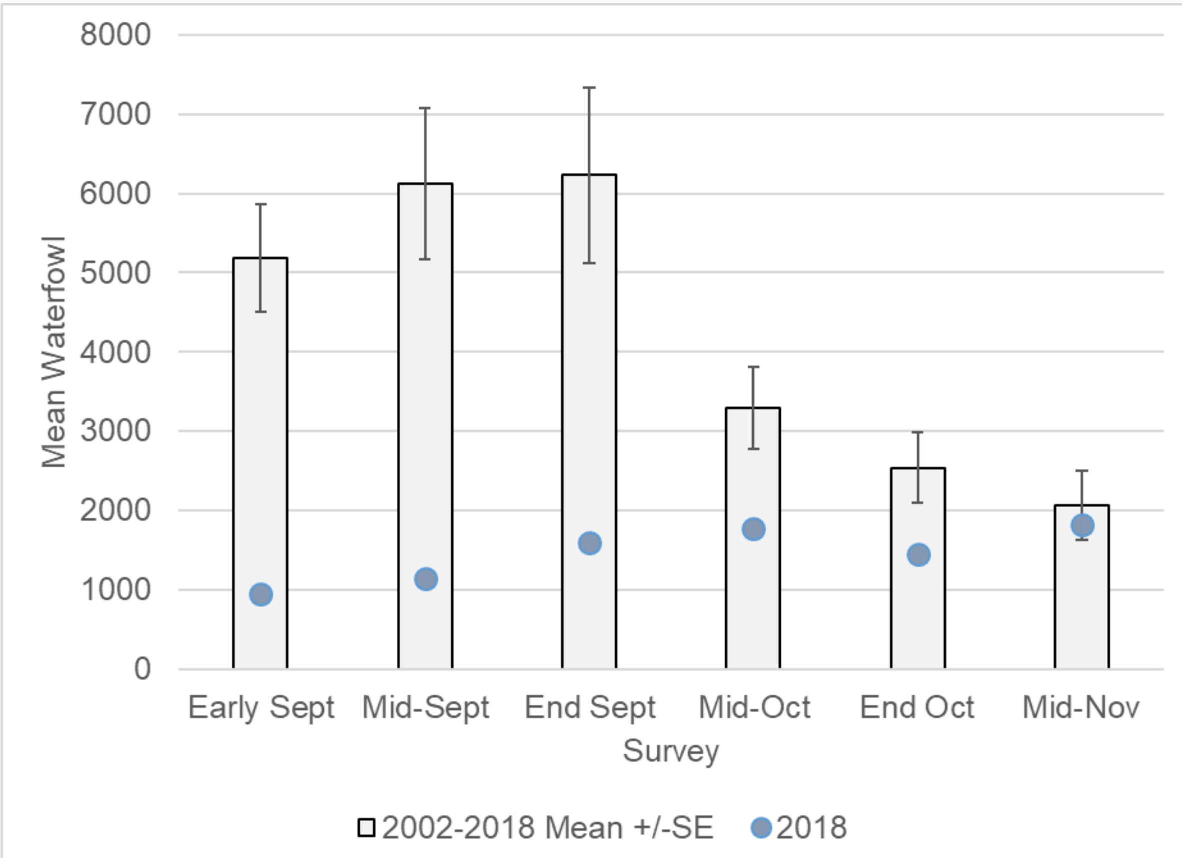

Figure 3.4-11. 2018 Mono Fall Waterfowl Survey Totals and 2002-2018 Means

A total 16 waterfowl species were detected on Mono Lake aerial surveys in fall of 2018 (Table 3.4-8). Northern Shoveler and Ruddy Duck were the most abundant species with Northern Shoveler comprising 31% (2,719/8,732) and Ruddy Duck 39% (3,389/8,732) of all waterfowl in 2018. The most significant decrease in species numbers, contributing to the record low count was observed in the numbers of Northern Shoveler, the most abundant fall migrant at Mono Lake. The total number of Northern Shoveler and Ruddy Duck recorded at Mono Lake in 2018 was well below the long-term 2002-2018 average (Figure 3.4-12), and Northern Shoveler numbers were the lowest recorded over this same time period. Ruddy Duck numbers were the third lowest since 2002. Other species showed reduced numbers as well at Mono Lake in 2018 including Canada Goose, Green-winged Teal and Mallard. Gadwall and Northern Pintail numbers were slightly above the long-term means (Figure 3.4-13).

Table 3.4-8. Species Totals, 2018 Mono Lake Fall Waterfowl Surveys

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Bufflehead	0	0	0	0	2	0	2
Canada Goose	12	18	18	28	48	13	137
Canvasback	0	0	0	0	4	0	4
Cinnamon Teal	10	2	0	0	0	0	12
Common Merganser	0	0	0	0	0	1	1
Gadwall	152	77	27	20	84	40	400
Greater White-fronted Goose	0	0	1	0	0	0	1
Green-winged Teal	54	13	86	0	45	50	248
Lesser Scaup	0	0	4	0	0	0	4
Mallard	33	71	51	43	120	83	401
Northern Pintail	2	27	19	59	15	871	993
Northern Shoveler	455	768	576	472	438	10	2719
Ruddy Duck	96	172	760	983	660	718	3389
Snow Goose	0	0	0	0	1	0	1
Tundra Swan	0	0	0	0	5	0	5
Unidentified Teal	128	0	46	170	30	40	414
Surf Scoter	0	0	0	0	1	0	1
Total	**942**	**1148**	**1588**	**1775**	**1453**	**1826**	**8732**

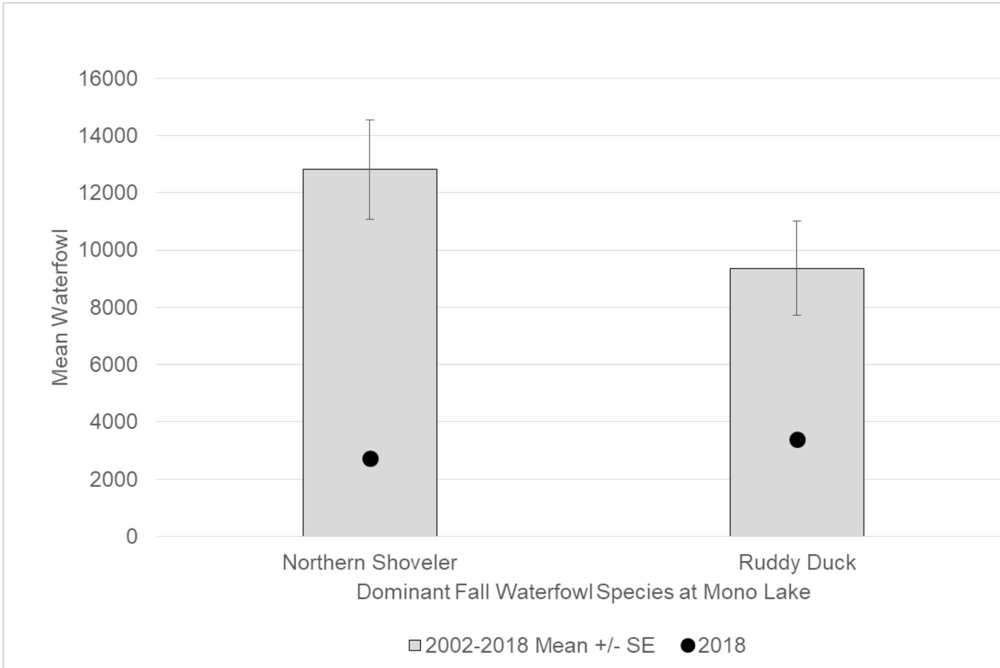

Figure 3.4-12. Total Northern Shoveler and Ruddy Duck, Mono Lake

Two most abundant species during the fall survey species are shown.

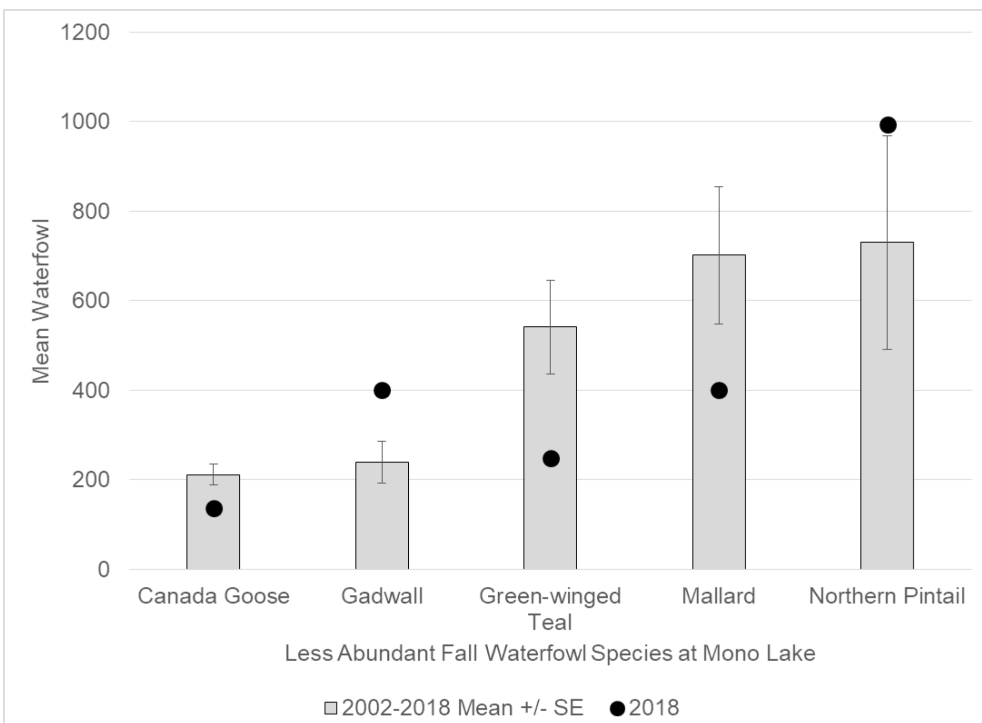

Figure 3.4-13. Mono Lake Waterfowl Species Totals, 2018

Less abundant species during the fall survey are shown.

Long-term Trends and Variables Influencing Fall Waterfowl Populations

There has been a downward trend in total fall waterfowl use at Mono Lake over the 2002-2018 period ($r = -0.594$, $r^2_{adj}=0.31$, $p=0.012$) (Figure 3.4-14). Species showing declining trends have been Cinnamon Teal, Green-winged Teal, and Ruddy Duck (Table 3.4-9). The use by the most abundant species, Northern Shoveler, has been highly variable, with no overall trend $r = -0.256$, $r^2_{adj}=0.003$, $p=0.332$).

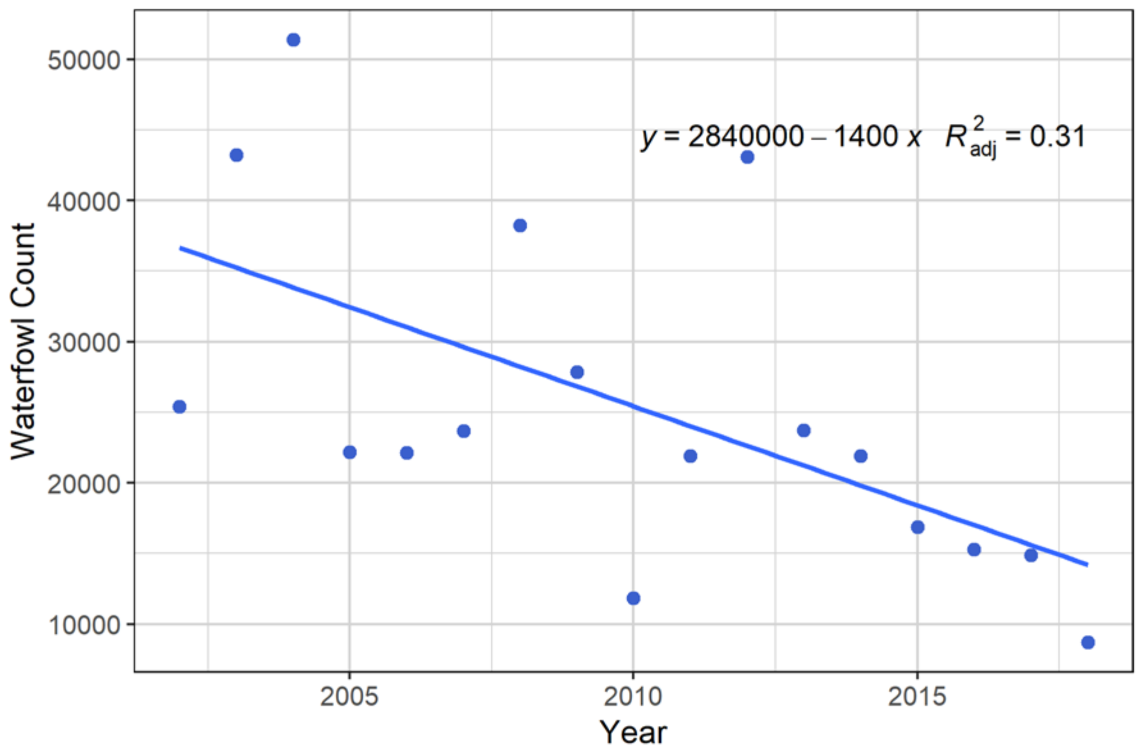

Figure 3.4-14. Total Fall Waterfowl Population Trend, Mono Lake, 2002-2018

Table 3.4-9. Trends in Fall Waterfowl Species at Mono Lake, 2002-2018.

Species	**r**	**p**	**r2**	**r2 adj**
Canada Goose	-0.281	0.274	0.079	0.018
Cinnamon Teal	**-0.550**	**0.022**	**0.302**	**0.256**
Gadwall	-0.086	0.744	0.007	-0.059
Green-winged Teal	**-0.782**	**0.000**	**0.612**	**0.586**
Mallard	-0.029	0.911	0.001	-0.066
Northern Pintail	0.328	0.199	0.107	0.048
Northern Shoveler	-0.256	0.322	0.065	0.003
Ruddy Duck	**-0.683**	**0.002**	**0.467**	**0.432**
Total Waterfowl	**-0.594**	**0.012**	**0.353**	**0.310**

Bold type indicates species with significant declines, 2002-2018.

The response of fall waterfowl populations at Mono Lake to lake elevation and *Artemia* population indices was examined. Unlike use by breeding waterfowl, fall waterfowl use has not been directly correlated with lake elevation either in of the two time periods evaluated – September of the current year, representing conditions at the time of peak use ($r = 0.134$, $r^2_{adj}=0.00$, $p=0.609$) or elevation the previous September ($r=0.449$, $r^2_{adj}=0.149$, $p=0.070$).

Fall waterfowl populations showed some relationship to fall *Artemia* fecundity, but not biomass. Fall waterfowl populations have been weakly correlated with fecundity of *Artemia* fecundity in September ($r = 0.528$, $r^2_{adj}=0.227$, p=0.0.5), explaining 22% of the variation in total fall waterfowl. The relationship to fecundity in October has been stronger, explaining approximately 55% of the variation ($r = 0.767$, $r^2_{adj}=0.548$, $p=0.004$).

3.4.3.4 Bridgeport Reservoir

Fall Waterfowl Population Size and Species Composition

The yearly total number of waterfowl at Bridgeport Reservoir has averaged 33,623 +/- 4,276 SE (Table 3.4-10). The lowest total count of 13,119 occurred in 2018, and the highest total count of 83,186 in 2005. Peak numbers have averaged 10,748, ranging from a low of 2,583 in 2014 to the highest single day count of 23,150 in 2005. The estimated annual fall waterfowl population of Bridgeport Reservoir is 11,392 +/- 1,421 SE. Population estimates have ranged from a low of 2,691 in 2014 to a high of 23,150 in 2005.

Table 3.4-10. Bridgeport Reservoir Yearly Waterfowl Population Indices

Year	Total	Peak	Population Estimate
2003	58,821	20,941	22,922
2004	30,547	11,860	13,378
2005	83,186	23,150	23,150
2006	43,705	15,238	15,238
2007	24,632	11,957	12,910
2008	17,184	5,486	5,486
2009	33,226	11,270	11,270
2010	35,828	8,140	9,768
2011	35,865	9,770	10,847
2012	33,328	15,582	14,639
2013	18,657	7,430	8,842
2014	13,119	2,583	2,691
2015	25,817	5,434	5,434
2016	28,279	7,993	9,736
2017	31,474	6,709	7,534
2018	24,307	8,427	8,427
Mean	**33,623**	**10,748**	**11,392**
Std Err	**4,276**	**1,406**	**1,421**

A total of 15 waterfowl species were detected on Bridgeport Reservoir aerial surveys in fall of 2018 (Table 3.4-11). The most abundant species at Bridgeport Reservoir in 2018 were Gadwall, Northern Shoveler, Northern Pintail, Green-winged Teal, and Ruddy Duck. Gadwall total were slightly below the long-term mean, while more significant declines were observed for Mallard, Northern Pintail and Northern Shoveler. Ruddy Duck numbers were above the mean (Figure 3.4-15).

Table 3.4-11. Species Totals, 2018 Bridgeport Reservoir Fall Waterfowl Surveys

Species	Early Sept	Mid-Sept	End Sept	Mid-Oct	End Oct	Mid-Nov	Species Totals
Canada Goose	80	200	0	0	0	0	280
Tundra Swan	0	0	0	0	0	6	6
Gadwall	1404	2406	902	140	130	30	5012
Mallard	0	505	752	131	185	208	1781
Cinnamon Teal	100	0	0	0	0	0	100
Northern Shoveler	3000	1510	400	0	70	8	4988
Northern Pintail	8	308	1210	831	50	80	2487
Green-winged Teal	502	1053	1162	412	280	255	3664
Unidentified Teal	80	2303	13	45	0	100	2541
Canvasback	0	0	0	10	0	0	10
Redhead	0	0	60	2	0	0	62
Ring-necked Duck	0	0	0	25	0	0	25
Bufflehead	0	2	0	63	146	87	298
Common Goldeneye	0	0	0	0	6	42	48
Common Merganser	93	39	22	8	22	5	189
Ruddy Duck	0	101	970	500	1050	195	2816
Total	5267	8427	5491	2167	1939	1016	24307

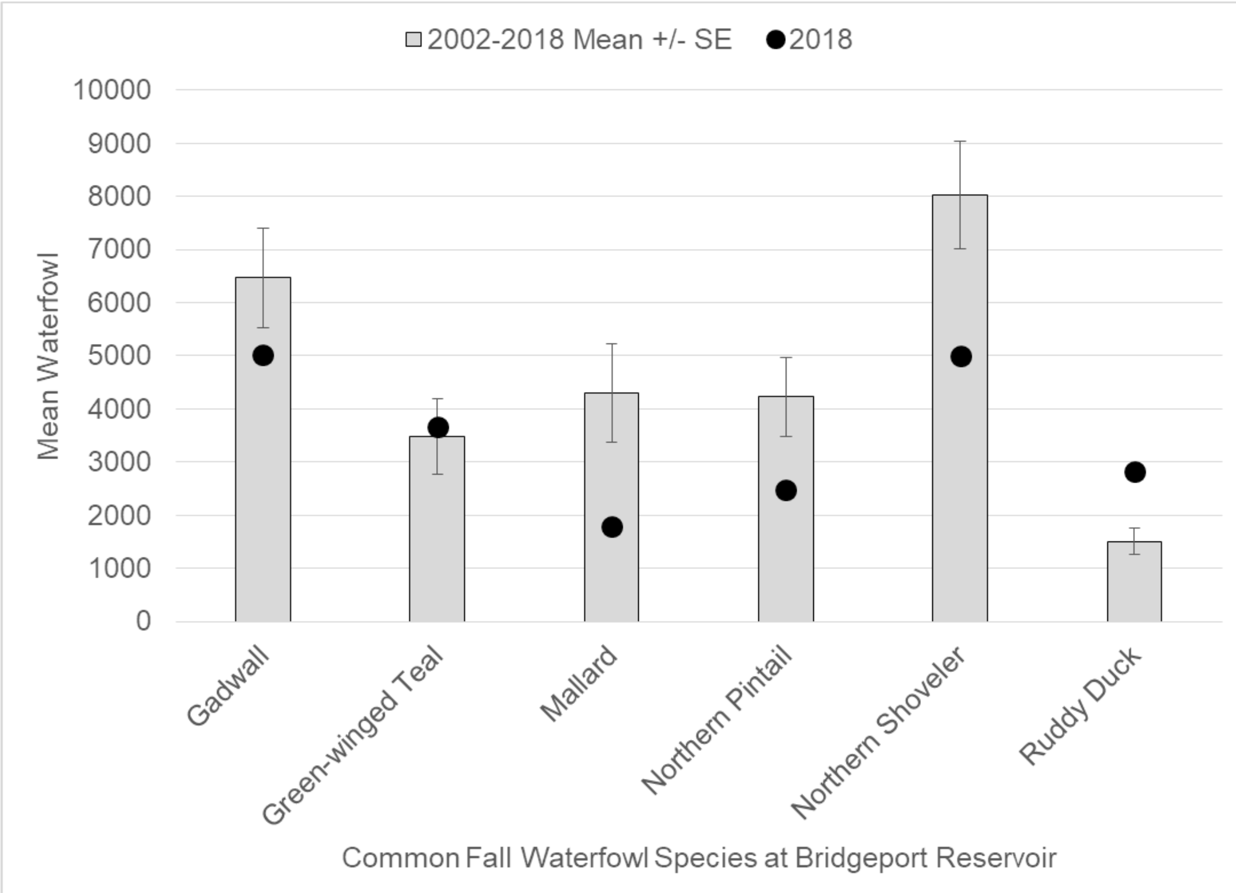

Figure 3.4-15. Bridgeport Reservoir Waterfowl Species Totals, 2018

Long-term 2003-2018 mean is provided as reference.

Spatial distribution

Of the three subareas at Bridgeport Reservoir, waterfowl use was always highest in the West Bay (Table 3.4-12). Waterfowl are found throughout the West Bay and among the several deltas and inlets created where Buckeye Creek, Robinson Creek, and the East Walker River enter the West Bay. Geese are often found out on the meadows in this area away from the water's edge. Waterfowl use in the East shore subarea occurs primarily in the southern half of this segment area, in proximity to inflow from the East Walker River and shallow water feeding areas and mudflats. In the North Arm, waterfowl tend to be few in number and scattered.

Table 3.4-12. Bridgeport Reservoir, Spatial Distribution by Survey, 2018

Survey	EASH	NOAR	WEBA
Early September	84	43	5140
Mid-September	13	82	8332
End of September	467	114	4910
Mid-October	142	17	2008
End of October	502	32	1405
Mid-November	129	46	841
Total waterfowl by shoreline segment	**1337**	**334**	**22636**

3.4.3.5 Crowley Reservoir

Fall Waterfowl Population Size and Species Composition

The yearly total number of waterfowl at Crowley Reservoir has averaged 47,676 +/- 4,906 SE (Table 3.4-13). The lowest total count of 25,474 occurred in 2006, and the highest total count of 82,006 in 2014. Peak numbers have averaged 11,958, ranging from a low of 3,791 in 2007 to the highest single day count of 18,219 in 2005. The estimated annual fall waterfowl population of Crowley Reservoir is 12,900 +/- 1,158 SE. Population estimates have ranged from a low of 6,035 in 2008 to a high of 20,021 in 2014.

Table 3.4-13. Crowley Reservoir Yearly Waterfowl Population Indices

Year	Total	Peak	Population Estimate
2003	74,107	15,555	19,058
2004	65,581	15,002	16,171
2005	57,449	18,219	18,219
2006	25,474	7,878	8,139
2007	17,955	3,791	6,099
2008	29,442	6,035	6,035
2009	36,441	11,695	12,268
2010	47,558	9,802	9,802
2011	29,670	11,290	11,290
2012	33,463	10,464	10,745
2013	62,362	16,089	16,089
2014	82,006	17,657	20,021
2015	65,133	16,117	17,134
2016	64,986	13,204	16,024
2017	33,341	7,819	8,596
2018	37,849	10,723	10,723
Mean	**47,676**	**11,958**	**12,900**
Std Err	**4,906**	**1,069**	**1,158**

A total of 18 waterfowl species were detected on Crowley Reservoir aerial surveys in fall of 2018 (Table 3.4-14). The most abundant species at Crowley Reservoir in 2018 were Gadwall, Mallard, Northern Shoveler, Northern Pintail, Green-winged Teal, and Ruddy Duck. Green-winged Teal, Northern Pintail, Northern Shoveler, and Ruddy Duck totals were only slightly below the long-term mean. (Figure 3.4-16).

Table 3.4-14. Species Totals, 2018 Crowley Reservoir Fall Waterfowl Survey

Species	**Early Sept**	**Mid-Sept**	**End Sept**	**Mid-Oct**	**End Oct**	**Mid-Nov**	**Species Totals**
Snow Goose	0	0	0	0	0	4	4
Canada Goose	210	149	230	240	136	6	971
Tundra Swan	0	0	0	0	6	25	31
Gadwall	865	982	1528	1258	258	60	4951
American Wigeon	12	30	25	22	10	92	191
Mallard	460	465	2219	2039	1050	745	6978
Cinnamon Teal	135	0	0	0	0	0	135
Northern Shoveler	2080	1785	1040	560	114	69	5648
Northern Pintail	125	775	1441	2150	295	237	5023
Green-winged Teal	610	730	680	1300	463	315	4098
Unidentified Teal	545	20	0	30	100	0	695
Canvasback	0	0	0	28	46	76	150
Redhead	0	0	58	28	40	2	128
Ring-necked Duck	0	0	8	2	36	73	119
Lesser Scaup	0	0	8	30	108	40	186
Surf Scoter	0	0	0	1	0	0	1
Bufflehead	0	0	12	146	485	277	920
Common Merganser	0	0	0	3	3	4	10
Ruddy Duck	16	121	2004	2886	1953	630	7610
Total	5058	5057	9253	10723	5103	2655	37849

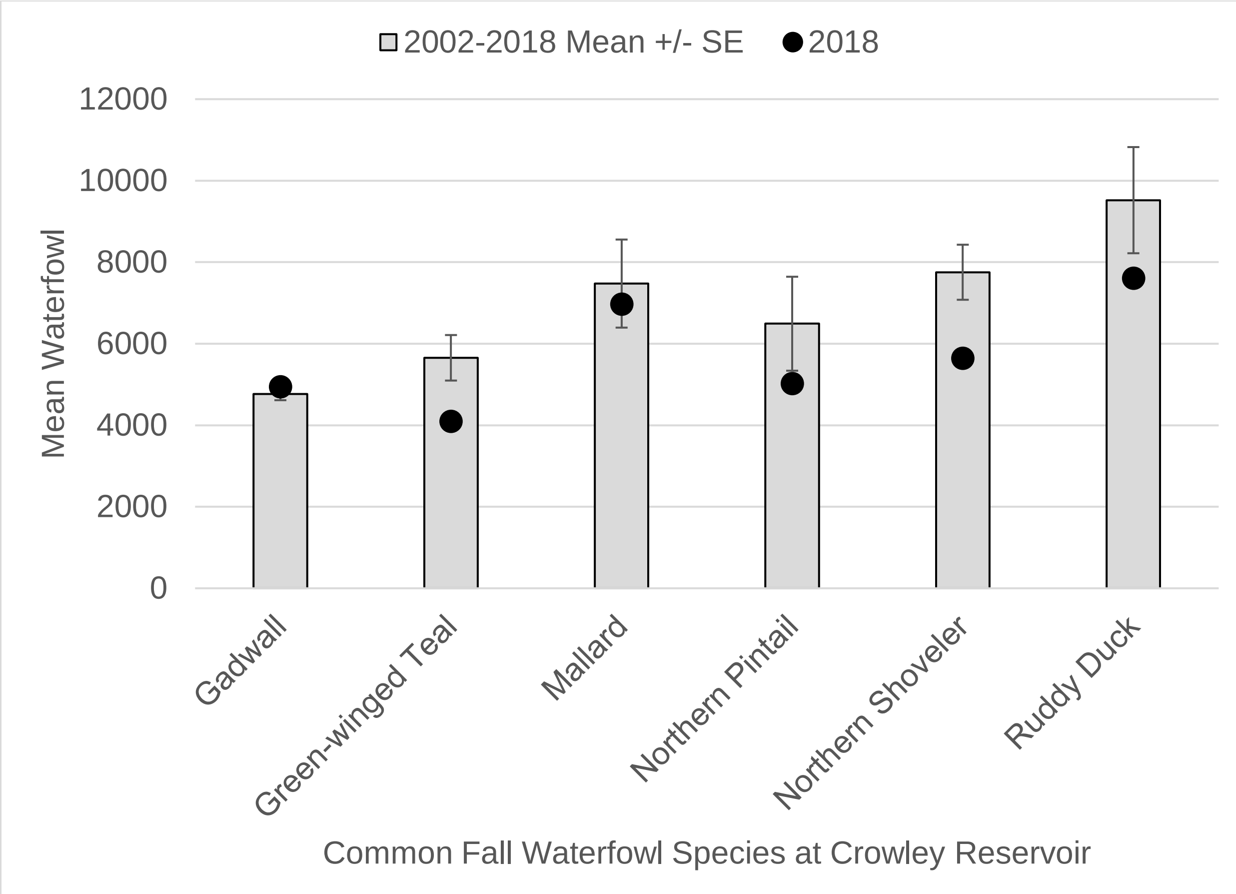

Figure 3.4-16. Crowley Reservoir Waterfowl Species Totals, 2018

Spatial Distribution

During the 2018 surveys, waterfowl at Crowley Reservoir were found concentrated primarily in two main areas – McGee Bay and the Upper Owens River delta (Table 3.4-15). The overwhelming number of waterfowl were in McGee Bay where they can be found all along the length of this shoreline subarea. 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. During the later fall surveys in October, diving ducks can be numerous with large flocks of Ruddy Ducks and other diving species just off shore and on the open water. 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. During early season surveys, few waterfowl are encountered at Chalk Cliffs. Waterfowl continued to show a pattern of late-season use only of the Chalk Cliffs area when significant 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. Hilton Bay has good waterfowl habitat with adjacent meadows and some fresh water inflow, but due to its small size, has supported small numbers of primarily dabbling ducks. 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 typically 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 3.4-15. Crowley Reservoir, Spatial Distribution by Survey, 2018

Survey	CHCL	HIBA	LASP	MCBA	NOLA	SAPO	UPOW
Early September	0	20	295	1965	0	90	2688
Mid-September	0	130	125	3219	17	41	1525
End of September	15	250	180	6112	18	51	2627
Mid-October	70	532	962	6227	86	56	2790
End of October	144	310	525	2827	224	168	905
Mid-November	536	131	231	843	123	145	646
Total waterfowl by shoreline segment	**765**	**1373**	**2318**	**21193**	**468**	**551**	**11181**

Comparison of Reference Data to Evaluate Trends

The comparison surveys have shown that Mono Lake attracts a disproportionally small number of waterfowl, despite its large size (Figure 3.4-17). The long-term mean annual waterfowl use at Mono Lake has been the lowest of the three surveys areas, although there has been some slight overlap in the overall mean with Bridgeport Reservoir. Total waterfowl use in 2018 was below the long-term means for all three survey area. The waterfowl community at Mono Lake also differs notably from the other two survey areas in that it is composed primarily of the few species typically associated with saline lakes. In contrast, the waterfowl communities of Bridgeport and Crowley Reservoirs are more diverse as is typical of fresh water systems.

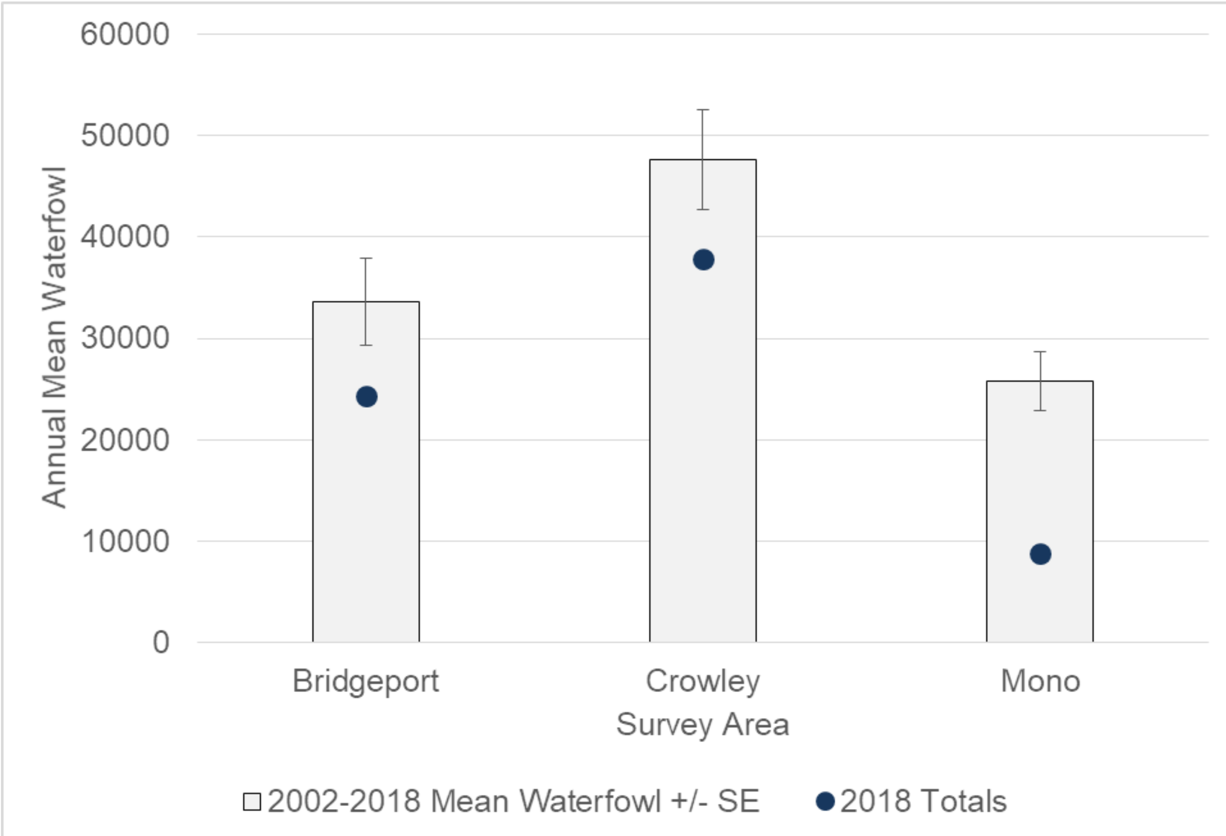

Figure 3.4-17. Comparison of Mean Fall Waterfowl at each of the Three Surveys Areas, 2003-2018

There has been no correlation between annual total or annual peak waterfowl numbers at Mono Lake and either Bridgeport or Crowley Reservoir (Table 3.4-16). Similarly, there have not been any correlations between the total number of Northern Shoveler at the three survey areas (Table 3.4-17). Ruddy Duck numbers at Mono Lake have been negatively correlated with Crowley Reservoir. Ruddy Duck populations at Crowley Reservoir have shown an increase over time, and since 2013, more Ruddy Ducks have been observed at Crowley Reservoir than at Mono Lake.

Table 3.4-16. Correlation of Waterfowl Population Indices - Mono Lake, Bridgeport Reservoir and Crowley Reservoir

	COMPARISON WATERFOWL SURVEY AREA			
	Bridgeport		Crowley	
MONO LAKE VARIABLE	*r*	*p value*	*r*	*p value*
Annual Peak Numbers	0.253	0.344	-0.236	0.931
Annual Total	0.104	0.700	0.0986	0.716

Table 3.4-17. Correlation of Total Annual Northern Shoveler and Ruddy Ducks at the Three Survey Areas

Comparison	Test result	Species	
		Northern Shoveler	Ruddy Duck
Mono Lake vs. Bridgeport	r	0.126	-0.464
	p value	0.642	0.070
Mono Lake vs. Crowley	r	-0.0307	-0.549
	p value	0.910	***0.0275**
Bridgeport vs. Crowley	r	-0.317	0.467
	p value	0.232	0.0683

3.4.4 Waterfowl Population Monitoring Program Evaluation

The Periodic Overview Report included an analysis of the waterfowl survey data to determine if the monitoring program could be streamlined, yet provide indices to the response of the waterfowl population to restoration. The 2018 data were incorporated into the analysis of summer breeding waterfowl population data, brood data, and total fall waterfowl counts, and the conclusions and recommendations put forth in the Periodic Overview Report reevaluated.

The mean waterfowl breeding population size and total broods are useful indices to evaluate long-term trends in the response of breeding waterfowl populations to restoration at Mono Lake. The waterfowl breeding parameters of the number of broods and the number of breeding waterfowl present per survey, and total annual broods and mean breeding population showed significant interrelatedness, suggesting some individual parameters may be useful as indices to breeding population size and productivity. The number of broods observed on any one survey was predictive of total broods, however the strength of this relationship was highest with Survey 3 (r^2_{adj} = 0.834, $p<0.001$) (Table 3.4-18). Likewise, breeding waterfowl numbers on any one survey were also predictive of the yearly mean breeding population size, however the strength of this relationship was highest for Survey 2 (r^2_{adj} = 0.883, $p<0.001$).

A comparable index for which to compare long-term trends for the breeding waterfowl population could be obtained by eliminating survey one and conducting surveys two and three. Survey 3 is most important to conduct for total broods and survey 2 has the strongest correlation with total breeding population. In addition, surveying only areas of potential waterfowl habitat would reduce survey effort without affecting the results.

Table 3.4-18. Regression Results for Breeding Waterfowl Indices

Total Broods vs. Broods Per Survey	**r**	**r^2_{adj}**	**p**
Broods-Survey 1	0.720	0.486	0.001
Broods-Survey 2	0.848	0.701	<.001
Broods-Survey 3	0.919	0.834	<.001
Mean Breeding Population vs. Breeding Waterfowl Per Survey	**r**	**r^2_{adj}**	**p**
Breeding Waterfowl-Survey 1	0.878	0.755	<.001
Breeding Waterfowl-Survey 2	0.943	0.883	<.001
Breeding Waterfowl-Survey 3	0.683	0.431	0.003

Long-term data from both Surveys 1-4 only and all six fall surveys indicate that Mill Creek, Wilson Creek and Simons Spring areas show a strong positive correlation with annual total lakewide waterfowl, and annual Northern Shoveler (Table 3.4-19). The relationship with total annual Ruddy Duck numbers is weak however, as most Ruddy Duck are offshore along the north shore. Under a reduced monitoring schedule, surveys 5 and 6 could potentially be discontinued as the difference in the strength of the correlation between 1-4 surveys and all six surveys is minimal. Based on the 2002-2018 data, Mill Creek, Wilson Creek and Simons Spring would be the most important sites to survey to provide an index to total waterfowl use at Mono Lake, however this data could not be used to track Ruddy Duck populations specifically.

Table 3.4-19. Correlation of Total Annual Waterfowl, Northern Shoveler, Ruddy Duck and Shoreline Subareas of Mill Creek, Simon's Spring and Wilson Creek

MICR,SISP,WICR			
Shoreline Survey Areas	**Total Waterfowl**	**Total Northern Shoveler**	**Total Ruddy Duck**
Fall surveys 1-4	*0.8691	*0.9339	0.3474
All six fall surveys	*0.8814	*0.9278	0.3992

* = significant at $p<0.05$

3.4.5 Waterfowl Survey Discussion

3.4.5.1 Summer Ground Surveys – Mono Lake Shoreline

Breeding Population Size and Composition

In 2018, the breeding waterfowl population showed signs of recovery following the extended drought from 2012-2016. Breeding waterfowl of 2018 appeared to respond to the increase in lake elevation that has occurred over the last years as a result of the extreme wet year in 2016-2017 and the normal runoff year of 2017-2018. Although runoff year 2016-2017 was an extreme wet year, the lake elevation was still low during the summer of 2017, and breeding waterfowl populations still depressed. At elevations below 6,382 feet, brood numbers and brood sizes have been reduced (LADWP 2018), and in 2017, lake elevation was well below 6,382 feet in spring and early summer. In the spring and summer of 2018, lake elevation was between 6381.5 and 6381.8 feet. The breeding waterfowl community has continued to demonstrate a positive response to the primary restoration objective of increasing the level of Mono Lake. In 2018, the total breeding waterfowl and brood numbers were at their highest since 2012 and both values within the long-term 2002-2018 mean.

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). At higher lake elevations, waterfowl breeding habitat quantity and quality at Mono Lake have been increased. One effect of increased lake elevation is that it restores or maintains connectivity between important waterfowl feeding areas at the outflow of springs or creeks, and on shore breeding and brooding areas of freshwater ponds and wetland vegetation. Increases in lake elevation also increase the number and extent of lake fringing ponds. This effect is seen primarily along the south shore, where the number and size of fresh water ponds is greater when the lake has been higher. As lake level drops, ponds along the south shore in particular dry up or become encroached with emergent vegetation.

The abundance and availability of aquatic invertebrates limits the number of breeding waterfowl and waterfowl brood survival (Sjoberg et al. 2000). The increased number of open water fresh or brackish ponds along the south shoreline associated with higher lake elevations creates additional foraging areas for breeding waterfowl and their broods. In addition to increasing the available wetlands, increases in lake elevation have also placed potential breeding ponds closer to favorable feeding areas at the outflow of creeks and springs where densities of *Artemia* may be higher (Dana and Herbst 1977). Thus, lake elevation may not only provide additional areas of high food abundance in temporary wetlands, but decrease exposure of ducklings or adults as they feed on shore.

Lake elevation may also be affecting breeding populations indirectly by affecting brood survival. One process by which this may occur is increased predation exposure and risk. As the lake level decreases, the distance between nesting areas with vegetation and high quality feeding areas, such as spring outflow sites, increases. This will result in an increased distance of overland travel by broods often on exposed barren lakebed between areas of cover and feeding sites. This effect is especially evident along the south shoreline where small changes in lake elevation result in more dramatic changes in degree of shoreline flooding. Not only might this increased distance increase their energy expenditure, but also increase the exposure of young broods to predation. Ducklings are flightless for approximately the first seven weeks of life, and suffer the highest mortality in the first two weeks of life (Ball et al. 1975, Cox et al. 1998). Predation and adverse weather have been cited as major causes of duckling mortality (Cox et al. 1998). Predators of young ducks include coyote (*Canis latrans*), California Gull (Gates 1962), raccoon (*Procyon lotor*) and mink. Reduced energy expenditures will support higher growth rates of ducklings, providing some protection against adverse weather and predation (Cox et al. 1998). Factors affecting brood survival may ultimately influence the breeding population because of the tendency of waterfowl to return to their natal area to breed (Doherty et al. 2002).

The abundance of *Artemia* at Mono Lake has influenced breeding waterfowl populations. Modeling indicates the spring biomass of *Artemia* has been positively correlated with breeding waterfowl numbers. Compared to other avian species, the energetic demands of nesting waterfowl are high, as egg and yolk size are disproportionally large relative to body size (Lack 1968). The availability of invertebrates has been found to be a major proximate factor determining the initiation of egg laying in ducks and waterfowl obtain the protein needed for egg formation through dietary intake on the breeding grounds (Krapu 1974, Choinière and Gauthier 1995). Waterfowl were frequently seen feeding in near shore, often in the outflow of springs or creeks where *Artemia* is expected to be an abundant prey item. *Artemia* numbers may influence the breeding population by affecting female condition.

Spatial distribution

Waterfowl breeding populations are concentrated into highly localized areas around the shoreline of Mono Lake, where fresh water resources occur for young ducklings. While breeding waterfowl have been observed in all subareas, long-term data indicate use has been concentrated in three subareas: Wilson Creek, Mill Creek and South Shore Lagoons. Even within those subareas, breeding waterfowl use has been concentrated in areas of appropriate nesting or feeding habitat. South Shore Lagoons and Wilson Creek and Mill Creek have supported a similar proportion of the overall breeding waterfowl community. The South Shore Lagoons has produced more broods, with most breeding activity in the Goose Springs area.

In 2018, the lakewide waterfowl breeding population was the highest since 2012, however a shift in distribution was noted in response to habitat conditions. Whereas Mill Creek and Wilson Creek are generally high use areas, use in these areas was below the long-term mean. The Goose Springs area of South Shore Lagoons absorbed most of the increased breeding activity observed in 2018. In 2018, multiple shoreline ponds were present in close proximity to one another on shore at Goose Springs including the freshwater ponds at the spring heads, a large open freshwater pond downstream, and a large brackish pond immediately on shore. These conditions were favorable nesting, breeding, feeding and escape. It is uncertain why use declined in Mill Creek, however field observations indicate that the high creek flows experienced in the summer of 2017 and 2018 created a deep channel at the mouth of the creek, effectively eliminating the shallow feeding areas where the majority of waterfowl feeding and brooding have been observed to take place.

Habitat Use

During development of the Plan, it was noted that there was little information on how waterfowl use Mono Lake habitats. Ground surveys allow an opportunity to record specific habitat and microhabitat types used by waterfowl. On shore water features including freshwater streams, freshwater ponds, brackish ponds, hypersaline ponds, and mudflats are all generally heavily used by all dabbling duck species.

Canada Goose is generally the only species that regularly uses meadow/marsh habitat and feeds with broods in alkaline wet meadow habitats near or on shore. Canada Goose was also regularly observed swimming in open water areas offshore, frequently in response to disturbance. On-shore water features were the landtype most heavily used by dabbling ducks, with freshwater and brackish ponds receiving the most use. Both freshwater and brackish ponds were used by ducks for feeding and resting. Ria areas were used almost exclusively for feeding. Mallard showed the highest proportional use of on shore water features such as fresh and brackish ponds, with relatively less use of ria.

3.4.5.2 Summer Ground Surveys - Restoration Ponds

Use of the Restoration Ponds by breeding waterfowl in 2018 was significantly affected by continuing infrastructure problems. The County Ponds were dry throughout summer of 2018, and County Pond East typically supports a large percentage of breeding waterfowl in the Restoration Pond area. Use of the DeChambeau Ponds was also very low, and infrastructure and water delivery problems present in 2017 continued.

3.4.5.3 Fall Aerial Counts

Mono Lake - Population size and species composition

Waterfowl use at Mono Lake in fall 2018 was extremely low and well below the long-term average. The causative factors influencing annual fall waterfowl numbers at Mono Lake have not been clearly identified. *Artemia* cyst production appear to partly explain the annual variation in waterfowl populations at Mono Lake. In open saline waters of Great Salt Lake, Northern Shoveler and Green-winged Teal were found to consume largely *Artemia* cysts and adults. In that study, cysts comprised a larger component of the diet than adult brine shrimp, making up 52% of the biomass of the diet of the shoveler, and 80% of Green-winged Teal diets (Roberts 2013). While some waterfowl species, such as Mallard and geese are typically seen in shoreline ponds or mudflats, other fall migrants including Northern Shoveler, Green-winged Teal and Northern Pintail, congregate near shore at creek deltas. *Artemia* are likely to be the most abundant prey item in these areas, however other potential dietary items may be present. A time budget study has not been conducted of waterfowl use of shoreline areas during fall migration, thus the importance of the different shoreline subareas for feeding, drinking, roosting, or bathing is not known. An understanding of how waterfowl use each subarea would provide a greater understanding of the specific resources available for waterfowl around the lake, and how they are being used.

The two key waterfowl species at Mono Lake are the dabbling duck Northern Shoveler and the diver Ruddy Duck, which generally comprise over 80% of total fall waterfowl numbers. In 2018, these two species only comprised approximately 40% of all fall waterfowl. Multiple factors may influence these migrating populations including productivity on breeding grounds, habitat conditions enroute, weather, and disease. In early fall of 2018, several large Botulism Type C outbreaks affecting waterbirds were recorded along the Pacific Flyway in Oregon, northeastern California and Nevada. Upwards of 25,000 waterfowl, shorebirds and other waterbirds became ill (Retrieved .April 1, 2019 from the Wildlife Health Information Sharing Partnership event reporting system (http://www.nwhc.usgs.gov/whispers/). The potential exists for large die offs such as these along the flyway to impact total waterfowl observed at Mono Lake in any one year. The reasons for the decrease in fall waterfowl numbers at Mono Lake in 2018 is not known, however similar decreases were not observed at either Bridgeport or Crowley Reservoir, suggesting waterfowl may have been responding to conditions specific to Mono Lake.

The differences in the characteristics between individual saline lakes with regard to parameters such as salinity, fresh water inputs, and water depth, can influence the quality of the habitat for waterfowl and therefore species composition and abundance. 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. 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 food resources at individual saline lakes can vary widely, depending on salinity and fresh water inputs. Closed lake systems can vary from brackish (1-3 gm/L) to highly saline (e.g. Mono Lake 80-90 gm/L). At moderate salinity levels aquatic invertebrate communities are more diverse than at higher salinities. Few invertebrate species are tolerant of high salinities, thus highly saline lakes such as Mono Lake have low invertebrate diversity, however, can support large number of some species. Depending on salinity, the invertebrate community of closed lake systems may include *Artemia*, Dipterans (alkali fly, midges), Corixids, water fleas (Daphnia), beetles (Coleptera). 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. For example, experimental studies have shown that at the prediversion salinity of 50 gm/L, twice the diatom diversity would have been supported and greater biomass and diversity of benthic algae (Herbst and Blinn 1998). The highly saline waters also limits the availability of vegetable food sources to isolated fresh water and brackish ponds as the salinity of the lake is above the tolerance of wetland plants.

Birds inhabiting saline environments encounter additional energetic costs associated with osmoregulation. Osmoregulation in waterbirds occurs through physiological, behavioral, or mechanistic adaptations. In some species, ingesting salts while feeding and drinking in saline environments cause large changes in the organs responsible for osmotic regulation including the kidneys, small intestine, and hindgut. Salt glands are the most efficient organ by which waterbirds cope with excess salt. Birds in marine environments have more well-developed salt glands than non-marine species (Gutiérrez 2014). In high salinity environments, the intestines of some birds increase in mass, so that the salt holding capacity, increases and more salt can be routed to the salt glands (Gutiérrez 2014). Salt glands hypertrophy when birds switch from fresh to saline habitats in order to maintain water and electrolyte balance (El-Gohary et al. 2013, Gutiérrez 2014). Maintaining large, functioning salt glands is physiologically demanding. Birds may also osmoregulate through behavioral or mechanistic actions. Behaviorally, birds may avoid saline habitats, or by feeding on prey with lower salt loads, or visit fresh water sources near feeding grounds. Other birds may use mechanical means of decreasing the intake of saline water such as using surface tension to deliver prey to the mouth or using the tongue to compress water off of prey (Rubega 1997, Mahoney and Jehl 1985).

Waterfowl using Mono Lake must balance the energetic costs of migration and molt and with food intake. The two most abundant and widespread secondary producers are brine shrimp and alkali flies. Other food resources are available at lake-fringing brackish and freshwater ponds, however these are localized at particular shoreline areas, and their presence and availability ephemeral.

Waterfowl diets vary according to the feeding environment and available food resources. Food items reported as being important to Northern Shovelers feeding in saline habitats include water boatmen (Corixidae) (Euliss and Jarvis 1991), copepods and rotifers (Euliss 1989), brine shrimp cysts (Roberts 2013, Boula 1986, Vest 2013) and alkali fly larvae (Roberts 2013. Boula 1986) and pupae (Boula 1986). Brine shimp adults are not as digestible and have lower caloric density as compared to other food sources, and may not be selected for when other food is available. The diet of Northern Shovelers at Mono Lake has not been studied; therefore the extent to which they use the various life stages of brine shrimp or alkali fly at Mono Lake is unknown. Although many dabbling duck species consume both vegetable and animal foods, many studies have found a preponderance of animal matter in the diet of Northern Shoveler. In saline lakes that lack aquatic vegetation and have limited vegetative food resources such as Mono Lake, waterfowl species whose diet is composed largely of animal manner can still find resources. Northern Shoveler also has a specialized bill morphology including very closely spaced lamellae, allowing for the effective filtering of small aquatic invertebrates (Gurd 2007). Northern Shoveler may be able to feed more efficiently at Mono Lake than other species, despite saline conditions because of their bill structure.

Although Northern Shoveler may be abundant at saline lakes, they do not have the physiological adaptation of well-developed salt glands for osmoregulation (Roberts 2013). Like most nonmarine waterfowl, Northern Shoveler need access to fresh water daily. Northern Shoveler can forage efficiently at saline sites however supporting only small aquatic invertebrates such as those found at Mono Lake, and osmoregulate through behavioral means by visiting fresh water resources.

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 is such that shallow-water feeding areas, especially those near springs, are widespread 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.

The spatial distribution of waterfowl at shoreline sites in fall suggests that waterfowl habitat at Mono Lake is highly localized. Although the Wilson Creek area makes up <2% of the entire shoreline area, it has supported 45% of all dabbling ducks. The combination of abundant spring flow, extensive wet meadow habitat upgradient, and shallow offshore gradient in the Wilson Creek bay likely contribute to creating a favorable shallow water feeding and loafing area.

The data suggest that waterfowl populations at Mono Lake are responding more to conditions at the lake itself, and have poor correlation to numbers and trends at the nearby freshwater lakes used as comparison sites.

4.0 SUMMARY OF THE MONO BASIN WATERFOWL HABITAT RESTORATION PROGRAM

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.

The progress made toward the target lake elevation has been slow. Although it has been 24 years since Decision 1631, the elevation of Mono Lake is still well below the target lake level. Despite the four periods of lake level rise, in which the lake rose 3 to 4 feet each time, there has been an overall trend of decreasing lake elevation. The ecological changes associated with this decrease have also affected lake-shore fringing waterfowl habitats, at least temporarily.

Restoration in the Mono Basin along the tributaries to Mono Lake has included the establishment of perennial flows in Rush Creek and Lee Vining Creek, and the reopening of side-channels in Rush Creek to restore waterfowl and riparian habitat in the Rush Creek bottomlands. The rewatering of Rush and Lee Vining Creek has undoubtedly provided significant ecological benefits to the wildlife and ecosystem of the Mono Basin. The benefits of the recovery of riparian resources along Rush and Lee Vining Creek have been described for songbird populations (Heath 2003). Restoration has improved nesting habitat for waterfowl species that nest in riparian areas in the Mono Basin, including Green-winged Teal and Mallard, due to the increase in availability of perennial water for feeding and escape by broods, and by supporting the growth of meadow and wetland vegetation for nesting. In wet years, wetlands of the Rush Creek bottomlands become inundated, creating small open water ponds that are attractive to nesting and migrating waterfowl. The total number of waterfowl that use the riparian corridor, however, is small (House 2013), especially as compared to the lake-fringing habitats, and the channel-opening restoration projects have likely had more direct conservation value for riparian-dependent species in the Mono Basin than for its waterfowl populations. Additional benefits may be realized for waterfowl as these systems mature. Some early indications of this may be an increase in breeding Green-winged Teal over time. Green-winged Teal are generally most closely associated with the fresh water creek systems. As these wetland and riparian areas have recovered from past water diversions, they may be providing improved habitat for nesting Green-winged Teal.

The establishment of perennial flow in Rush and Lee Vining Creeks has resulted in the reestablishment of deltas and presumed hypopycnal areas along the perimeter of Mono Lake near these outflow areas. These delta areas are very important wildlife areas, and are used by many waterbirds for feeding, resting, bathing, and drinking. Although waterfowl use of the deltas has been higher than that observed along the riparian corridors, the use of the restored Rush and Lee Vining Creeks by fall migratory waterfowl has accounted for less than 5% of all waterfowl use. In the delta areas, waterfowl have been observed close to shore during summer ground counts and fall aerial surveys. Extensive use by waterfowl of areas presumed to be hypopycnal areas, such as those offshore of Rush and Lee Vining Creeks, has not been evident. The extent of these hypopycnal areas, and how they benefit waterfowl, has not been demonstrated conclusively.

Order 98-05 provided for funds to be set aside for waterfowl habitat restoration in the Mono Basin. The Restoration Ponds represent a potential location in the Mono Basin for waterfowl habitat enhancement. Waterfowl habitat at the Restoration Ponds might benefit from upgrades to the existing water delivery system, to allow for more flexibility in water delivery to individual ponds. The system is also in need of repair, as recent failures in the water delivery infrastructure have affected water deliveries to individual ponds.

Although the Plan includes a rather exhaustive monitoring program, the Waterfowl Habitat Restoration monitoring program suffers from a lack of coordination between the various monitoring components. This may limit the ability to interpret patterns in waterfowl use of Mono Lake in response to restoration, should they exist. The Waterfowl Restoration Program might also benefit from coordinated monitoring schedules for some tasks, and a more focused monitoring approach, to address currently unanswered questions. These changes might not only be beneficial in terms of understanding waterfowl habitat and use, but would also add to our understanding of the ecological factors that may influence use of Mono Lake by other important waterbird groups.

5.0 RECOMMENDATIONS

The time period for restoration of waterfowl habitat has been greatly extended due primarily to the protracted time period that has been required for lake elevation recovery. The Plan states that monitoring will focus on waterfowl habitats rather than a projected number of waterfowl. The Plan also states that monitoring should consider the duration required for restoration to occur, the goals and objectives of the particular project, and the level of effort needed to collect the data (Drewien, Reid, and Ratcliff 1996). Decision 1631 recognized that raising the elevation of Mono Lake could take roughly 29 to 44 years depending upon the assumptions made regarding future hydrology. However, the original monitoring Plan was developed under the assumption that the lake elevation would recover and reach its target level within approximately 20 years after Decision 1631.

The Plan had proposed a schedule for the discontinuation of components of the monitoring program, most sun-setting after the lake had reached its target elevation, and ecosystem stability demonstrated. Decision 1631 required LADWP to prepare a restoration plan with reasonable, financially feasible restoration measures. LADWP has complied with the Decision 1631 and Order 98-05, and in some cases, has conducted monitoring in excess of that originally proposed. Waterfowl habitat restoration is not complete, however, since the target lake level has not been achieved, and some required monitoring also has yet to be completed. In light of the extended time period required for restoration, and the level of monitoring that has been conducted to date, a less-frequent but more focused approach for long-term monitoring of waterfowl habitat in the Mono Basin is proposed.

Lake-fringing ponds, springs, deltas, and nearshore habitats of spring outflow areas, are the habitats most used by waterfowl and many other waterbird species that use Mono Lake. Changes in these areas are not being adequately assessed by the current monitoring program. Future monitoring should focus on changes to these habitats as a function of lake level changes, as well as long-term changes in these habitats. This data may be useful to evaluate the response of waterfowl and other waterbird species to lake level changes at Mono Lake.

Specific recommendations are presented below for each component and Table 5-1 provides a comparison of the current required monitoring, and the proposed changes to the monitoring program. Recommendations for the limnology and waterfowl population monitoring put forth in the Periodic Overview Report were reevaluated during the preparation of this report, and some modifications made. Although performing particular monitoring tasks at certain lake elevations might be ideal in order to insure data at various lake elevations, this is often difficult in practice. Data collection on 5-year intervals, as has been done for vegetation monitoring, will continue to provide data on long-term trends, and at various lake elevations over time.

The following are my specific recommendations that should be taken into consideration for the waterfowl monitoring program:

1) **Lake elevation** – No change to current monitoring; continue to monitor lake elevation on a biweekly basis

2) **Stream Flows** - No change to current monitoring; continue to monitor daily stream flow

3) **Spring Surveys** – Continue to monitor at five-year intervals

4) **Limnological monitoring** – Continue the annual limnological monitoring program, but incorporate spatial and temporal reductions. Reduce the monitoring of water chemical/physical properties and *Artemia* populations between March and November at 8 stations (2, 4, 6, 7, 8. 9, 10, and 12). Conduct sampling of water chemical/physical properties at Station 6 in February and December. Continue *Artemia* fecundity sampling at 8 stations between June and October. Discontinue the instar analysis.

5) **Vegetation transects** – the vegetation transect sampling was required at five-year intervals until 2014, at which point LADWP could evaluate the need to continue the program. As the monitoring data indicate the establishment of relatively stable, late seral vegetation communities, I recommend suspending the vegetation transect study at this time. Once the target lake elevation is reached, conducting a final year of vegetation transect monitoring could be instructive.

6) **Landtype mapping** – continue at 5-year intervals; conduct ground-truthing and the use of salinity meters to ensure proper classification of shore-fringing water features such as freshwater, brackish and hypersaline ponds. Consider documenting community composition by shoreline subarea, at least for areas at or below the 6,392 foot contour.

7) **Fall waterfowl counts** – Develop a long-term monitoring plan in consult with the Parties, taking into consideration effectiveness, cost, safety, and ease of implementation. The aerial counts that have been conducted annually since 2002 are effective and efficient. Difficulties have been encountered over the last

few years in maintaining air support. LADWP has been very fortunate to have been able to hire local pilots over the last 17 years that are highly qualified with extensive experience with local mountain conditions. The benefit of having a pilot based locally is familiarity with flying conditions, ease of coordinating and rescheduling flights as needed with changing weather conditions, allowing a more consistent survey schedule.

Ground based surveys would have an advantage in that implementation would not rely on the future availability of aircraft. Ground-based surveys would also impart less risk to observers, as low level, slow speed waterfowl flights carry an inherent risk. Based on the effective analysis conducted this year, ground-based surveys of Mill Creek, Simon’s Spring and Wilson Creek would provide an index of both total fall waterfowl use and the lakewide Northern Shoveler population. Ground-based surveys, however would not allow for the effective tracking of Ruddy Duck numbers. The trade-offs of the various options should be considered.

8) **Fall comparison counts** - The Plan recognized that the importance of comparison data might justify the need to continue the counts on an annual basis. The data suggest that waterfowl populations at Mono Lake are responding more to conditions at the lake itself, and have poor correlation to numbers and trends at the nearby freshwater lakes used as comparison sites. Although the comparison data has been instructive and has helped substantiate conclusions regarding waterfowl response to local conditions at Mono Lake, annual counts at these nearby freshwater reservoirs are not necessary to evaluate the response of waterfowl to restoration at Mono Lake. Comparison counts at Bridgeport and Crowley Reservoirs can be reduced to every five years to continue to provide an index to long-term trends in Mono County that may influence use of Mono Lake. Supporting the continuation of comparison counts at nearby saline lake systems such as Owens Lake would also be useful.

9) **Summer ground counts** - Reduce the number of counts per year to two (conduct surveys 2 and 3 only). These surveys will not only continue to provide an index to the response of breeding waterfowl to restoration (brood number, breeding waterfowl population size), but will be useful for the documentation and evaluation of on-the-ground conditions. At the target lake elevation, conduct all three surveys. Survey effort may be further reduced by only surveying areas supporting potential waterfowl habitat such as shoreline ponds, spring outflow areas and eliminating areas of extended barren playa without ponds or springs.

10) **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. LADWP should complete the second time budget study in either 2019 or 2020 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 hypopycnal areas for feeding, resting, or drinking.

11) **Conduct a hypopycnal area investigation.** It was hypothesized in the Decision 1631 that "Near the mouths of the tributary streams, a phenomena called 'hypopycnal stratification' occurs in which the lighter fresh water flowing into the lake floats on the top of the denser saline water already in the lake." Furthermore, Section 6.4 of Order 98-05 states that the lake level of 6,392 feet will restore a significant amount of waterfowl habitat by restoring large hypopycnal areas near the mouths of Rush and Lee Vining Creek. If hypopycnal areas do not occur, or if waterfowl are not using them to the extent proposed, then expectations regarding the response to restoration may need to be reevaluated. The current limnological monitoring method is not designed to accurately test this hypothesis because stations are too far spread from each other and also, most importantly, are far from the deltas. Relating limnological monitoring to waterfowl monitoring is crucial to understand waterfowl use of Mono Lake.

The Waterfowl Director recommends conducting an investigation into the existence of and spatial/temporal extent of hypopycnal stratification to be able to relate the findings to waterfowl use. This investigation would be a short-term focused study intended to demonstrate the presence and extent of hypopycnal areas at specified locations, including Rush Creek, Lee Vining Creek, and Wilson Creek, and possibly others. This study would be conducted during peak runoff periods (June/July) and again in fall (September) during peak waterfowl migration. The study would preferably be conducted the same year as the time budget study. The study would include limnological sampling along a transect perpendicular to the shoreline to document salinity profiles and invertebrate abundances.

12) **Conduct an invertebrate inventory at the Mono Lake springs.** It is recommended that this be conducted in conjunction with the next spring survey. Productivity may be related to water quality, surrounding vegetation, or substrate.

The differences between springs and the food resources they support may help explain the spatial distribution and habitat use patterns of waterfowl at Mono Lake that may be influencing populations. The results of this study will be evaluated, and further recommendations made.

13) **Develop techniques to improve the documentation of annual changes in shoreline habitats.** Annual monitoring of shoreline habitat is still recommended, however methods of documenting the conditions should be improved. The current method of taking photographs annually from a helicopter provides only a qualitative visual assessment of the response of important waterbird habitat features to lake elevation changes. In order to focus on changes to important habitat features, an improved method of documenting the availability of shoreline ponds that is both feasible and efficient should be developed. One method that could be explored involves the use of an unmanned aerial vehicle to conduct the annual photography of shoreline habitats. The use of a UAV would likely improve the quality and usefulness of the images obtained by being able to more precisely control the location, angle, elevation and height above ground from which the images are taken. This monitoring could focus on specific areas which are of interest due to waterfowl use or the anticipated changes in shoreline habitat.

14) **Explore the option of conducting waterfowl counts using an unmanned aerial vehicle**. The reliability of results from aerial surveys of waterfowl depends on the experience and training of the observer, lighting conditions, and detectability of the species present. Aerial surveys of Mono Lake also require a highly trained pilot with experience in low level, low speed, high altitude flying, which comes with inherent risk. Use of a UAV may allow improved documentation of fall waterfowl surveys, and some studies indicate that the accuracy of counts may be improved.

15) **Consider repairs or upgrades to the infrastructure of the Restoration Ponds for the purpose of waterfowl habitat improvement in the Mono Basin.** Currently, the infrastructure of the ponds is in a state of disrepair. Only a portion of the $275,000 originally earmarked for waterfowl restoration projects in the Mono Basin has been used as the other potential waterfowl habitat improvement projects including prescribed fire and the development of scrapes were determined by the Parties to be either not feasible, impractical, or insufficient benefit to justify. Habitat at the ponds might be enhanced by rotational or seasonal flooding of ponds as opposed to permanent inundation of just a few ponds.

16) **Improve the sharing of information between LADWP and California State Parks regarding tamarisk locations and treatment efforts so that efforts are not duplicated.** Although an interagency program has not been established to control saltcedar or other non-native vegetation, LADWP has been opportunistically treating salt cedar along the creeks and California State Parks is also conducting surveillance and treatment. The sharing of information between agencies would assist in assessing the progress toward eradication efforts.

17) **Reevaluate the Mono Basin Waterfowl Habitat Restoration Monitoring Program after the 2022 monitoring season.**

Table 3.4-1. Summary of the Current and Recommended Changes to the Waterfowl Habitat Monitoring Program

Mono Basin Habitat Restoration Monitoring Program; Current Program and Recommended Changes			
Monitoring Component/ Recommended Measure	**Description**	**Required Frequency per Order 98-05 and 1996 Plan**	**Recommendation**
Hydrology	Lake Elevation	Weekly through one complete wet/dry cycle after the lake level has stabilized.	No change
	Stream Flows	Daily through one complete wet/dry cycle after the lake level has stabilized.	No change
	Spring Surveys	5-year intervals (August) through one complete wet/dry cycle after the lake level has stabilized.	No change; continue to monitor at 5-year intervals
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. • Conductivity and water temperature profiles at 9 stations February-December • 9-m integrated sampling for ammonium and chlorophyll at 7 stations February-December • DO, Ammonium, Chlorophyll *a* depth profile at Station 6 February-December • *Artemia* population sampling at 12 stations February-December • *Artemia* fecundity at seven stations	Continue annual monitoring with temporal and spatial reductions. • All monitoring at Stations 2, 4, 6, 7, 8, 9, 10, and 12 between March and November • All monitoring but *Artemia* population sampling at Station 6 in February and December • *Artemia* fecundity at Stations 2, 4, 6, 7, 8, 9, 10, and 12 between Jun and October • Discontinue instar analysis

Table 5-1. Cont. Summary of the Current and Recommended Changes to the Waterfowl Habitat Monitoring Program

Mono Basin Habitat Restoration Monitoring Program; Current Program and Recommended Changes			
Monitoring Component/ Recommended Measure	**Description**	**Required Frequency per Order 98-05 and 1996 Plan**	**Recommendation**
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.	Suspend the vegetation transect monitoring at this time. Once the target lake elevation is reached, conduct a final year of vegetation transect monitoring.
	Aerial photographs of lake-fringing wetlands and Mono Lake tributaries	Five-year intervals until target lake elevation of 6,392 feet is achieved.	Continue at five-year intervals; conduct ground-truthing to ensure proper classification of shore-fringing water features coincident with this mapping.
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 through one complete wet/dry cycle after the target lake elevation of 6,392 feet is achieved. From 2002-2018, six aerial counts have been conducted at Mono Lake, Bridgeport Reservoir, and Crowley Reservoir	Develop an acceptable long-term monitoring plan based on effectiveness, cost, safety, and ease of implementation. An initial proposal is to conduct four fall grounds counts two-week intervals at Mill Creek, Wilson Creek and Simons Spring, starting the first week of September. Conduct six aerial surveys of Mono Lake once every five years. These six aerial counts are to be at two-week intervals as was conducted from 2002-2018, until the lake reaches the target elevation of 6,392 feet and goes through one complete wet/dry cycle. Reduce frequency of fall comparison counts at Bridgeport and Crowley Reservoirs to every five yrs. Explore the use of an unmanned aerial vehicle (UAV) in conducting future waterfowl counts at Mono Lake.
	Aerial photography of waterfowl habitats	Conducted during or following one fall aerial count. All waterfowl population survey work will continue through one complete wet/dry cycle after the target lake elevation of 6,392 feet is achieved.	Explore the use of an unmanned aerial vehicle (UAV) or other techniques for this annual monitoring activity.

Mono Basin Habitat Restoration Monitoring Program; Current Program and Recommended Changes			
Monitoring Component/ Recommended Measure	**Description**	**Required Frequency per Order 98-05 and 1996 Plan**	**Recommendation**
Waterfowl Population Surveys and Studies, continued	Ground counts	Total of eight ground counts annually (two in summer, six in fall). All waterfowl population survey work will continue through one complete wet/dry cycle after the target lake elevation of 6,392 feet is achieved. From 2002-2018, three summer ground counts were conducted; fall counts were done via aerial surveys	Reduce the number of summer counts per year to two (conduct survey 2 and 3 only) and only survey appropriate breeding habitat. Upon reaching target elevation, conduct all three surveys to document population at 6,392 feet. Four fall ground counts will be conducted, replacing the aerial counts.
	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.	A time budget study was completed in fall 2000 on Ruddy Ducks. It is recommended that LADWP complete the second time budget study focusing on shoreline use by waterfowl by the end of 2020.

6.0 LITERATURE CITED

American Public Health Association (APHA). 1998a. Method 4500-NH3 H. Flow Injection Analysis (PROPOSED) in Standard Methods for the Examination of Water and Wastewater, 20th Edition. Clesceri, L. S., Greenberg, A. E., and Eaton, A. D., eds. Washington DC; 1998. pp. 4-111–4-112.

American Public Health Association (APHA). 1998b. Method 4500-NO3 I. Cadmium Reduction Flow Injection Method (Proposed) in Standard Methods for the Examination of Water and Wastewater, 20th Edition. Clesceri, L. S., Greenberg, A. E., and Eaton, A. D., eds. Washington DC; 1998. pp. 4-121–4-122.

Ball, I.J., D.S. Gilmer, L.M. Cowardin, and J.H. Riechmann. 1975. Survival of Wood Duck and Mallard Broods in North-Central Minnesota. The Journal of Wildlife Management. Vol. 39. No. 4. Pp. 776–780.

Beschta, R. L. 1994. Rush Creek – Flows, Channels, and Riparian Conditions: Pre-1941 and Today.

Boula, K.M. 1986. Foraging ecology of migrant waterbirds, Lake Abert, Oregon. MS Thesis, Oregon State University, Corvallis.

Browne, R.A. and G. Wanigasekera. 2000. Combined effects of salinity and temperature on survival and reproduction of five species of *Artemia*. J. Exp. Marine Bio. and Ecology 244:29–44.

Choinière, L, and G. Gauthier. 1995. Energetics of reproduction in female and male greater snow geese. Oecologia 103(3): 379–389.

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

Cox, R.R., M.A. Hanson, C.C. Roy, D.H. Johnson, and M.G. Butler. 1998. Mallard duckling growth and survival in relation to aquatic invertebrate. The Journal of Wildlife Management, Vol. 62, No. 1. pp. 124–133.

Dana, G. L. 1981. Comparative population ecology of the brine shrimp *Artemia*. Master's thesis. San Francisco State University. San Francisco, CA.

Dana, G. and D. Herbst. 1977. Secondary producer level. *In* An ecological study of Mono Lake, California. David Winkler Ed. University of California, Davis.

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

Doherty, P., J. Nichols, J. Tautin, J. Voelzer, G. Smith, D. Benning, V. Bentley, J. Bidwell, K. Bollinger, A. Brazda, E. Buelna, J. Goldberry, R. King, F. Roetker, J. Solberg, P. Thorpe, and J. Wortham. 2002. Sources of variation in breeding-ground fidelity of mallards (*Anas platyrhynchos*). Behavioral Ecology 13(4): 543–550.

Drewien, Reid and Ratcliff. 1996. Mono Lake Basin Waterfowl Habitat Restoration Plan. Prepared for Los Angeles Department of Water and Power. February 1996.

Euliss, N. H. 1989. Assessment of drainwater evaporation ponds as waterfowl habitat in the San Joaquin Valley, California. Ph. D. thesis. Oregon State University.

Euliss, N. H. and R. L. Jarvis. 1991. Feeding ecology of waterfowl wintering on evaporation ponds in California. The Condor 93: 582–590.

El-Gohary, Z., Fawkeia I. E., Hanaa A. H. and Hamoda, A. M. 2013. Structural Adaptations Associated with Osmotic Stress of The Salt Gland in Birds. The Egyptian Journal of Hospital Medicine. Vol 51: 332–345.

Ficklin, D.L., I.T. Stewart, and E.P. Mauer. 2013. Effects of projected climate change on the hydrology in the Mono Lake Basin, California. Climate Change. 116(1): 111–131.

Gates, J. M. 1962. Breeding biology of the Gadwall in northern Utah. Contribution of the Utah Cooperative Wildlife Research Unit. U.S. Fish and Wildlife Service.

Gollop, J. B, and W. H. Marshall. 1954. A guide for aging duck broods in the field. Mississippi Flyway Council. Tech. Sect. Report.

Gurd, D. B. 2007. Predicting resource partitioning and community organization of filter-feeding dabbling ducks from functional morphology. The American Naturalist 169(3):334–343.

Heath, H. 1924. The external development of certain phyllopods. Journal of Morphology Vol 38(4):453–483.

Heath, S. 2003. Mono's tributary streams as songbird habitat. Mono Lake Committee Newsletter. Spring 2003.

Herbst, D. 1977. Entomology. *In* An ecological study of Mono Lake, California. David Winkler Ed. University of California, Davis.

Herbst, D. 1990. Distribution and abundance of the alkali fly (*Ephydra hians*) Say at Mono Lake, California (USA) in relation to physical habitat. Hydrobiologia 197: 193–205.

Herbst, D. 1993. Natural history of the Mono Lake alkali fly. Appendix I. Mono Basin EIR. Prepared by Jones and Stokes.

Herbst, D., and T. J. Bradley. 1993. A population model for the alkali fly at Mono Lake: depth distribution and changing habitat availability. Hydrobiologia 267: 191–201.

Herbst, D., and Dana, G. 1977. Salinity tolerance in *Ephydra* and *Artemia*. *In* An ecological study of Mono Lake, California. David Winkler Ed. University of California, Davis.

Herbst, D., and Dana, G. 1977. Entomology. *In* An ecological study of Mono Lake, California. David Winkler Ed. University of California, Davis.

Herbst, D. and D. Blinn. 1998. Experimental mesocosm studies of salinity effects on the benthic algal community of a saline lake. J. Phycol. 34:772–778.

Herbst, D., F. P. Conte, and V. J. Brookes. 1988. Osmoregulation in an alkaline salt lake insect, *Ephydra* (*Hydropyrus*) *Hians* Say (Diptera: Ephydridae) in relation to water chemistry. Journal of Insect Physiology 34:903–909.

Hofer, S. 2003. Determination of Ammonia (Salicylate) in 2M KCl soil extracts by Flow Injection Analysis. QuikChem Method 12-107-06-2-A. Lachat Instruments, Loveland, CO.

Horne, A. J. 2003. Report on Bridgeport Reservoir beneficial use impairment: limnology in the summer-fall 2000 and comparisons with 1989. Report prepared for Lahontan Regional Water Quality Control Board. South Lake Tahoe.

House, D. 2013. Assessment of waterfowl habitat in the Rush Creek Bottomlands. Prepared for the State Water Resources Control Board. May 2013.

Jehl, J. R. 1986. Biology of the Red-necked Phalarope (*Phalaropus lobatus*) at the western edge of the Great Basin in fall migration. Great Basin Naturalist Vol. 46. No. 2, Article 1.

Jehl, J. R. 1988. Biology of the Eared Grebe and Wilson’s Phalarope in the nonbreeding season: a study of adaptations to saline lakes. Studies in Avian Biology No. 12. Cooper Ornithological Society.

Jehl, J. R. 2002. Waterfowl populations at Mono Lake, California, 2001. Hubbs Sea World Research Institute. Technical Report 2002-330.

Jellison, R., H. Adams, and J. Melack. 2001. Reappearance of rotifers in hypersaline Mono Lake, California, during a period of rising lake levels and decreasing salinity. Hydrobiologia 466(1): 39–43.

Jellison, R. G.L. Dana, and J.M. Melack. 1989a. Phytoplankton and brine shrimp dynamics in Mono lake, California. 1988 Annual Report to the Los Angeles Department of Water and Power.

Jellison, R. G.L. Dana, and J.M. Melack. 1990. Phytoplankton and brine shrimp dynamics in Mono Lake, California. 1989 Draft Annual Report to the Los Angeles Department of Water and Power.

Jellison, R. and J.M. Melack. 1993. Meromixis in hypersaline Mono Lake, California. 1 Stratification and vertical mixing during the onset, persistence, and breakdown of meromixis. Limnol. Oceanogr. 38(5): 1008–1019.

Jellison, R., J. Romero, and J.M. Melack. 1998. The onset of meromixis during restoration of Mono Lake, California: Unintended consequences of reducing water diversions. Limnol. Oceanogr. 43(4): 706–711.

Jellison, R., K. Rose and J. M. Melack. 2003. Assessment of internal nutrient loading to Crowley Lake, Mono County. Final Report to the State Water Resources Control Board.

Jellison, R. 2011. Field and laboratory protocols for Mono Lake limnological monitoring. Prepared for Los Angeles Department of Water and Power. June 21, 2011.

Jellison, R. and K. Rose. 2011. Mixing and Plankton Dynamics in Mono Lake, California. Marine Sciences Institute. University of California, Santa Barbara.

Knepel, K. 2003. Determination of Nitrate in 2M KCl soil extracts by Flow Injection Analysis. QuikChem Method 12-107-04-1-B. Lachat Instruments, Loveland, CO.

Krapu, G., and A. Klett, and D. Jorde. 1983. The effect of variable spring water conditions on Mallard reproduction. The Auk 100: 689–698.

Lack, D. 1968. Ecological adaptations for breeding in birds. London.

LADWP. 1996. *Mono Basin Waterfowl Habitat Restoration Plan.* Prepared for the State Water Resources Control Board. In response to Mono Lake Basin Water Right Decision 1631

Mahoney, S. A., and J. J. Jehl, Jr. 1985. Adaptations of migratory shorebirds to highly saline and alkaline lakes: Wilson's Phalarope and American Avocet. The Condor 87: 520:527.

Marvin, K. T. and Proctor, R. R. 1965. Stabilizing the ammonia nitrogen content of estuarine and coastal waters by freezing. Limnology and Oceanography. Vol 10. pp. 288–90.

McBain and Trush, Inc, and R. Taylor and Associates. 2010. Mono Basin stream restoration and monitoring program: synthesis of instream flow recommendations to the State Water Resources Control Board and the Los Angeles Department of Water and Power. Final Report. April 30, 2010.

Melack, J. M. 1983. Large, deep salt lakes: a comparative limnological analysis. Hydrobiologia 105:223–230.

Pietz, P., G. Krapu, D. Brandt, and R. Cox, Jr. 2003. Factors affecting Gadwall brood and duckling survival in prairie pothole landscapes. The Journal of Wildlife Management. Vol 67(3): 564–575.

Roberts, A. J. 2013. Avian diets in a saline ecosystem: Great Salt Lake, Utah, USA. Human-Wildlife Interactions 7(1):158–68.

Rogers, D.B., S.J. Dreiss and D.P. Groeneveld. 1992. Near-Shore Groundwater and Salt-Flat Processes at Mono Lake, *White Mountain Research Station Symposium*, 4, 367–395.

Rubega, M.A. 1997. Surface tension prey transport in shorebirds: how widespread is it? Ibis 139: 488–493.

Russell, I.C. 1889. Quaternary history of Mono Valley, California. University of Michigan Library.

Sjoberg, K., H. Poysa, J. Elmberg, and P. Nummi. 2000. Response of Mallard ducklings to variation in habitat quality: an experiment of food limitation. Ecology 81(2): 329–335.

Starrett, G. L., and W. M. Perry. 1985. Multiple generation salinity acclimation experiment. Report to the Los Angeles Department of Water and Power.

State Water Resources Control Board (SWRCB). 1994. Mono Lake Basin Water Right Decision 1631.

State Water Resources Control Board (SWRCB). 1998. Order WR 98-05. Order requiring stream and waterfowl habitat restoration measures. September 2, 1998.

State Water Resources Control Board (SWRCB). 2013. Mono Basin Settlement Agreement. Settlement agreement regarding continuing implementation of Water Rights Orders 98-05 and 98-07.

Strickland, J.D.H., and T.R. Parsons. 1972. A practical handbook of seawater analysis. Bulletin 167 (2nd ed.). Fisheries Research Board of Canada, Ottawa, Canada.

Thun, M. A., and G. L. Starrett. 1986. The effect of cold, hydrated dormancy and salinity on the hatching of *Artemia* cysts from Mono Lake, California, U.S.A. Water Quality Division, Department of Water and Power, City of Los Angeles, Los Angeles, California.

Vest, J.L. 2013. Winter ecology of waterfowl on the Great Salt Lake, Utah. All Graduate Theses and Dissertations. 2051. https://digitalcommons.usu.edu/etd/2051

Vorster, P. 1985. A water balance forecast model for Mono Lake, California. Monograph 10. USDA Forest Service Region 5.

Williams, W. D. 2002. Environmental threats to salt lakes and the likely status of inland saline ecosystems in 2025. Environmental Conservation 29(2): 154–167.

Winkler, D W. 1996. California Gull (*Larus californicus*), version 2.0. In The Birds of North America (P. G. Rodewald, editor). Cornell Lab of Ornithology, Ithaca, New York, USA.

Wurtsbaugh, W.A., C. Miller, S. Null, R. Justin DeRose, P. Wilcock, M. Hahenberger, F. Howe, and J. Moore. 2017. Nature Geoscience 10: 816–821.