BEFORE THE SECRETARY OF THE INTERIOR

PETITION TO THE U.S. FISH AND WILDLIFE SERVICE TO LIST WILSON'S PHALAROPE (PHALAROPUS TRICOLOR) UNDER THE ENDANGERED SPECIES ACT AS A THREATENED SPECIES AND TO CONCURRENTLY DESIGNATE CRITICAL HABITAT

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MONO LAKE COMMITTEE
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March 28, 2024

Notice of Petition

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      Pursuant to Section 4(b) of the Endangered Species Act ("ESA"), 16 U.S.C. § 1533(b); Section 553(e) of the Administrative Procedure Act, 5 U.S.C. § 553(e); and 50 C.F.R. § 424.14(a), the Center for Biological Diversity hereby petitions the Secretary of the Interior, through the United States Fish and Wildlife Service ("FWS," "Service"), to protect Wilson's phalarope (*Phalaropus tricolor*) as threatened under the ESA. FWS has jurisdiction over this petition. This petition sets in motion a specific process, placing definite response requirements on the Service. Specifically, the Service must issue an initial finding as to whether the petition "presents substantial scientific or commercial information indicating that the petitioned action may be warranted." 16 U.S.C. § 1533(b)(3)(A). FWS must make this initial finding "[t]o the maximum extent practicable, within 90 days after receiving the petition." *Id.* Petitioner also requests that critical habitat be designated for Wilson's phalarope concurrently with the species being listed, pursuant to 16 U.S.C. § 1533(a)(3)(A) and 50 C.F.R. § 424.12.

      Petitioner the Center for Biological Diversity ("Center") is a nonprofit, public interest

environmental organization dedicated to the protection of imperiled species and the habitat and climate they need to survive through science, policy, law, and creative media. The Center is supported by more than 1.7 million members and online advocates throughout the country. The Center works to secure a future for all species, great and small, hovering on the brink of extinction. The Center submits this petition on its own behalf and on behalf of its members and staff with an interest in protecting Wilson's phalarope and its habitat. The Center's Senior Utah Campaigner Deeda Seed is the primary point of contact on this petition.

Ryan Carle is the Science Director for the non-profit Oikonos – Ecosystem Knowledge. Ryan has worked for more than 15 years on shorebird and seabird science and conservation in the U.S. and in South America. Ryan is a co-founder and co-coordinator of the International Phalarope Working Group (founded in 2019). Ryan is a co-author of the Conservation Plan for Wilson's Phalarope, version 2.0 (Castellino *et al*. 2023). Ryan conducts phalarope research at Mono Lake and coordinates collaborative research on phalaropes and their habitat across the western hemisphere with current work in the U.S., Canada, Chile, and Argentina. Ryan received an M.Sc. in Marine Science from Moss Landing Marine Laboratories at San Jose State University in 2014.

Terry Tempest Williams has been called "a citizen writer," a writer who speaks and speaks out eloquently on behalf of an ethical stance toward life. A naturalist and fierce advocate for freedom of speech, she has consistently shown us how environmental issues are social issues that ultimately become matters of justice. She is currently a writer-in-residence at the Harvard University Divinity School and is a member of the Board of Directors of the Center for Biological Diversity.

Benjamin W. Abbott, Ph.D. is an associate professor of aquatic ecology at Brigham Young University. His research and teaching focus on aquatic ecosystems, catchment hydrology, and biogeochemistry. He has published papers and books on Great Salt Lake and Utah Lake and is recognized as an expert in global water security and the ecohydrology in the western U.S.

Ron Larson, Ph.D. is retired from the US Fish and Wildlife Service after working there for two decades as a biologist. Most of that time was spent at the Klamath Falls, Oregon, field office where he was involved with conservation of ESA-listed species and minimizing the environmental impacts of water development. Ron has been monitoring the ecological conditions of Oregon's Lake Abert since 2008. This led to writing "A Natural History of Oregon's Lake Abert in the Northwest Great Basin Landscape," University of Nevada Press, 2024. He is a board member of the Oregon Lakes Association and has written extensively on Lake Abert.

Nathan D. Van Schmidt, Ph.D. is a Science Director for San Francisco Bay Bird Observatory. As head of their Waterbird Research program, he has researched Wilson's phalarope's population dynamics, habitat use, and the drivers of their decline within California, along with numerous other waterbird species within San Francisco Bay. He founded their Sustainable Landscapes research program to develop interdisciplinary models of water sustainability in the context of future development and climate change in the American West. His past work has included work on the saline lake waterbird ecology and the terrestrial ecosystems of the Great Basin for the U.S. Geological Survey.

Utah Physicians for a Healthy Environment (UPHE) is dedicated to protecting the health and well-being of the residents of Utah by promoting science-based education and interventions that result in progressive, measurable improvements to the environment and our health. UPHE

was formed in 2007 during one of Utah's worst inversions. UPHE is the largest civic organization of health care professionals in Utah, and one of the largest in the Western US. The organization consists of approximately 500 health care professionals, including clinicians, academicians, researchers, biologists, toxicologists, engineers, and air quality specialists, and another 7,500 members and supporters of the general public within Utah.

Utah Youth Environmental Solutions (UYES) is a youth-led organization that empowers young people in Utah to mobilize around climate and environmental issues through legislation, education and action. Our mission is to connect students to environmental advocacy by cultivating reciprocal relationships between Utah's youth, environmental organizations, and community leaders.

The Mono Lake Committee is a non-profit citizens' group dedicated to protecting and restoring the Mono Basin ecosystem, educating the public about Mono Lake and the impacts on the environment of excessive water use, and promoting cooperative solutions that protect Mono Lake and meet real water needs without transferring environmental problems to other areas. Supported by 16,000 members, the Committee has worked since 1978 to balance the water needs of people and wildlife, with longstanding support for scientific research to monitor ecological health and implement the recovery of Mono Lake.

Kyriana (Kiki) Tarr is an Ecologist at the non-profit Oikonos–Ecosystem Knowledge. Kiki has studied the ecology of phalaropes at Mono Lake since 2022 and received a B.A. in Environmental Studies and Biology from UC Santa Cruz in 2022.

References have been uploaded via a secure link provided by the FWS Branch of Domestic Listing, and can also be found here:

https://drive.google.com/drive/folders/1okyK0G6gqGaJkfty0rO7sUynlzA4FPSx?usp=sharing

Submitted this 28th of March, 2024:

Deeda Seed, Center for Biological Diversity, on behalf of the petitioners

EXECUTIVE SUMMARY

The Wilson's phalarope (*Phalaropus tricolor*) is a shorebird that breeds in interior North America and undertakes a long-distance migration to South America for winter. It is facing the imminent threat of becoming an endangered species due to the ecological collapse and desiccation of saline lakes in the Great Basin, a critical link in its migratory journey.

After breeding in wetland areas in Canada and the U.S., Wilson's phalaropes migrate to "staging" sites where they rapidly molt feathers and double their body weight in preparation for a 3,000–4,000-mile-long migration to South America. Much of the world population concentrates at large saline lakes during this post-migration staging period, especially at Great Salt Lake, Utah; Lake Abert, Oregon; and Mono Lake, California. Forty to sixty percent of the world population, and up to 90% of the adults in the world, typically occur simultaneously at these sites each year. Large saline lakes provide the unique habitat required by migrating phalaropes, in particular an extreme abundance of alkali fly, brine fly, and brine shrimp prey. Wilson's phalaropes overwinter in South America, where they specialize on saline lake habitat in the High Andes and in lowland Argentina.

Wilson's phalaropes' specialization on saline lakes makes them highly dependent on a small number of crucially important sites. Saline lakes are some of the most threatened habitats on Earth, due to declining water levels primarily caused by diversion of freshwater inflows. Anthropogenic climate change is exacerbating the global decline of saline lakes by increasing evaporation rates and the frequency and intensity of drought. Shrinking saline lakes become increasingly salty, negatively affecting aquatic invertebrates first through sub-lethal impacts on growth and development. Eventually, if lakes become too saline, there are lethal impacts on the ability of algae and invertebrates to reproduce and survive, causing the base of the food web to collapse. Such a collapse at Great Salt Lake, Lake Abert, or Mono Lake would trigger a trophic cascade affecting hundreds of thousands to millions of birds of many species, including the Wilson's phalarope population across a significant portion of its range.

The threat of ecosystem collapse at saline lakes is an imminent near-term reality globally and in the western U.S. In 2022, Great Salt Lake reached its lowest water level in recorded history, exceeding the salinity tolerance thresholds for brine fly and brine shrimp reproduction. Lake Abert dried up in 2014–2015 and 2021–2022, with major negative impacts on brine shrimp, alkali fly, and bird populations. Mono Lake also reached water levels dangerous for its ecological health in 2022. In all these cases, diversion of freshwater for human use was the primary driver of the decline of water levels.

An exceptionally wet winter in 2022–2023 resulted in a moderate, likely short-term, rise in lake levels at these sites. However, one wet winter has not changed the chronic water overuse that keeps these lakes on the brink of collapse. Without major changes to water policy, Great Salt Lake and Lake Abert will be lost as Wilson's phalarope habitat in the near future. Simply put, these sites are currently on track to become dry playas or lifeless brine pools too salty for invertebrates and the web of life they support.

Wilson's phalarope has life history traits that make it particularly vulnerable to habitat loss, including reliance on this small number of saline lakes for the rapid energy-refueling necessary for long-distance migration. The species is estimated to have declined in total

population by approximately 70% since the 1980s. Average counts at four of the most historically important staging sites declined by 36-98% from the 1980s to 2019–2021. While the global population of Wilson's phalarope is estimated to be 1 million individuals, the loss of a single large site like Great Salt Lake or Lake Abert could cause abrupt decline. Many smaller and/or more ephemeral staging habitats for Wilson's phalaropes have already been degraded or lost, reducing "back-up" habitat and increasing the importance of the remaining large saline lake habitat.

Land use and climate change are also threatening Wilson's phalaropes in their wintering and breeding grounds. In South America, their core wintering habitat in the Andes is the center of the "lithium triangle." This area—one of the driest on Earth—has seen a rapid expansion of lithium mining and the accompanying depletion of groundwater and surface water. These and other mining activities are diminishing Wilson's phalarope wintering habitat, putting additional pressure on the species. At their North American breeding grounds, climate change is projected to cause an additional 30–46% loss of suitable nesting habitat by 2100 primarily by reducing surface water in areas used for breeding and staging.

Without immediate protection, Great Salt Lake and Lake Abert, which together constitute a significant portion of the species range, could cease to provide viable Wilson's phalarope habitat in the near future. Likewise, wintering habitat in the Andes will be seriously degraded in the foreseeable future. Wilson's phalarope currently has no specific national or international legal protections, except the Migratory Bird Treaty Act, which only prohibits direct take of birds. Nor are there legally binding state-level protections for the species except in Minnesota. Great Salt Lake and Lake Abert have no legal mechanisms for protection, and little to no guaranteed water rights. There is a lack of regulatory oversight over lithium mining development in South America, with governments promoting mining over environmental protection. Given the present and impending destruction of its habitat, lack of regulatory mechanisms to protect it or its habitat, and other factors, in particular climate change, that threaten its continued existence, ***Wilson's phalarope is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range, qualifying it as Threatened under the Endangered Species Act.***

Contents

I. INTRODUCTION

Among shorebird species, the Wilson's phalarope stands out for its unusual behaviors, diet, migration, and habitat. This species does not fit neatly into the "typical shorebird" paradigms. Firstly, like other phalaropes but unlike all other shorebirds, Wilson's phalaropes spend most of their time swimming, rather than walking on the shore (Frank and Conover 2021, p. 9). While swimming, they often spin in circles to create a vortex drawing prey to the surface (Larson 2023, p. 236-237). Second, breeding phalaropes reverse the typical sex-roles: females are larger and more colorful than males, and they actively vie for the males' attention (Colwell & Oring 1988, p. 569, Jehl 1988, p.36). Once a mate is secured, the females lay the eggs and leave the caring for the eggs and chicks entirely up to the males (Colwell & Oring 1988, p. 575, 577). Finally, their southward migration is a unique "molt migration" where they travel to specific areas to replace feathers before continuing the rest of the journey (Jehl 1988, p. 50-51, Jehl 1999, p 40).

During the molt migration, Wilson's phalaropes depend on hypersaline lake habitat, mostly in the interior of North America (Jehl 1988, p. 50-51, Jehl 1999, p 40). Their tolerance for the salty water pays off as these lakes provide a nearly inexhaustible supply of alkali flies, brine flies, and brine shrimp (Mahoney & Jehl 1985, p. 520, Jehl 1988, p. 1, Frank & Conover 2021, p. 27). At these sites, Wilson's phalaropes more than double their body weight and engage in one of the fastest complete molts of body feathers documented in birds, all in a few short weeks (Jehl 1988, p. 52-53).

After stopping for their molt migration, Wilson's phalaropes embark on an epic migration of approximately 3,500 miles to South America (Jehl 1988, p. 52-53). There, they use saline lagoons high in the Andes mountains and large hypersaline lakes in the lowlands of Argentina (Hurlbert 1984, p. 47, Castellino et al. 2024, p. 11). In making this migration, Wilson's phalaropes create a living link between the saline lake habitats of North and South America. Their migration story connects human communities across hemispheres, and the breathtaking murmurations of their immense flocks have inspired many people to care about saline lake habitats.

Wilson's phalaropes' dependence on saline lakes is a vulnerability for the species, as these habitats have rapidly declined in the 21st century to become one of the most endangered ecosystems on Earth (Wurtsbaugh et al. 2017, p. 1, Wang et al. 2018, p. 926, Wine and Laronne 2020, p. 8). Globally, shorebird species are rapidly declining, especially long-distance migrant species (Thomas et al. 2018, p. 255, Kolecek et al. 2021, p. 1). The Wilson's phalarope is situated at this precarious intersection: a long-distance migrating shorebird dependent on a small number of imminently threatened saline lakes.

Over half the world population of Wilson's phalaropes concentrates at just a few saline lakes in the boreal summer (Jehl 1988, p. 50-51, Jehl 1999, p 40). These sites, particularly Great Salt Lake, Utah, and Lake Abert, Oregon, are experiencing ecological collapse as their levels drop and their salinities increase (Larson et al. 2016, p. 389, Abbott et al. 2023, p. 2). The case of Great Salt Lake has made news around the world, with the lake experiencing ecological deterioration, and on a trajectory toward ecological collapse, since around the year 2000 (USGS 2023, *entire*, Abbott et al. 2023, p. 2, Great Salt Lake Strike Team 2023, p. 17). The lake has

continuously been at elevations with "Adverse effects" or "Serious adverse effects" for wildlife, public health, and industry since the year 2000 (Great Salt Lake Strike Team 2024, p. 16). Salinity now chronically exceeds 13% and regularly exceeds 17%, exceeding the recovery and maintenance thresholds for the algae and other microorganisms constituting the base of the Great Salt Lake food web (Frantz et al., 2023, p. 18). Despite a brief rebound because of record-breaking precipitation in the winter of 2022–2023, the north and south arms of Great Salt Lake remain at "Adverse effects" and "Serious adverse effects" elevations, respectively, and the lake's fundamental situation has not changed (Steed 2024, p. 11). Little progress has been made securing water for the lake (Steed 2024, p. 13-14), and the lake could drop below its 2022 levels within a few years.

Great Salt Lake's decline is attributable to three causes: consumptive water use (67-73%), weather variability (15-23%), and climate change (8-11%; Great Salt Lake Strike Team 2023, p. 11). The lake's level is most sensitive to changes in river inflow and direct precipitation (Mohammed and Tarboton 2012, p. 10). Estimates of consumptive water use in the watershed are not precise but range from 1.8 to 2.5 million acre-feet per year since 2000 (Great Salt Lake Strike Team 2023, p. 13, Merck and Tarboton 2024, p. 20). Partitioning of consumptive water use is similarly uncertain, but in general, agriculture is estimated to account for 64-75% of consumptive water use in the Great Salt Lake watershed, followed by municipal and industrial water use (9-16%), managed wetlands (8-12%), mineral extraction from the lake (7-9%), and reservoir evaporation (1-10%; Abbott et al. 2023, p. 7, Great Salt Lake Strike Team 2024, p. 13, Steed 2024, p. 10).

Based on the best available hydrological and societal-impacts data a minimum streamflow requirement of 2.4 to 2.5 million acre-feet per year to Great Salt Lake has been recommended to restore the lake to a healthy level over the next decade (Abbott et al. 2023, p 13, Steed 2024, p. 25). That is approximately one million acre-feet per year more than the annual average streamflow to the lake since 2012 (~1.6 million acre-feet per year; Abbott et al. 2023, p. 13, Steed 2024, p. 24). Achieving this would require a 30–50% reduction in consumptive water use in the watershed (Abbott et al. 2023, p. 13, Steed 2024, p. 38-41). Despite much local to global attention since 2022, progress on water acquisition for and delivery to Great Salt Lake has thus far fallen far short of these goals. A fund created to purchase water rights for Great Salt Lake has been insufficiently supported and has obtained only 0.03% of the water needed to restore the lake so far (GSLWET 2024, *entire*). A new law designating letting water flow to Great Salt Lake as a "beneficial use" could benefit the lake (Miller 2022, *entire*)**,** but the Utah Legislature has not designated a legally-enforceable target lake level or procured new water rights for the lake. Finally, actually delivering water earmarked for the lake remains challenging because of inadequate water gauging and infrastructure issues (Larsen 2022b, *entire*).

Lake Abert gets most of its water from the Chewaucan River, on which water rights are highly over-allocated and exceed the actual water in the river most years (Moore et al. 2016, p. 151). Upstream diversions annually remove approximately half or more of the flow of the river, primarily for flood-irrigation of grass-hay (Moore et al. 2016, p. 151). As a result of this, Lake Abert has been virtually dry for four of the last ten years (Larson 2023, p. 86), with no substantial progress toward addressing the water diversions driving its desiccation and no state or local-level legal action to protect Lake Abert.

A third important North American saline lake, Mono Lake, California, has better legal protection of its water inflows than Great Salt Lake or Lake Abert, but remains at precariously low lake levels due to ongoing regulated water diversion. At Mono Lake's current water level, its ecosystem remains vulnerable to climate-change impacts such as prolonged drought. Furthermore, if Great Salt Lake and Lake Abert were to collapse, Mono Lake is too small to provide sufficient food during the molt migration for the entire population of Wilson's phalaropes.

The destruction of Wilson's phalarope habitat in the Western Hemisphere is part of a larger pattern of global decline of saline lakes generally (Wurtsbaugh et al. 2017, p. 1). About half of global lake water occurs in saline lakes with no outlet to the ocean (Abbott et al. 2019, p. 5). These terminal lakes provide ecosystem services essential to biosphere integrity and human well-being (Wine and Laronne 2020, p. 2). However, nearly all terminal lakes are experiencing rapid decline due to human water overuse, with approximately 60% of global saline lake area (excluding the Caspian Sea) already gone (Wine et al. 2022, p. 4). The loss of these lakes triggers long-term environmental, health, and economic suffering at continental scales (Hassani et al. 2020, p. 2). There are no examples of successful restoration of large saline lakes, making this one of the greatest unsolved ecological crises of the Anthropocene (Wurtsbaugh et al. 2017, p. 1, Abbott et al. 2023, p. 2).

The lack of legal mechanisms to protect Wilson's phalaropes as a species, coupled with the lack of protection of Great Salt Lake, Lake Abert, and other key habitats, leaves Wilson's phalaropes facing the likelihood of serious population decline in the foreseeable future. Loss of Great Salt Lake alone is enough to cause the Wilson's phalarope population to crash and put the species in danger of regional extirpation. The likely loss of multiple important saline lake habitats in North and South America over the next decade could put the species on the path to extinction. We are petitioning for the species to be listed as Threatened because it is ***likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.***

II. BIOLOGY

A. Taxonomy

Wilson's phalarope, *Phalaropus tricolor.*

There are three phalarope species worldwide — red phalarope (*P. fulicarius*), red-necked phalarope (*P. lobatus)*, and Wilson's phalarope (*P. tricolor*). All are currently taxonomically considered to be in the genus *Phalaropus* and the family Scolopacidae (Ericson *et al*. 2003, p. 7, Chesser et al. 2023, *entire*). Wilson's phalarope is a monotypic species. Genetic analysis suggests that the red and red-necked phalaropes are sister species and that Wilson's phalarope is more distantly related (Dittmann & Zink 1991, p. 773). This also fits with the behavior and ecology of the species, with red and red-necked phalaropes spending significant portions of their annual cycle at sea, while Wilson's phalaropes use mainly terrestrial habitats.

B. Species description

Wilson's phalarope is the largest of the three phalarope species, with a length of 22–24 centimeters (9.25 inches; Colwell & Jehl 2020, *Plumages, molts, and structure section*). Compared to the red phalarope and the red-necked phalarope, it has a longer and thinner bill and proportionately longer neck and legs.

Figure 1*: Female in flight showing the lack of white wing-stripe. © Steve Mlodinow*

Phalaropes are highly aquatic shorebirds with lobed toes that enable them to swim (Hohn 1969, p. 105). Unlike most bird species, phalaropes, including Wilson's, have pronounced reversed sexual dimorphism, in which females are larger and have more brightly colored breeding plumage than males (Colwell & Oring 1988, p. 569, Jehl 1988, p.36). At Mono Lake, California, a staging site where Wilson's phalaropes achieve their maximum mass, female mass averaged 75.6 g (14.6 SD, range 52–122, n = 138) and male mass averaged 56.9 g (10.0 SD, range 38–103, n = 345). In the same sample of birds, female and male wing length averaged 131.7 mm (3.3 SD) and 121.2 (2.8 SD), respectively (Jehl 1988, p. 35).

Wilson's phalarope is distinguished from other phalaropes by its lack of a white wing-stripe in flight (**Fig. 1**) and a completely white rump (Sibley 2014, p. 194). During the breeding season the female has a pale bluish-grey crown, nape, and hindneck, and a conspicuous black band covering the sides of the face and neck (**Fig. 2**). This band is chestnut red on the sides of the mantle, becoming an orange-pink wash on its breast, with the strongest coloration on the sides of the neck. The remainder of the underparts is entirely white. The mantle and upper scapulars are mostly pale grayish, except for chestnut red bands on the edges of the mantle and one across the scapulars. The male's breeding plumage resembles that of the female but is duller (**Fig. 3**); the chestnut red bands are replaced by a duller orange-brown wash and the upperparts (crown, nape, mantle, and scapulars) are blackish-brown instead of pale grey (**Fig. 3**). The

nonbreeding plumage (**Fig. 4**) is similar in both sexes, with entirely pale grayish upperparts except for the white upper tail coverts. A clear white supercilium contrasts with the gray crown and hindneck and the grey stripe behind the eye that extends down to the neck. Juvenile plumage is dark brownish on the upperparts with broad buff fringes on the feathers, giving a scaly appearance. The breast sides are washed with buff, while the rest of the underparts are white, and the legs yellowish.

Figure 2: Female Wilson's phalarope in breeding plumage. Photo by Ron Larson.

Figure 3. A male Wilson's phalarope in breeding plumage. This individual is outfitted with a transmitter—the antenna coming off the back of the tail. Photo by Ryan Carle.

Figure 4*: non-breeding plumage of Wilson's phalarope. Photo by Ryan Carle.*

C. Diet

Unlike most shorebirds, Wilson's phalaropes forage principally while swimming (Frank & Conover 2021, p. 29). They feed mainly on small invertebrates, particularly alkali flies (*Cirrula* [*Ephydra*] *hians)*, brine flies (*Ephydra gracilis*, also known as *E. cenerea*)*,* brine shrimp (*Artemia* spp.), chironomids, and *Daphnia* spp. (Jehl 1988, p. 43, Frank & Conover 2021, p. 27). Foraging Wilson's phalaropes often conspicuously spin around in the water to create a vortex that draws invertebrates to the surface (Jehl 1988, p. 43). They also feed actively on land by rapidly chasing and pecking prey from the ground or air (Jehl 1988, p. 43). They often associate with flocks of other shorebirds, ducks, or flamingo species to feed on invertebrates that those species stir up in the water (Siegfried & Batt 1972, p. 667, Hurlbert 1984, p. 53; Gutierrez & Soriano-Redondo 2018, p. 132).

All of the Wilson's phalarope major migratory staging sites are perennial hyper-saline water bodies with an abundant supply of the invertebrate prey species on which Wilson's phalaropes specialize (Jehl 1988, pp. 52-53, Jehl 1999, pp. 37, 39). At these sites, alkali flies and brine flies are primary diet items (Boula 1986, p. 29, Jehl 1988, p. 44, Frank & Conover 2021, p. 30). The presence and great abundance of these flies at Great Salt Lake (hereafter GSL), Lake Abert, and Mono Lake is likely why these sites attract immense numbers of Wilson's phalarope (**Fig. 5;** Jehl 1988, p. 5). The brine fly (*E. gracilis)* is the dominant fly species at GSL, but does not occur at Mono Lake or Lake Abert, whereas the alkali fly (*C. hians)* is dominant at Mono Lake and Lake Abert and occurs in smaller numbers at GSL. Brine shrimp (*Artemia monica* at Mono Lake, *A. franciscana* at the other sites) are also extremely abundant at GSL, Lake Abert, and Mono Lake.

Figure 5: A Wilson's phalarope feeding on abundant alkali flies at Lake Abert. Photo by Ron Larson.

At GSL, brine and alkali flies (which were lumped for analysis) of all life stages were the dominant prey of phalaropes in hyper-saline areas: 100% of individuals sampled had adult brine flies in their gizzards (Frank & Conover 2021, p. 30, 33). Brin shrimp were present in only 1% of Wilson's phalarope upper digestive tracts and 0.7% of gizzards at GSL (n = 146 birds; Frank & Conover 2021, p. 33). Chironomid larvae and *Daphnia* were also often in the stomachs of individuals foraging in Farmington Bay, an area of GSL with relatively lower salinities (Frank & Conover 2021, p. 30).

At Lake Abert, alkali fly adults and pupae were each present in 82% of the stomachs of sampled Wilson's phalaropes, and alkali fly larvae were present in 52% (n = 78 birds; Boula 1986, p. 29). Alkali flies were also by far the dominant prey in terms of percent mass of stomach contents of birds sampled at Lake Abert (Boula 1986, p. 29). Brine shrimp made up 1.6% of the mean aggregate mass of prey in Wilson's phalaropes at Lake Abert, occurring in 10% of sampled birds there (n = 78 birds; Boula 1986, p. 29).

At Mono Lake, alkali flies occurred in 58% of sampled adult female stomachs, and 76% of adult male stomachs (Jehl 1988, p. 41-43). In the same study at Mono Lake, brine shrimp occurred in 75% of sampled female stomachs and 57% of adult male stomachs(Jehl 1988, p. 41).

It is notable how few brine shrimp were recorded in Wilson's phalarope diet, especially at GSL and Lake Abert, considering their great abundance at those sites. A potential bias in all research on Wilson's phalarope diet at staging sites is that diet has been quantified based on stomach-contents analysis, which may under-represent quickly digested prey (Hoenig et al. 2022, p. 1) such as brine shrimp.

Diet at other migratory sites: In a sample of 12 birds collected in September at playa lakes in the Texas High Plains, adult Ditpera flies were the most important prey (71% of the average aggregate prey, 83% frequency of occurrence; Baldassarre and Fischer 1984, p. 222). *Corixidae* (water boatmen), adult *Coleoptera* beetles, and larval *Diptera* and were the next most consumed prey (9%, 9%, and 6% of average aggregate prey, respectively; Baldassarre and Fischer 1984, p. 222). In a sample of 100 stomachs collected from saline lakes in the Southern Great Plains of Texas and New Mexico, Ephydradae flies were the most common prey item in

two different years during summer/fall (July-November; 40% and 21% aggregate dry mass, respectively; Andrei et al. 2009, p. 143). *Artemia* brine shrimp, *Corixidae*, *Chironomidae* midges and *Tipulidae* (crane flies) also made up >10% of aggregate dry mass in at least one of the two years (Andrei et al. 2009, p. 143). Spring diet (sampled March-June) at the same sites over two years was dominated by *Chironomidae* midges (51% and 40% aggregate dry mass in different years; Andrei et al. 2009, p. 143). Other spring prey included *Ceratopogonidae* (biting midges; 31% and 7%), *Artemia* brine shrimp (18% in one year, 0% and unavailable at these sites the other), and *Corixidae* (15% and 4%; Andrei et al. 2009, p. 143). *Artemia* were not always available at these sites, but when they were, phalaropes consumed them in greater proportion than their relative availability in both spring and summer/fall (Andrei et al. 2009, p. 143). In one spring/fall when Artemia were not present, phalaropes disproportionately selected *Ephydra* flies (Andrei et al. 2009, p. 143).

Diet on the wintering grounds: Diet is little studied in South America, but lakes used by large numbers of Wilson's phalaropes in the Andes were all dominated by either calanoid copepods (*Boekella poopoensis*) or brine shrimp (*A. salina;* Hurlbert 1984, p. 47). Chironomid midges (*Paratrichocladius* spp.), and brine flies (*Ephydridae* spp.) were also abundant at sites used by Wilson's phalaropes in the Andes and could be important prey (Hurlbert 1984, p. 47). *Artemia* brine shrimp also occur in many of the sites most frequently used by phalaropes in lowland Argentina, including Laguna Mar Chiquita (Vanhaecke et al. 1987, p. 137).

Diet on the breeding grounds: Diet is also little studied on the breeding grounds. One study of 106 stomachs collected from across the breeding range from May to September found that *Diptera* flies (including alkali, brine flies, and mosquito larvae) were the dominant prey item (43% of total prey; Wetmore 1925, p. 9). Other important prey were aquatic *Heteroptera* insects (24% of total), especially waterboatmen (Corixidae spp.), *Coleoptera* beetles (20% of total), especially aquatic beetles, and *Crustacea* including brine shrimp (4%; Wetmore 1925, p. 9-10).

D. Reproduction

The first birds arrive on the breeding grounds in April through early May (Reynolds et al. 1986, p. 306). Courtship begins during northbound migration and is mostly characterized by female-female aggression as they compete to mate with males (Höhn 1967, p. 240, Kagarise 1979, p. 19, Colwell & Oring 1988, p. 572). After pairs are formed, females select a nest site, usually in vegetation on the edge of a lagoon or in upland areas, but always within 100 m of wetlands (Höhn 1967, p. 240, Colwell & Oring 1990, p. 298). Nests are usually situated in mixed vegetation that is generally taller and denser than that used by other prairie-breeding shorebirds (Colwell & Oring 1990, p. 299). After clutch completion (usually four eggs), females leave incubation of eggs and care of young entirely to the males (**Fig. 6;** Colwell & Oring 1988, p. 575, 577). Females depart breeding grounds and arrive at staging areas in western North America by mid-June (Jehl 1988, p. 36). Males generally arrive at staging areas two or more weeks later than females, because they remain at the breeding site longer to care for the eggs and young (Jehl 1988, p. 36).

Figure 6: Wilson's phalarope nest in short-grass meadow vegetation adjacent to Chaplin Lake, Saskatchewan, Canada. Photo by Ryan Carle.

E. Current and Historic Range

Wilson's phalaropes breed in the interior of the north-central United States and south-central Canada, migrate to staging areas mostly in the Intermountain West of the U.S., and then migrate to wintering sites in western and southern South America (**Fig. 7;** Colwell & Jehl 2020 *Movements and Migration section*).
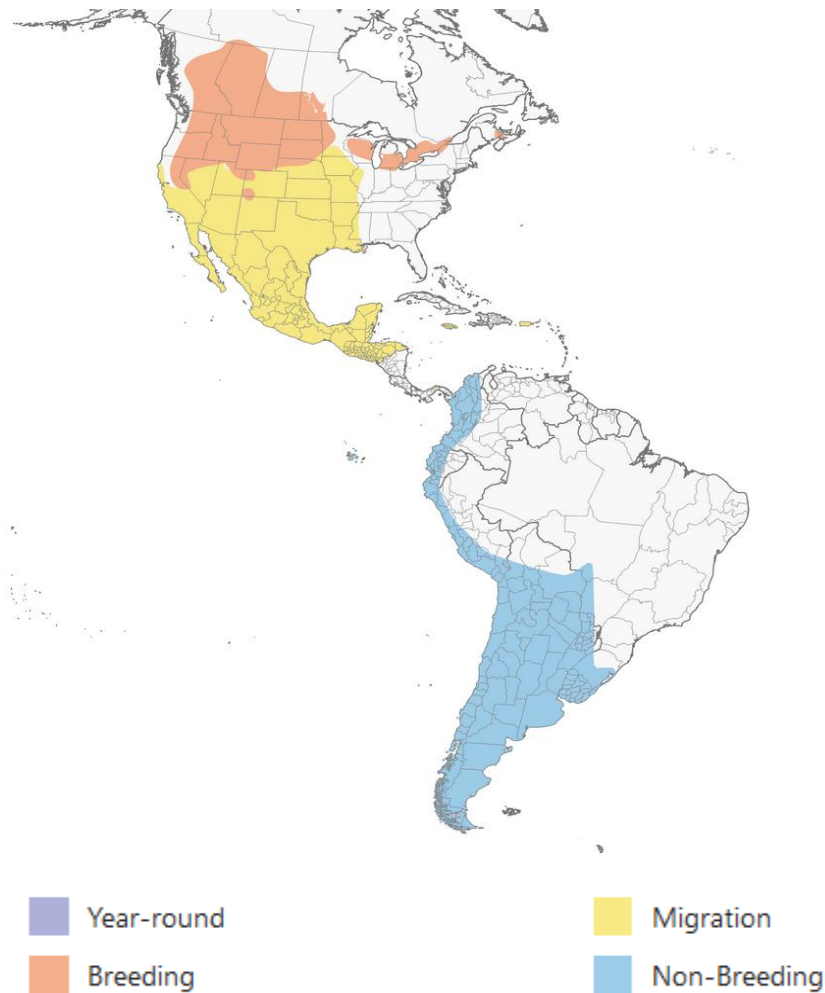
Figure 7: *Distribution of the Wilson's phalarope. Map from* [Colwell and Jehl 2020](#)*.*

Breeding Range

The breeding range of the Wilson's phalarope covers much of interior western North America in areas with prairie and grassland habitat adjacent to wetlands ([Colwell & Jehl 2020](#), *Distribution section*). With the replacement of much of the native prairies of Canada and the U.S. by agriculture, currently the primary breeding habitat of Wilson's phalarope is around shallow water bodies in highly disturbed mixed grass prairies and agricultural areas (Hohn 1967, p. 220, DeGraaf & Rappole 1995, p. 222). In Canada, the species breeds in British Columbia, Alberta, Saskatchewan, Manitoba, and locally in southern Yukon, Ontario, and Quebec (**Fig. 8;** Hohn 1967, p. 220, DeGraaf & Rappole 1995, p. 221, Sibley 2014, p. 194). In the United States, core breeding-range states include Montana, Wyoming, Idaho, North Dakota, South Dakota, Colorado, northern Utah, northern Nevada, north-eastern California, and eastern Oregon (DeGraaf & Rappole 1995, p. 221, Sibley 2014, p. 194, Fig. 1). There are also localized nesting records from Arizona, New Mexico, Nevada, Colorado, central to southern California, Nebraska, Texas, Minnesota, Illinois, Indiana, Kansas, Ohio, Massachusetts, and Alaska (Sutton & Arcilla

2018, p. 74, DeGraaf & Rappole 1995, p. 221, Castellino et al. 2024, p. 7).
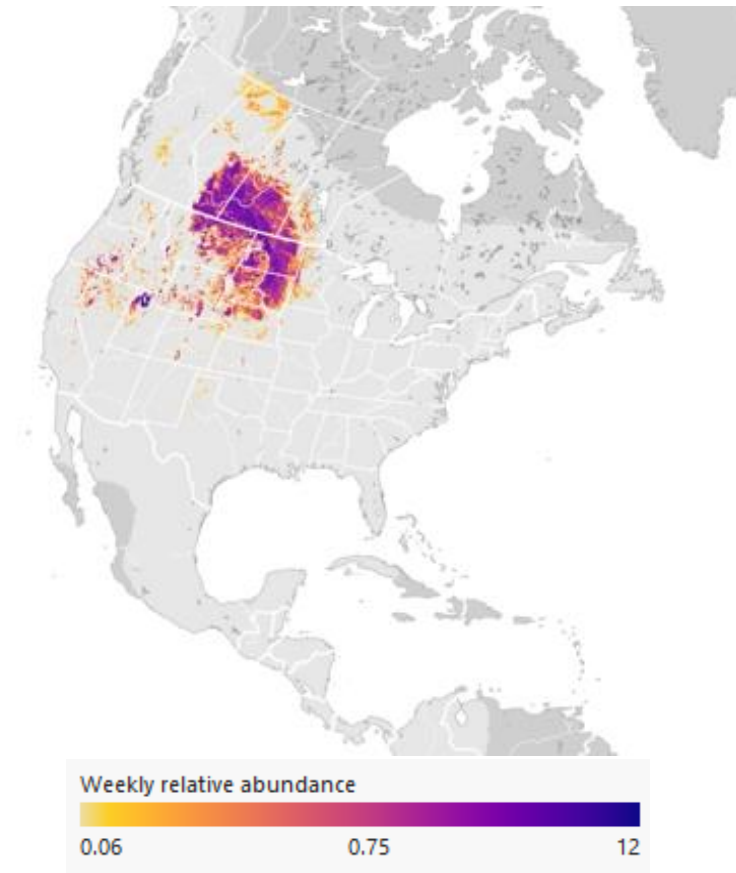
Figure 8: *Average Wilson's phalarope abundance from analysis of eBird data for the week of June 7th, reflecting the species' breeding range. Color scale shows relative abundance across a 3 km × 3 km spatial grid. Relative abundance is defined as the count of individuals of a given species detected by an expert eBirder on a 1 hour, 2 kilometer traveling checklist at the optimal time of day. For example, if relative abundance is 10 in one area and 5 in another area, then abundance is estimated to be twice as high in the first area—thus, dark purple areas reflecting relative abundances of >12 birds (sometimes much greater) have 12x or greater abundances than yellow, orange, or light purple areas with relative abundances <1. Map from eBird 2023.*

A major contraction of the breeding range occurred during the early 20th century, driven by the extensive loss of prairie wetlands (Dahl 1990, p. 1, Colwell & Jehl 2020, *Conservation and Management section*). Destruction of wetland habitat used for breeding caused documented range contractions in Iowa, Minnesota, Indiana, and Missouri (Colwell & Jehl 2020 , *Distribution section*). The species has not recovered in these regions. For example, the species now "rarely" nests in Iowa, where only a few birds engage in nesting behavior each year (Dinsmore & Dinsmore 2023, p. 81). Wilson's phalarope was a widespread and common breeder throughout Minnesota in the 19th century, but the range greatly contracted there by the 1970s and 1980s, to only the northeast of the state (Pfannmuller *et al.* 2017, *entire*). The range has

contracted again since the 1980s and the species is now considered a "rare and sparsely distributed" nester in the state (Pfannmuller *et al.* 2017, *entire*). Accordingly, it has been listed as Threatened by Minnesota (Minnesota Department of Natural Resources 2021, Appendix B, p. 5).

During the 1990s, there was indication of range expansion, with more nests recorded in the periphery of the breeding range (e.g. Alaska, Quebec; Colwell & Jehl 2020, *Distribution section*). However, this may have been driven by wetland loss and/or drought in the core breeding range rather than a population expansion (Colwell & Jehl 2020, *Distribution section*). Since then, BBS data indicates the strongest declining trends in the periphery of the breeding range, suggesting any range expansion during the 1990s has not been maintained (Ziolkowski et al. 2023, *entire*).

Molt Migration

Wilson's phalarope is a long-distance migrant, undertaking annual movements from breeding grounds in central North America to wintering grounds in central and southern South America (Jehl 1988, p. 34). Of the five distinctive migratory flyways in North America (Brown *et al.* (2001, p. 33-41), the species mainly uses the Intermountain West, where the large saline lakes are located (**Fig. 9;** Jehl 1988, p. 50-51, Oring *et al.* 2013, p. 3). In the first stage of south-bound migration after nesting, Wilson's phalaropes carry out a molt-migration, meaning they move to specific locations where they undergo a complete molt of body feathers (Jehl 1988, p. 52). During that time, large concentrations of Wilson's phalaropes are found on large hypersaline lakes in western North America (Jehl 1988, p. 48-51).
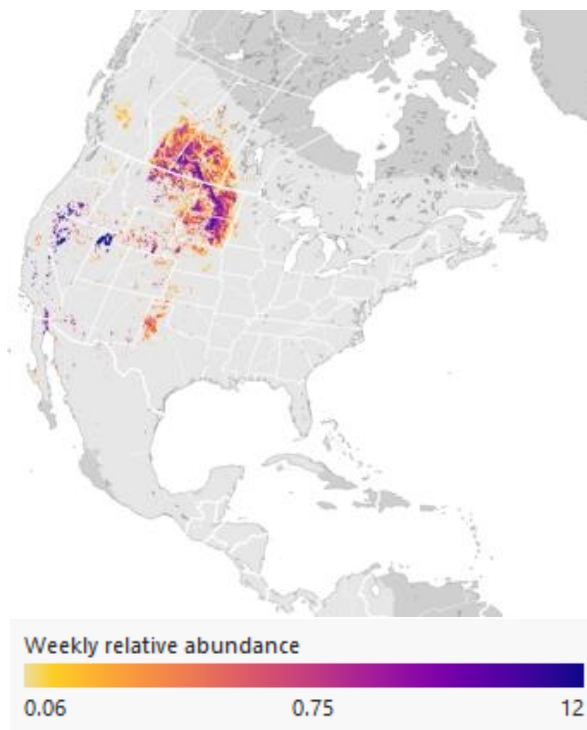
Figure 9: Average Wilson's phalarope abundance from analysis of eBird data for the week of July 12th, reflecting the typical peak molt migration period (Jehl 1988, p. 37, Carle et al. 2023,

p. 18). Relative abundances of Wilson's phalaropes at Great Salt Lake were in the 1,000-2,000 range per grid cell, versus relative abundances shown in yellow, orange and red on this map, which were in the range of <0.05 birds per grid cell—thus, Great Salt Lake's abundances shown here were approximately 20,000-40,000 times those of areas shown in yellow, orange, and red. See Fig. 8 caption for detailed eBird abundance methods. Map and methods from eBird 2023.

These large saline lakes provide abundant invertebrate food resources, especially alkali and brine flies, which enable a rapid complete molt of body feathers and a doubling of body mass prior to a presumed non-stop flight to South America (Jehl 1988, p. 53). Based on surveys covering much of western North America in the 1980s, important molt migration stops are GSL, Lake Abert (Oregon), Mono Lake (California), Lahontan Valley (Nevada), Old Wives' and Chaplin Lakes (Saskatchewan), and San Francisco Bay (California; Jehl 1988, pp. 50-51). Up to 90% of the world's adult Wilson's phalaropes molt/stage in the Intermountain West's hypersaline lakes prior to migrating to South America (**Fig. 10;** Jehl 1988, p. 50-51, Brown et al. 2001, p. 36, Oring *et al*. 2013, p. 3). Smaller numbers of birds (hundreds to low thousands per site) use wetlands in the Central Flyway during fall migration, such as playas and saline lakes in west Texas and New Mexico (Andrei et al. 2006, p. 376), and wetlands in Kansas such as Quivera National Wildlife Refuge (hereafter, NWR) and Cheyenne Bottoms (Parmelee et al. 1969, p. 21, Skagen & Knopf 1994, p. 97, Hands 2008, p. 75).

Figure 10: *Wilson's phalaropes staging on Lake Abert, August 2012. Photo by Ron Larson.*

Southbound migration and wintering distribution

Adults depart from the staging areas from late July to August, and nearly all birds have left by mid-September (Jehl 1988, p. 36). Adults then make a presumed rapid and direct non-stop flight from the staging areas to coastal western South America, crossing the Pacific Ocean (Jehl 1988, p. 52). The theory that birds follow this route is based on the low number of fall records for Central America and northern South America, the exhausted state of birds arriving at coastal sites in Ecuador, departure dates from staging sites, and arrival dates in South America (Jehl 1988, p. 52). The first migrants arrive on the west coast of South America by early August and subsequently move south and east (Jehl 1988, p. 53). Within the wintering region, there are two distinct areas of primary importance: the high Andes (including northern Argentina, Chile, Bolivia, and southern Peru) and the central lowlands of Argentina (Hurlbert 1984, p. 47,

Castellino et al. 2024, p. 11, Fig. 4). The highest abundances in the wintering range have been recorded in Bolivia and Argentina (Castellino et al. 2024, p. 11). Individuals also sometimes reach as far south as Tierra del Fuego in Argentina (**Fig. 11;** Roesler & Imberti 2015, p. 88). Occasionally, birds winter as far north as northern Mexico and the southern U.S. (eBird 2023, *Wilson's phalarope account*).
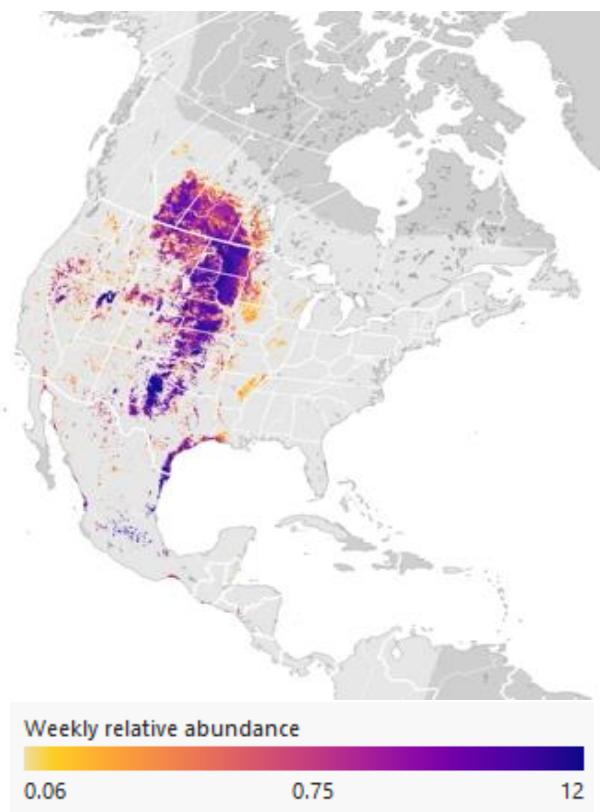

Weekly relative abundance

0.06   0.75   12

*Figure 11: Average Wilson's phalarope abundance from analysis of eBird data for the week of February 7th, reflecting the wintering period in South America. See **Fig. 8** caption for detailed eBird abundance methods. Map and methods from eBird (2023).*

Northbound migration

The return migration of the Wilson's phalarope starts in March, and the first birds arrive on the breeding grounds in April to early May (Reynolds et al. 1986, p. 306). Little is known of the return migration, but birds appear to use interior flyways through South America, continuing overland through Central America and Mexico and/or crossing the Gulf of Mexico (**Fig. 12;** Lesterhuis & Clay 2010, p. 22). After arrival in North America, most migrants appear to pass through the south-central Great Plains and the Intermountain West region to reach the breeding grounds (eBird 2023, *Wilson's phalarope account*). Phalaropes migrating northward stop at sites including a network of playas and saline lakes in northwest Texas and eastern New Mexico (Andrei et al. 2006, p. 376), and wetlands in Kansas (Parmelee et al. 1969, p. 21, Skagen & Knopf 1994, p. 97, Hands 2008, p. 75). Aggregations at these sites are typically smaller than aggregations at fall migration staging sites (Jehl 1988, p. 50-51, Parmelee et al. 1969, p. 21, Skagen & Knopf 1994, p. 97, Hands 2008, p. 75). The large saline lakes in North America are

used less extensively during north-bound than south-bound migration, likely because prey availability is lower in the spring at these sites (for example, Mono Lake, located at nearly 7,000' elevation, is cold and wintry in March-April, with few invertebrates available to phalaropes; Jehl 1988, p. 1, 19, Strauss et al. 2002, p. 228).

Figure 12: Average Wilson's phalarope abundance from analysis of eBird data for the week of February 7th, reflecting the wintering period in South America. See Fig. 8 caption for detailed eBird abundance methods. Map and methods from eBird (2023).

F. Habitat Requirements

Wilson's phalarope is a long-distance migrant, with specific habitat requirements at each stage of its annual cycle. During the breeding season they require prairie, meadow, or agricultural areas adjacent to wetlands, during the molt migration they require large hypersaline lakes with alkali flies and brine flies, and during the wintering period they require saline lagoon habitat in South America. We describe habitat requirements for each of these phases in detail below, starting with the molt migration, because large North American saline lakes are the most threatened habitat of the species, and their loss is the most likely to result in the endangerment of the species.

Molt migration and Staging habitat

From June–August, flocks of tens of thousands to hundreds of thousands of post-breeding Wilson's phalaropes congregate at large saline lakes in western North America (Jehl 1999, p. 40). Peak numbers of Wilson's phalaropes at these sites typically occur in July (Jehl 1988, p. 37, Frank & Conover 2023, p. 408). At these "staging sites," they rapidly replace their body feathers and double their body mass (Jehl 1988, p. 46). Wilson's phalarope is one of only two shorebird species known to have a molt migration (Jehl 1987, p. 187), meaning they make an intermediate migration to molt before continuing to wintering grounds.

At saline lakes, adults molt the entire body plumage, the rectrices, and up to six primaries and associated coverts in 32–40 days (Burger & Howe 1975, p. 447, Jehl 1987, p. 174-175, Jehl 1988, p. 46). This molt takes place at "extraordinary speed," that "may match the minimum duration of any bird species" (Jehl 1988, p. 46). The rapidity of this molt is made possible by the prey available at large saline lakes, primarily alkali and brine flies. Jehl (1988, p. 46) noted, "This rapid molt is made possible by the superabundance of invertebrate prey; at its peak birds can be replacing 60-80% of the feathers in a given tract." Also at staging sites, adults amass fat loads of up to 54% of total body mass; for most shorebirds, fat loads of 45% of body mass is the maximum (Jehl 1997, p. 541). For some individuals, this mass gain results in a period of being too heavy to fly, a phenomenon unknown for other shorebirds species (Jehl 1997, p. 539). This extreme weight gain is necessary to complete the subsequent non-stop migration of approximately 4,000 km to South America, which is near the maximum possible distance within the energetic range predicted for a bird of their size (Jehl 1988, p. 53). As reported by Jehl (1988, p. 54), on describing why Wilson's phalaropes use saline lakes despite the physiological challenges of the saline environment: "…the body molt is intense and completed with great rapidity, evidently owing to their ability to exploit superabundant prey… While staging, ...[they] lay on enormous fat reserves in a very short time. For adult phalaropes, which can double their weight in several weeks, the fat is used to fuel a nonstop flight to South America."

Few sites host the abundance of appropriate invertebrate prey needed for phalaropes to successfully complete the energy-intensive rapid complete molt, doubling of the body mass, and subsequent non-stop migration (Jehl 1999, p. 37). Accordingly, the species is heavily reliant on the three remaining large, perennial hypersaline lakes in western North America that have high abundances of alkali fly, brine fly, and brine shrimp prey: GSL, Lake Abert, and Mono Lake (Conover & Bell 2020, p. 249, Jehl 1999, p. 37). The critical importance of these sites to the species has been described as "likely of importance for species persistence" (Oring & Reed 1997, p. 10), and highlighted by many scientific experts on the species:

> *"In fact, GSL is the most important stopover site in the world for both phalarope species."*
> Conover & Bell (2020), p. 249 *(referring to both Wilson's and red-necked phalaropes)*
>
> *"...the great majority of the adult population appears to participate in a molt migration to one of three hypersaline lakes in the western United States-Mono Lake, CA, GSL, UT, and Lake Abert, OR, where the birds molt and fatten for several weeks in summer to*

prepare for a nonstop flight to South America (Jehl 1988)…Consequently, single censuses at each major locality at this peak of migration in each year have the potential of being able to detect the bulk of the adult population. Jehl (1999), p. 37.

Among staging sites, GSL hosts far and away the largest numbers of Wilson's phalaropes and is the only site that consistently has hundreds of thousands of birds present annually (Jehl 1999, p. 40, Carle *et al*. 2023, p. 11). Great Salt Lake hosts at least one-third of the world population of Wilson's phalaropes annually (Conover & Bell 2020, p. 249-250). This may even be a conservative estimate of the proportion of the world population of the species using GSL. Depending on which world population estimate is used (1.5 million from Jehl 1988 [p. 52] or 1 million from Castellino et al. 2024, [p. 15]), GSL has hosted up to 40%-60% of the world population at once (603,000 individuals; Jehl 1999, p. 40). The saline lake staging sites are primarily used by adults; the peak count at GSL of 603,000 birds represents 81% of the world population of adults (based on Jehl's 1988's [p. 52] world population estimate of 741,000 adults). The only other staging site with a record of >100,000 birds is Lake Abert (Jehl 1999, p. 40). In surveys of six major staging sites from 2019–2022, only GSL, Mono Lake, and Lake Abert consistently had flocks numbering in the tens of thousands (Jehl 1988, pp. 50-51, Jehl 1999, p. 40, Carle et al. 2023, p. 11), though other sites have historically had counts in the tens of thousands prior to significant declines (discussed below).

GSL, Lake Abert, and Mono Lake share features that enable brine flies, alkali flies and brine shrimp to reach large populations sizes and concentrations: 1) the lakes have accumulated dissolved minerals over thousands of years making them hypersaline, so they contain abundant nutrients enabling a high biomass of algae, which provides food for the invertebrates (Larson 2023, p. 47, 99), 2) the high salinity excludes fish, meaning algae, alkali flies, brine flies, and brine shrimp have few competitors or predators other than birds (Larson 2023, p. 95, 100). Alkali and brine flies are also numerous in saline lakes and because they have three, co-occurring and overlapping life stages: larvae, pupae, and adults (Larson 2023, p. 113). Thus, fly biomass and productivity is larger than the number of adult flies would suggest, and all of these life stages are eaten by phalaropes (Larson 2023, p. 114-115). At GSL, Lake Abert, and Mono Lake, alkali flies and/or brine flies can be so numerous during the staging period that the shore looks as if it was covered by tar (**Fig. 13**). At the GSL, brine fly larvae can reach densities of 16,000–25,000/m^2 and pupal cases can have estimated densities of 37,000,000 per mile of shoreline (Wurtsbaugh 2009, p. 5, Utah DNR 2022, *entire*). At Lake Abert, alkali fly adult abundance was estimated at 80,000/m^2 (Swarth 1983, p. 28). Alkali fly larvae and pupae at Mono Lake are also numerous, with densities of 15-30 individuals per square inch (or 23,000-46,000/m^2 if extrapolated) on the undersides of submerged tufa rocks (Dana & Herbst 1977, p. 62).

Figure 13: *Dense aggregations of alkali flies on the shores of Lake Abert. Photo by Ron Larson.*

Brine shrimp are also abundant at all three lakes. Brine shrimp are extremely abundant at GSL, where there were 32 companies commercially harvesting the cysts and that resulted in an annual economic value of $50–100 million (Belovsky et al. 2011, p. 5). Brine shrimp abundance at Lake Abert was estimated at 300 billion adults, with a total biomass of 7 million kg (Conte and Conte 1988, p. 167). At Mono Lake, *Artemia monica* total abundance during mid-summer was estimated to be multiple trillions (Conte et al. 1988, p. 179), with 500,000 pounds of shrimp annually harvested by fishers over a small part of the lake (Dana & Herbst 1977, p. 61).

Alkali flies, brine flies, and brine shrimp are energy-rich food sources, and their high abundance at saline lakes ensures that the tens of thousands to hundreds of thousands of birds stopping there have plenty of food for their immense energy requirements. Alkali and brine flies are generally thought to have a greater lipid content and caloric content per individual than brine shrimp, though there is variation in energy content between sites, shrimp species, and fly life stages (Caudell & Conover 2006, p. 253, Herbst 2023, p. 7). At GSL, the gross energy content and digestibility of brine fly adults was similar to that of *Artemia,* whereas brine fly larvae had significantly less energy concentration than brine shrimp (Caudell & Conover 2006, p. 253). At Lake Abert, alkali fly larvae had more than double the % lipid and total caloric content per individual than *Artemia* (Herbst 2023, p. 7)*.* Alkali fly larvae at Mono Lake also had greater energy than *Artemia*, though not as high as at Lake Abert (Herbst 2023, p. 7). Regardless of these differences, both flies and brine shrimp are highly digestible, energy-rich prey items (Caudell & Conover 2006, p. 253), and Wilsons phalaropes eat both (Jehl 1988, p. 43).

Alkali and brine flies, and brine shrimp, are found in great abundance at only a few large perennial saline lakes that phalaropes rely on (Jehl 1999, p. 39). Jehl (1988) made it clear that the

species requires the habitat provided by large, perennial saline lake staging sites with energy rich invertebrates, which allow them to successfully migrate and molt. As Jehl (1999) stated: "[It] is highly probable [that all major staging sites have been identified], because the permanent, invertebrate rich, hypersaline lake habitat required by staging Wilson's phalaropes is localized and has been confirmed by intensive field work throughout the western United States," (Jehl 1999, p. 39).

Jehl (1994, p. 270) also underscored the importance of large perennial saline lakes habitats, because "back-up" sites providing redundancy have been lost: "The health of bird populations that use unstable habitats is to a large extent dependent on the availability of back-up sites that can be used when conditions change. Unfortunately, there is not much redundancy left in the saline and alkaline lakes of the west." When they are available, smaller numbers of phalaropes visit those "backup sites": habitats across interior western North America including ephemeral saline wetlands, alkaline ponds, and sewage ponds (Jehl 1988, p. 48). Other than GSL, Lake Abert, and Mono Lake, sites that have historically hosted congregations of >10,000 Wilson's phalaropes during the staging period include San Francisco Bay salt ponds (Jehl 1988, p. 48), the Lahontan Wetlands complex (Nevada; Neel & Henry 1996, p. 17, Jehl 1999, p. 40), and a complex of lakes in southern Saskatchewan (Jehl 1988, p. 50). Notably, all of these areas have hyper-saline habitats supporting alkali flies and/or brine shrimp. In annual surveys since 2019, none of these sites have had peak annual counts exceeding the low thousands, except for one count of 14,535 at Lake Chaplin, Saskatchewan in 2022 (Carle et al. 2022, p. 25-26, Carle et al. 2023, p. 18). See the section below with detailed information on numbers and trends at specific staging sites. The loss of more ephemeral "backup sites," and importance of large perennial saline lakes, was underscored by a study that identified surface water declines from 1984–2018 of 27% and 47%, respectively in snowmelt watershed lakes and wetlands in the Intermountain West flyway region (Donnelly et al. 2020, p. 2046). As stated by Donnelly et al. (2020, p. 2054), "Lake and wetland declines we identify in snowmelt watersheds may signal a loss of plasticity in migratory networks."

Within-site habitat use of staging sites has been most extensively studied at GSL (Frank & Conover 2019, p. 1, 2021, p. 538) and Mono Lake (Jehl 1988, p. 39-40). All studies investigating within-site habitat use of Wilson's phalaropes at staging sites have found that birds tend to clump together in groups and associate with distinct habitat features, rather than distribute themselves uniformly across the lake surfaces (Jehl 1988 p. 39-40, Paul & Manning 2002, p. 43, Frank & Conover 2019 p. 548, 2021, p. 8,). Frank & Conover (2021, p. 548) noted that, "On Great Salt Lake, both phalarope species forage in high-density flocks with several birds per square meter, while 99% of the GSL surface area is devoid of phalaropes." At both GSL and Mono Lake, phalarope prefer shallow water areas with rocky substrates (Jehl 1988, p. 39-40, Carle & Rubega 2020, p. 11, Frank & Conover 2021, p. 8-9). These shallow, rocky areas tend to have high abundances of alkali and brine flies because submerged rocks (microbialites at GSL and tufa rocks at Mono Lake) provide substrates for brine fly pupal attachment (**Fig. 14;** Herbst & Bradley 1993, p. 195, Lindsay *et al*. 2019, p. 106). At GSL, phalaropes also occur in relatively lower salinity areas such as Farmington Bay, where they eat more chironomid larvae, corixids, and *Daphnia* (Frank & Conover 2019, p. 8, Frank & Conover 2021, p. 32). Phalaropes also seek out freshwater sources for drinking and bathing at Mono Lake, especially deltas of creek mouths

(Jehl 1988, p. 40). Phalarope habitat use at GSL and Mono Lake has changed in response to habitat changes: Ogden Bay and Bear River Bay at GSL had regularly hosted phalaropes but ceased to do so when those sites dried up in 2015, and phalaropes concentrated into a narrow remaining strip of water in Farmington Bay as its water level dropped that year (Paul & Manning 2002, p. 35, Frank & Conover 2021 p. 7, 11). The North Arm of GSL once hosted flocks of hundreds of thousands of phalaropes but became too saline to support brine shrimp and flies after railroad causeways were constructed in 1959 (White et al. 2015, p. 2, 13). Despite making up a large proportion of GSL, the North Arm has been rarely used by phalaropes since it became too saline for invertebrates (Frank & Conover 2021, p. 11). At Mono Lake, Wilson's phalaropes aggregated in shallow, rocky areas in 2019 and 2020 that had been dry during surveys in the 1980s and 1990s due to water diversions (Carle & Rubega 2020, p. 11).

There is evidence that different demographic groups have both different timing and select different staging sites. The major saline lake staging sites are used primarily by adults, whereas juveniles occur at those sites in smaller numbers and use a wider variety of habitats (Jehl 1988, p. 53). Males and females also use staging sites in different proportions (Jehl 1988, p. 47-51). In the 1980s, Mono Lake and Lake Abert had flocks disproportionately composed of females and males, respectively (Jehl 1988, p. 47, 51), and Jehl (1988, p. 51) theorized that adult females also predominate at GSL. Jehl also theorized that adults tend to stage at their first stopping point, rather than stopping at multiple staging areas (Jehl 1988, p. 52). Juveniles migrate later than adults, do not amass such large fat reserves, and use the saline lake staging sites in smaller numbers (i.e., hundreds to low thousands; Jehl 1988, p. 53).

Figure 14: *Wilson's phalaropes preening in their preferred habitat at Mono Lake—shallow areas with rocky substrates and an abundance of alkali flies. Photo by David Carle.*

Wintering habitat

The main non-breeding habitats used by Wilson's phalarope in the wintering region are ephemeral or permanent saline lagoons in the high Andes and lowland Argentina (Hurlburt 1984, p. 50, Castellino et al. 2024, p. 11-12). Some of the largest congregations of Wilson's phalaropes are found in Argentina (Castellino et al. 2024, p. 11). Laguna Mar Chiquita, a large saline lake in central Argentina (Córdoba province), is considered the most important non-breeding site for Wilson's phalarope due the species' high abundances there (**Fig. 15;** Castellino et al. 2024, p. 12). At Laguna Mar Chiquita, ≥400,000 birds are recorded simultaneously during most austral summers (Castellino et al. 2024, p. 12).

Figure 15: Laguna Mar Chiquita, Argentina. Photo by Ryan Carle.

In northwest Argentina another important site for the species is the Monumento Natural Laguna de los Pozuelos (Jujuy province; Castellino et al. 2024, p. 12). The largest flocks recorded at this site were 500,000 birds in January 1988 and 449,108 birds in February 2021 (Castellino et al. 2024, p. 12). Another important site in Argentina is Lago Epecuén (Buenos Aires province), a hypersaline lake that has had up to 140,000 Wilson's phalaropes and regularly has flocks of tens of thousands of birds (Castellino et al. 2024, p. 12). Laguna del Palar, a shallow lagoon that is part of the Ramsar site Lagunas de Vilama, also had 143,649 birds in February 2020, but only 418 birds were present in February 2021 (Castellino et al. 2024, p. 12). The Salinas Grandes, an ephemeral lake near Laguna Mar Chiquita, has records of Wilson's phalaropes ranging from 7,300-20,000 birds (Castellino et al. 2024, p. 12). Wilson's phalaropes also occur throughout the Patagonia region of Argentina. Although abundances are lower than in

northern Argentina, unpublished records and eBird data indicate counts of 5,000-20,000 birds at some Patagonian sites (Castellino et al. 2024, p. 12).

During the 1970s, the Bolivian altiplano was considered the core part of the species' wintering range, based on counts of 500,000–1,000,000 birds across that region (Hurlbert *et al.* 1984, p. 51). Large congregations of Wilson's phalaropes have occurred in the southern part of Bolivia (Potosí Department), with counts of ≥100,000 from Laguna Loromayu, Laguna Hedionda Norte, Laguna Calina, and Laguna Pastos Grandes in the late 1970s and early 1980s (Hurlbert *et al.* 1984, p. 50). However, in recent years, numbers in these lakes have been much lower, with maximum counts not exceeding the low tens of thousands since 2010, and not exceeding 10,000 since 2020 (**Table 3;** Castellino et al. 2024, p. 11, 18, 22).

Wilson's phalarope is a regular visitor to the altiplano of Chile, though the numbers are usually lower than in neighboring areas of Argentina and Bolivia (Hurlburt et al. 1984, p. 55). The highest counts in Chile are from Salar de Surire in the 15th Region, where there were 13,590 birds in February 2020 and 50,000 in the 1990s (**Fig. 16;** Castellino et al. 2024, p. 12). Wilson's phalaropes also use the Patagonian region of Chile, often in scattered small groups but sometimes in flocks of >1,000 birds (Castellino et al. 2024, p. 12).

Figure 16: *Wilson's phalaropes foraging alongside flamingos at the Salar de Surire in Chile in February 2020. Photo by Rozy Bathrick.*

In Ecuador, the artificial salt lakes of Ecuasal, located along the southern coast, are an important stopover site (Castellino et al. 2024, p. 10-11). Numbers of Wilson's phalaropes at Ecuasal peak between August and September, with numbers in the range of 15,000–44,300 individuals (Castellino et al. 2024, p. 11). Wilson's phalaropes also occur at some sites in Peru and Paraguay (Castellino et al. 2024, p. 11).

Northbound migration

During northbound migration, Wilson's phalarope can be found in inland marshes, flooded fields, salt works, sewage-ponds, and coastal lagoons and estuaries in Mexico and the central and western U.S. (Castellino et al. 2024, p. 12-13, eBird 2023, *Wilson's phalarope account*). The saline lake sites used for staging during fall migration do not host similarly large concentrations of phalaropes during the spring migration (Jehl 1988, p. 55). This is likely due to lower food availability at those lakes during that time of year, when brine shrimp and flies are mostly not in their adult stages (Jehl 1988, p. 19). However, northbound migrating birds use smaller wetlands located in northwest Texas and eastern New Mexico, where they preferentially used saline lakes over freshwater habitats, and preyed on brine shrimp and alkali flies (Andrei et al. 2006, p. 381, Andrei et al. 2009, p. 143). Northbound phalaropes also use historically large wetlands in the Great Plains including Cheyenne Bottoms and Quivera NWR in Kansas (Parmelee et al. 1969, p. 21, Skagen & Knopf 1994, p. 97, Hands 2008, p. 75). Wetland extent of both Cheyenne Bottoms and Quivera NWR have been greatly reduced due to aquifer depletion and water diversion (Aber et al. 2016, p. 33, Llopes-Jensen 2023, *entire*). In north-central Oklahoma, phalarope abundance was significantly positively associated with wetlands at the 2,000 m scale, indicating that the species prefers the largest wetland habitats available in that region (Elmore et al. 2023, p. 6). Large congregations of northbound birds have also been recorded in several wetlands in Mexico, with flocks of up to 42,000 birds observed at Lago Texcoco and 60,000 at Bahía Santa María (Castellino et al. 2024, p. 12).

Breeding habitat

Wilson's phalaropes nest semi-colonially at shallow ponds and lakes, ranging from fresh to hyper-saline, and preferably close to wet-meadow vegetation (**Fig. 17;** Hohn 1967, p. 221, Bomberger 1984, p. 128). However, the species also may nest in swales along streams, shallow sloughs fringed with short grasses, and hay meadows or pastures up to 100 m from water (Hohn 1967, p. 221, Colwell and Oring 1990, p. 298). Additional breeding habitats include taiga interspersed with moist, grassy muskeg, and aspen-grove parklands (Hohn 1967, p. 220). Of 269 nests found by Kantrud and Stewart (1984, p. 429) in North Dakota, 30% were near semi-permanent ponds, 60% were near seasonal wetlands, and 7% were near alkali ponds or other types of wetlands. Despite being known as a semi-colonial breeder, nest densities vary greatly and probably depend to some extent on habitat availability and quality, with greater densities on high quality habitat (Kagarise 1979, p. 17, Kantrud & Stewart 1984, p. 430).

Figure 17: *Phalarope nesting habitat at Chaplin Lake, Saskatchewan, characterized by short-grass prairie adjacent to wetlands. There were approximately 7 nests found in the area shown in this photo. Photo by Ryan Carle.*

III. POPULATION STATUS AND TREND

A. Population Estimates

From the 1980s to present, the most frequently used world population estimate for the Wilson's phalarope has been 1,500,000 individuals. This estimate originated from Jehl (1988, p. 52), based on a total of 741,000 birds (mostly adults) counted during July 1986 at major migratory staging areas in western North America. Based on an assumption that the number of juveniles, which migrate later, was similar to the number of adults observed in July, the July 1986 count total was multiplied by two and rounded up to 1.5 million (Jehl 1988, p. 52).

Subsequent reports of Wilson's phalarope global population estimates (e.g., Morrison et al. 2006, p. 80, Andres et al. 2012, p. 188, Lesterhuis & Clay 2010, p. 13), used Jehl (1988, p. 52)'s 1.5 million number, based on an absence of new evidence, and because it is somewhat similar to the sum of fall migration estimates of populations for North American flyways from other sources (approximately 900,000 birds across the Interior, Pacific, and Eastern flyways; Morrison et al. 2001, p. 48). Most authors have listed the 1.5 million estimate as of low accuracy (Morrison et al. 2001, p. 48, Morrison et al. 2006, p. 80).

In 2020, a survey was completed across the Wilson's phalarope non-breeding range in South America, with the goal of obtaining a new world population estimate (Castellino et al. 2024, p. 15). Surveys were coordinated across Peru, Chile, Bolivia, and Argentina during a ten-day window in early February (Castellino et al. 2024, p. 15). The non-breeding grounds should include all demographic groups, including juveniles (Castellino et al. 2024, p. 15). The total number of birds directly counted in the non-breeding range was 854,673, but based on coverage

calculations and error estimations, it was suggested to set the estimate of the Wilson's phalarope population at 1 million individuals (Castellino et al. 2024, p. 15). Whether this estimate of 1 million birds represents a decrease from the 1.5 million estimated in 1986 (Jehl 1988, p. 52) is unclear, given the differences in methodology and areas covered by the two surveys (Castellino et al. 2024, p. 15).

B. Population-wide Trends

The Wilson's phalarope world population is believed to have undergone a significant historical decline during the early 20th century due to habitat loss with prairie conversion to farmland in North America (Lesterhuis & Clay 2010, p. 31). The first data available on populations are from the 1970s and 1980s, offering a baseline to compare with contemporary surveys. Overall, studies converge on an indication that Wilson's phalarope populations declined sharply range-wide between the 1980s and 1990s, and have not recovered to 1980s levels. In some areas, populations appear to have declined further since the 1990s. From the 1980s to present, a declines of >70% total population was indicated by analyses of International Shorebird Survey (ISS) surveys (Smith et al. 2023, p. 9). Surveys at major North American staging sites showed a similar pattern, with about half as many birds registered in the 1990s compared to the 1980s (Jehl 1999, p. 40-41). GSL, Lake Abert, and Mono Lake surveys from 2019–2022 showed major declines compared to 1980s average high counts, and relatively similar flock sizes to the 1990s (Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18). At several other staging sites of hosting tens of thousands of birds during the 1980s, numbers have decreased drastically, including a 98% decline in phalaropes at San Francisco Bay salt ponds (LaBarbera et al. 2023, p. 25), and high counts only in the hundreds at the Lahontan wetlands in recent years (Senner et al. 2021, p. 93).

Breeding Bird Survey (BBS) data likewise indicated a declining trend in the species, with largest declines in the 1990s (Ziolkowski et al. 2023, *entire,* **Fig. 5**). Data are more limited for South America, but at several sites in Bolivia that had 100,000 or more birds in 1979, high counts have not exceeded the low tens of thousands since 1999 (**Table 3;** Castellino et al. 2024, p. 18, 22). Major sites in Argentina, such as Laguna Mar Chiquita and Laguna de Los Pozuelos, have had highly variable numbers, with some recent years' counts similar to historic highs (around half a million at each site), but many other years with counts an order of magnitude lower (**Table 3;** Castellino et al. 2024, p. 18-19). Sites in Chile and Peru also have had substantially smaller numbers in recent years than in the 1970s and 1980s (**Table 3**; Castellino et al. 2024, p. 11-12).

The recent analysis of International Shorebird Survey data (ISS) by USFWS and Canadian Wildlife Service authors found an approximately 70% decline in abundance of the species in the U.S. and Canada since 1980 (Smith et al. 2023, p. 7). That study evaluated trends over 3-generation time periods within the overall study period (1980–2019). The 3-generation time period for Wilson's phalaropes was estimated to be 10 years (Smith et al. 2023, p. 5). In the most recent 3-generation period (2009-2019), that analysis found Wilson's phalarope populations were stable, but large declines occurred between 1980–2000 (**Fig. 18;** Smith et al. 2023, p. 8-9, supplemental materials p. 14, Figure S5).
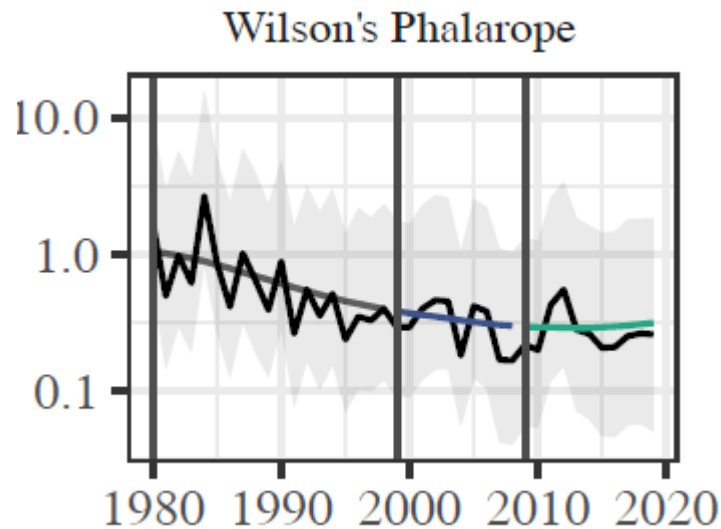
Figure 18: *Figure from Smith et al. 2023 (supplementary material p. 14, Fig. S5), showing smoothed and unsmoothed population trajectories (displayed as estimated mean count per survey on a log scale) for Wilson's phalarope, 1980−2019. The most recent three−generation trends, the previous three−generation trends, and any prior years are separated by vertical bars, and displayed in different colors. Smith et al.'s (2023) analysis indicated an approximately 70% decline in Wilson's phalaropes across the breeding range from 1980–2019.*

Based on the Smith et al. (2023) analyses, Wilson's phalarope may qualify for IUCN Red List "Endangered" status under the criteria of a ≥50% decline over any 10-year or 3-generation period (IUCN 2012, p. 18–19). Smith *et al* (2023) considered their results robust because of their analytical methods, although coverage was relatively limited in the middle and western parts of the North American continent, where Wilson's phalaropes more often occur (Smith *et al.* 2023, p. 11). Nevertheless, the population decrease occurring in the 1980s and early 1990s observed by Smith *et al*. (2023, p. 9) was consistent with an observed major decline in numbers at important staging sites in the 1990s (specifically Great Salt Lake, Lake Abert, and Mono Lake; Jehl 1999, p. 41). Those staging sites had combined annual high-count totals in the 400,000–600,000 range in the 1980s and in the 100,000–300,000 range in the 1990s (Jehl 1999, p. 40). Likewise, Breeding Bird Survey results from 1980-1990 indicated a declining trend during that the 1980s and 1990s (**Fig. 18;** Ziolkowski et al. 2023, *entire*).
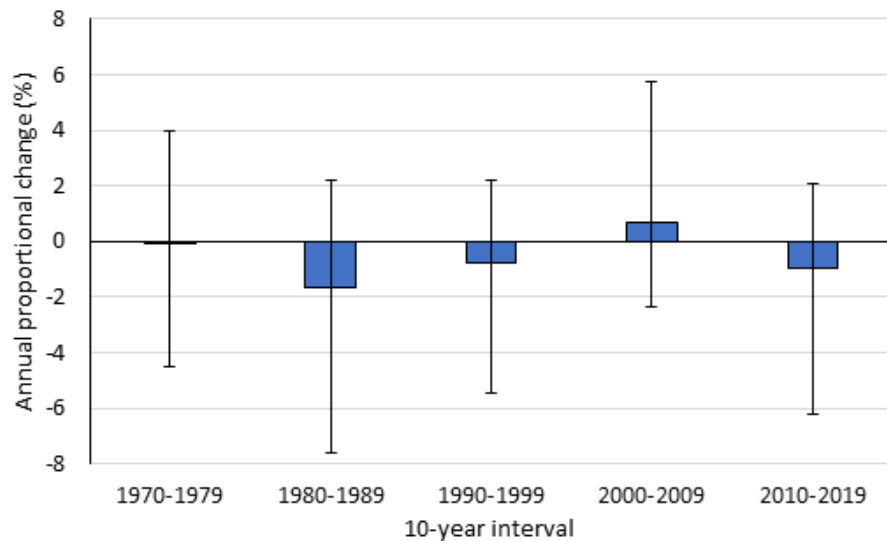
Figure 19: *BBS trends for Wilson's phalaropes over 10-year (3-generation) intervals. Trend is defined as yearly geometric mean of proportional changes in population size, expressed as a percentage. Error bars are upper and lower 95% CI. Trends shown are from results analysis are from [Ziolkowski et al. 2023](#), entire, available at: [https://www.mbr-pwrc.usgs.gov/](#). Figure from Castellino et al. 2024, p. 16.*

The Breeding Bird Survey (BBS) provides a means of looking at Wilson's phalarope trends based on long-term standardized surveys across North America during the breeding season ([Ziolkowski et al. 2023](#), *entire*). In BBS analysis, trends are defined as a yearly geometric mean of proportional changes in population size, expressed as a percentage, where a trend of zero indicates no annual change in proportional population size (Sauer & Link 2011, p. 91). BBS results reported below are all from USGS analyzed results portal ([Ziolkowski et al. 2023](#), *entire*). Analysis of BBS data from the entire time-series (1966–2019) indicated a moderately declining survey-wide trend for Wilson's phalarope (-0.58% mean change per year; **Fig. 19; Table 1**). Survey-wide trends for 3-generation periods (approximately 10 years for Wilson's phalarope; Smith et al. 2023, p. 5), were stable in the 1970s, negative in the 1980s and 1990s, positive from 2000-2009, and negative from 2010-2019 (**Fig. 19**). Within specific BBS Bird Conservation Regions, there were strong declining trends (mean % annual change >-1.0) for the entire time-series for the Shortgrass Prairie, Great Basin, Northern Rockies, Boreal Taiga Plains, and Southern Rockies/Colorado Plateau regions, and stable trends (mean annual % change within 0.1 of zero) for the Prairie Potholes and Badlands and Prairies regions (**Table 1**). There were insufficient data to calculate reliable results for other regions. Large-scale surveys like BBS and ISS are best suited for surveying breeding areas of Wilson's phalarope. In staging areas like Great Salt Lake or Mono Lake, birds often swim >1 km from shore and surveying them requires specialized methods (Jehl 1999, p. 41). Migratory staging areas are best suited for focused boat- or plane-based surveys and provide a good opportunity to survey the population when it is spatially concentrated (Jehl 1999, p. 41).

Table 1: Wilson's phalarope population trends from the Breeding Bird Survey, 1966–2019. Negative trends >0.1 are bolded. Trend is defined as yearly geometric mean of proportional changes in population size, expressed as a percentage. Credibility ratings are defined as blue = data with at least 14 samples in the long term, of moderate precision, and of moderate abundance on routes, yellow = data with a deficiency because either regional abundance is <0.1 birds per route, long-term sample is based on <14 routes, results are too imprecise to detect a long-term 3%/year change. Results shown here are from those reported by Ziolkowski et al. 2023, *entire,* available at: https://www.mbr-pwrc.usgs.gov/.

Survey region	Trend (annual % change)	95% CI lower	95% CI upper	Credibility	N survey routes
Survey-wide	**-0.58**	-1.61	0.37	Blue	625
U.S-wide	**-0.94**	-2.10	0.12	Blue	409
Canada-wide	0.06	-1.78	1.73	Blue	216
Prairie Potholes	-0.10	-1.48	1.18	Blue	259
Badlands and Prairies	0.08	-1.92	2.18	Blue	80
Northern Rockies	**-2.39**	-4.44	-0.38	Yellow	75
Boreal Taiga Plains	**-3.22**	-6.70	0.12	Yellow	28
Great Basin	**-2.21**	-3.97	-0.42	Yellow	103
Southern Rockies/Colorado Plateau	**-3.47**	-6.45	-0.48	Yellow	25
Shortgrass Prairie	**-1.23**	-5.58	2.10	Yellow	31
Central Mixed-grass Prairie	0.15	-3.17	3.86	Yellow	13

The U.S. Shorebird Conservation Plan scored Wilson's phalaropes as "highly imperiled" based on their declining population trend, and a "species of high conservation concern" overall (Brown et al. 2001, p. 59). The proposed action for Wilson's phalaropes in that plan was to "halt declines, then restoration to 1972 levels," with a recovery goal to grow the world population from its estimated level of 1.5 million birds to a target of 2.8 million (Brown et al. 2001, p. 54). USFWS (2004, p. 2) identified Wilson's phalaropes as a species of "High Concern" in their 2004 list of High Priority Shorebirds, based on the scores used in Brown et al. (2001, p. 54). A 2012 assessment led by USFWS reported an "apparent long-term decline" for the species, with more stability over the short-term (Andres et al. 2012, p. 188).

North American staging sites trends: In phalarope-focused surveys at North American staging sites (specifically GSL, Lake Abert, and Mono Lake), there was a major decline in numbers in the 1990s (combined totals in the 100,000–300,000 range) compared with the 1980s (totals in the 400,000–600,000 range; Jehl 1999, p. 40). Since 2019, surveys have been coordinated at GSL, Mono Lake, Lake Abert, San Francisco Bay, Chaplin Lake, and Owens Lake, with surveys every two weeks from July to September (Carle et al. 2023, p. 2). Totals used here from those surveys are of same-day sums of Wilson's and unidentified phalaropes during the week-long survey window with the highest count of Wilson's (Carle et al. 2023, p. 10). High counts from those surveys provide a comparison with Jehl's (1988, p. 50-51) surveys across staging sites in 1986-1987 (**Table 2**). Jehl's (1988, p. 50-51) combined total in July 1986 from GSL, Mono Lake, and Lake Abert was 455,135 birds (**Table 2**). Coordinated surveys during 2019–2022 at those three sites yielded an average peak count of 305,263 birds (Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18), a 33% decrease compared with Jehl's (1988, p. 50-51) totals. Compared with Jehl's (1999, p. 40) multi-year surveys at those three sites, the average 2019–2022 peak counts across the sites were 42% lower than 1980s averages (**Table 2**, Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18). Surveys at GSL, Mono Lake, Lake Abert, Chaplin Lake, and San Francisco Bay registered 41% fewer birds (average peak totals 2019–2022; Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18) compared with 1986–1987 counts (Jehl 1988, p. 50-51; **Table 2**). Owens Lake was dry in the 1980s, which is why it is not included in these comparisons.

Table 2: Comparisons of historic and contemporary Wilson's phalarope high counts at staging sites.

Site	1986-1987 high count[1]	1980s average high count[2]	1990s average high count[2]	Average 2019-2022 high count[3]	% change 1986-87 and 2019-2022 average	**% change, 1980s and 2019-2022 averages**	% change 1990s and 2019-2022 averages
GSL	387,000	395,500	319,728	254,089	-34%	**-36%**	-21%
Mono Lake[4]	56,320	52,280	22,582	19,386	Survey methods not comparable, see text	Survey methods not comparable, see text	Survey methods not comparable, see text
Lake Abert	11,825	74,766	17,325	31,788	+168%	**-57%**	+83%
SF Bay	40,000	No data	No data	798	-98%	No data	No data
Chaplin Lake	35,000	No data	No data	5,411	-84%	No data	No data
GSL, Mono, and Abert total	455,135	522,046	359,662	305,263	-33%	**-42%**	-15%
Total of all 5 sites	530,145	No data	No data	311,472	-41%	No data	No data

[1]Jehl 1988, p. 50-51, [2]Jehl 1999, p. 40, [3]Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18, [4]Note that 2019-present Mono Lake surveys covered approximately 20% of the lake surface, vs. historic counts that attempted to census the entire population of phalaropes on the whole lake

Methodological and coverage details of contemporary and historical surveys differed, however contemporary surveys have shown consistently lower numbers than those reported by Jehl (1986, p. 48-51). Methodology used in historic and contemporary surveys at GSL and Lake Abert were similar, albeit historic survey methodologies were not well documented. Contemporary surveys at Mono Lake cover approximately 20% of the lake surface, with standardized site selection stratified toward areas expected to have phalaropes present (Carle et al. 2023, p. 6). Thus, contemporary Mono Lake data cannot be directly compared with Jehl's counts, which attempted to count all birds on the lake. If Jehl's (1999, p. 40) average 1980s count from Mono Lake (52,280) were used in place of the contemporary numbers when

summing 2019–2022 totals from GSL, Abert, and Mono, the combined totals from those sites would still be 35% lower than average 1980s total from Jehl (1999, p. 40). Generally, at specific staging sites, 2019-2021 peak numbers were substantially lower than in the 1980s, and similar to the lowest-count years in the 1990s (**Table 2;** Jehl 1999, p 50-51, Carle et al. 2022, p. 26-27).

C. Trends at specific sites

GSL: In all studies on phalarope staging sites and migration, GSL stands out as hosting by far the largest flock sizes of Wilson's phalaropes in North America (Jehl 1988, p. 50-51, Jehl 1999, p. 40, Conover & Bell 2020, p. 249, Carle et al. 2023, p. 10). The peak count of 603,000 Wilson's phalaropes at GSL in 1991 (Jehl 1999, p. 40) represented ~81% of the adult world population and 40% of the entire world population, based on Jehl's (1988, p. 52) world population estimates. Based on an updated world population estimate of 1 million birds total (Castellino et al. 2024, p. 15), GSL's peak count represents 60% of the entire world population. In surveys from 1982–1997, the number of Wilson's phalaropes at GSL ranged from 118,000-603,000, with an average of 336,566 (SD 170,102, 9 years with complete survey coverage). Even the lowest annual peak counts at GSL exceed the highest peak counts at any other site (with the exception of one year with possibly 150,000 birds at Lake Abert; Jehl 1999, p. 40, Carle et al. 2023, p. 10). Numbers declined by 55% at GSL during the 1994–1997 period (average 214,025, SD 72,825, 4 years) compared to the 1982–1991 period (average 493,250, SD 113,508, 4 years). During 2019–2022, average Wilson's phalarope counts at GSL were 254,089 (SD 73,948), ranging from 337,698 in 2019 to 165,705 in 2022 (Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 10). The 2019–2022 average high count was 48% lower than the 1980–1991 average for GSL (Jehl 1999, p. 40). It is noteworthy that 2022 had one of the lowest GSL peak counts on record (165,705 birds; only 1994 had a lower peak count, at 118,000), coinciding with negative ecosystem impacts of the drying lake.

Lake Abert: Lake Abert is a shallow, fishless, hyper-saline lake located in southeastern Oregon is one of the most important staging sites for the species (Jehl 1999, p. 37). The highest published annual peak count of phalaropes (probably composed mainly of Wilson's) at Lake Abert was 230,000 birds in 2013 (Larson et al. 2016, p. 397). That count represented 31% of the world adult population, or 15–23% of the entire world population (based on either Jehl 1988's (p. 52) world estimate of 1.5 million or the 2020 estimate of 1 million; Castellino et al. 2024, p. 15). In 2012 and 2013, there were high counts from formal surveys at Lake Abert of 214,000 (August 3rd) and 230,000 (July 24th) unidentified phalaropes, respectively (Larson et al. 2016, p. 397). Though these flocks were not identified to species, based on the dates, and photo documentation on eBird, it is likely the flocks were composed primarily of Wilson's phalaropes (Jehl 1999, p. 38 estimated 75% of late July phalaropes at Lake Abert to be Wilson's). A credible eBird record, also from July 24th, 2013, estimated 330,000 Wilson's phalaropes, accompanied with photo documentation (**Fig. 20;** Hinkle 2013, *entire*). That count represented 22–33% of the world population (based on either Jehl 1988's [p. 52] world estimate of 1.5 million or the 2020 estimate of 1 million, Castellino et al. 2024, p. 15).
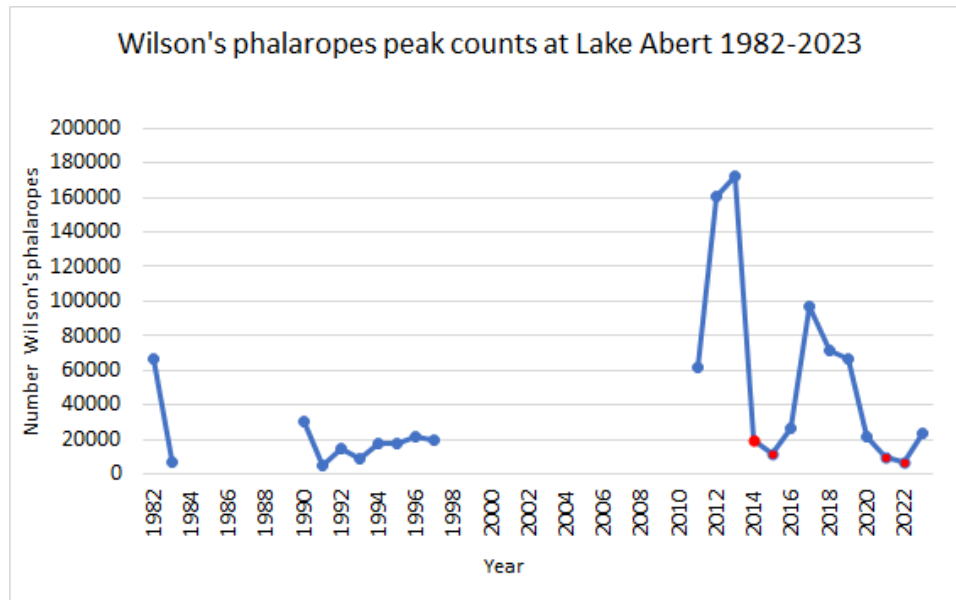
Figure 20: *Photo of Wilson's phalaropes at Lake Abert on July 24th, 2013, from* *Hinkle 2013.*

The next highest peak annual count at Lake Abert was 65,000–70,000 in 1982, which was also considered a minimum (Jehl 1999, p. 40). In the period from 1982–1997, the average annual peak count was 36,061 (SD 17,770, *n* = 10 years), ranging from 7,000–67,000 (Jehl 1999, p. 40). No data are available from 1998–2011. From 2011-2022, East Cascades Audubon Society (ECAS) began twice monthly waterbird surveys at Lake Abert, May to October (Senner et al. 2018, p. 2, methods described in Carle et al. 2023, p. 6). From 2011–2022, average peak counts were 35,840 birds (SD 34,333), ranging from 6,500–115,000 birds (*n* =11 years; Larson et al. 2016, p. 397, Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18). Comparing high counts from the 1980s (Jehl 1988, p. 50-51) to those from 2019–2022 (Wilson's and unidentified combined on the same day, during dates before August; Carle et al. 2022, p. Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18), there was a 58% decrease at Lake Abert (**Table 3**).

Notably, Wilson's phalarope numbers at Lake Abert declined during periods when the lake became nearly completely desiccated in 2014–2015 and 2021–2022 (**Fig. 21;** Senner et al. 2018, p. 326, Carle et al. 2023, p. 10-11). An analysis using phalarope counts and lake conditions from the 1990s to 2016 found that at lower lake levels phalarope abundance was positively correlated with lake area (Senner et al. 2018, p. 324). During 2014–2015, when Lake Abert was desiccated, Wilson's phalarope counts averaged 15,873 birds (SD 5,697) compared with 131,470 (SD 60,969) from 2011–2013 (**Fig. 21**). Lake levels rose and numbers of phalarope recovered somewhat in 2016–2019, but in 2021 and 2022 the lake once again desiccated, reaching an area of <1 mile^2 (Hall et al. 2023, p. 4). Phalarope abundance again decreased, with a mean abundance of phalaropes in 2021 and 2022 of 8,177 (SD=2,272; **Fig. 21**). In 2023, inflows brought the lake up to 40 miles^2 and the abundances of phalaropes increased, with a peak of 31,190 (Oregon High Desert Association unpublished data). Thus some recovery of phalarope

numbers occurred after desiccation events, but with numbers still an order of magnitude below those seen in 2011–2012 (**Fig. 21**).

Figure 21: Peak Wilson's phalaropes counts at Lake Abert 1982-2023. Only complete counts are shown. When annual high counts were not identified to species, flocks were estimated to be 100% Wilson's phalaropes if the count occurred before July 15, and 75% Wilson's phalaropes if the count occurred between July 16–August 15 (based on Jehl 1999, p. 38). Lake desiccation events in 2014–2015 and 2021–2022 are shown in red. Data are from Jehl 1999, p. 40, Larson et al. 2016, p. 397, Carle et al. 2022, p. 25-27, and Carle et al. 2023, p. 18, and Oregon High Desert Association unpublished data (2023 only).

Mono Lake: Mono Lake is a terminal hyper-saline lake in the western Great Basin that is fish-less and has an abundance of brine shrimp and alkali flies (Winkler 1977, p. 2). Jehl (1999, p. 37) identified Mono Lake one of the three most important Wilson's phalarope staging areas in North America (along with GSL and Lake Abert). The highest published estimate of Wilson's phalarope numbers at Mono Lake is 93,000 (single survey, Winkler 1977, p. 100), followed by 77,950 by Jehl (1988, p. 37-38; based on full-season counts of different demographic groups and turn-over rates). Annual lake-wide peak counts from 1980–1990 averaged 49,590 birds (SD 14,497; Jehl 1999, p. 40). The 1990–1997 average annual peak counts from the same survey method was 22,566 birds (SD 11,543; Jehl 1999, p. 40), representing a decline by over half from 1980s totals (**Fig. 22**). After Jehl's (1999, p. 40) work, no published survey data are available until 2019–2022, in which a survey sampling approximately 20% of the lake with point counts registered an average annual high-count of 19,002 Wilson's phalaropes (SD 18,192) with an annual high count range of 3,165–45,143 birds (Wilson's phalarope and unidentified phalarope spp. totals combined from the peak count day of Wilson's; Carle et al. 2022, p. 25-27, Carle et al.

2023, p. 18).

Peak annual counts of Wilson's Phalaropes at Mono Lake 1976-2023
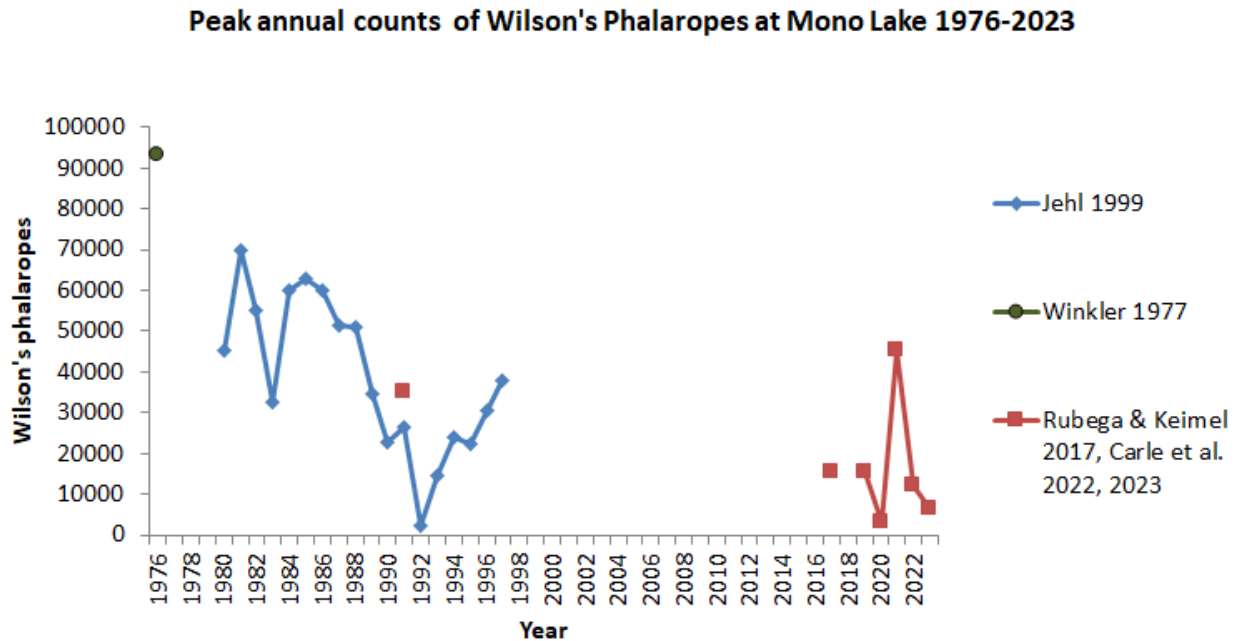
Figure 22: Peak annual counts of Wilson's phalaropes at Mono Lake from 1976-2023. Data are from Winkler 1977, p. 2, Jehl 1999, p. 40, Rubega & Keimel 2017, p. 13, Carle et al. 2022, p. 25-27, Carle et al. 2023, p. 18, and Oikonos unpublished data (2023 only). Surveys with different methods are shown in different colors. Coverage of each survey method differed substantially. Rubega & Keimel and Carle et al. surveys covered approximately 20% of the lake at standardized count stations.

The Carle et al. (2022, 2023) surveys are not directly comparable to the Jehl (1999) surveys because of methodological differences but followed a similar protocol and sampling effort to a 1991 survey that registered 35,225 Wilson's phalaropes (Rubega & Keimel 2017, p. 13). Based on Jehl's (1999, p. 40) results, there was a major decline of phalarope numbers at Mono Lake from the 1980s to the 1990s and currently numbers have not recovered to 1980s levels. In coordinated surveys of phalaropes across six major staging sites in 2020-2022, Mono Lake had the second or third largest high counts of all sites (exceeded only by GSL each year and Lake Chaplin in 2022), despite the survey there not completely covering the whole lake (Carle et al. 2022, p. 26-27, 2023, p. 18). A large count of Wilson's phalarope at Mono lake in 2021 (45,361 birds) may have been related to the desiccation of Lake Abert that year—far fewer birds were seen at Abert when it dried up and they may have moved to Mono Lake (Carle et al. 2021, p. 16-17). The relatively large counts at Mono Lake compared to smaller sites, and the use of Mono Lake when other sites are dry, underscores its importance as a staging site for the species.

The Lahontan complex of wetlands in central Nevada (which include the Carson Sink and Stillwater National Wildlife Refuge) historically hosted tens of thousands of staging Wilson's phalaropes, comparable with numbers at Mono Lake and Lake Abert (Neel & Henry

1996, p. 17, Jehl 1999, p. 40). The highest counts for Wilson's phalaropes at Lahontan Wetlands are 67,000 in 1987 and 40,000 in 1953 (Neel & Henry 1996, p. 17). From 1986–1990 the site hosted ≥10,000 Wilson's phalaropes annually (Jehl 1999, p. 40). From 1991–1997, high counts were only in the hundreds or low thousands, with a high of 6,200 birds in 1996 (Jehl 1999, p. 40, Shuford et al. 2002, p. 165). From 1986–2019, the median Wilson's phalarope high count for fall migration was only 295 birds, with a maximum of 13,776 (31 years with data, 8 years with counts during Wilson's phalarope peak timing in July; counts in July-September were classified as "fall"; Senner et al. 2021, p. 105). Thus, there appears to have been a major decline in phalarope use of the site between the 1980s (tens of thousands of birds) to the 1990s (thousands of birds), with numbers not recovering since (typically hundreds of birds since the 2000s; Neel & Henry 1996, p. 17, Senner et al. 2021, p. 93). A pattern noted at the site during surveys from 1989–1996 was that overall shorebird abundance tracked wetland extent (Shuford e al. 2002, p. 148). Notably, the counts of tens of thousands of phalaropes in the 1986–1990 period at the Lahontan Wetlands occurred in wet years during the 1980s, and the lower counts in the 1990s occurred during region-wide drought. In 1992, for example, the area recorded an all-time low (as of then) area of wetlands, and fewer than 800 Wilson's phalaropes were present (Neel & Henry 1996, p. 18, Jehl 1988, p. 40).

In southern Saskatchewan, there are a number of saline lakes that have hosted significant numbers of staging phalaropes. Lakes in southern Saskatchewan are mostly saline (>3 g/L of salt), <10 m deep, receive most of their water from snowmelt rather than rainfall, and experience high evapotranspiration levels (Hammer & Haynes 1978, p. 183, Hammer et al. 1990, p. 321). The most important of these lakes to phalaropes, based on numbers registered, are Chaplin Lake, Reed Lake, and Old Wives Lake (hereafter referred to as the Chaplin complex; Jehl 1988, p. 50). The Chaplin complex (spanning 42,000 ha) is ranked as Globally Important for shorebirds under Canada's Important Bird Areas program, and as a WHSRN site, for its importance to shorebirds (Schmutz 2000, p. 26, WHSRN 2023, *entire*). The Chaplin complex are intermittent saline lakes, and Chaplin and Old Wives are <3 m deep (Reed Lake's depth is unknown; Hammer & Haynes 1978, p. 192). Chaplin Lake is the second largest saline inland water body in Canada (WHSRN 2023, *entire*). The Chaplin complex contains aquatic invertebrates that phalaropes feed on, including brine shrimp *Artemia salina* (Schmutz 2000, p. 23).

Wilson's phalaropes breed in this region, and it is one of the first staging areas where migrating flocks appear in the late summer (Jehl 1988, p. 52). Wilson's phalarope high counts and dates for these lakes are the following (maximum recorded count in parentheses): Old Wives Lake (51,500), Chaplin Lake (35,000), Reed Lake (25,000 birds), East Coteau Lake (6,600), Big Quill Lake (3,000), and Last Mountain Lake (1,500, all high counts were in 1986, Jehl 1988, p. 50). Flocks at these sites in June are usually composed of adult females, and later flocks are composed mainly of adult males (Jehl 1988, p. 52). Surveys other than Jehl (1988, p. 50) have found lower numbers at these lakes, ranging in the hundreds to thousands. Surveys of Chaplin Lake in July 1993 and June 1994 recorded 4,500 and 7,100 Wilson's phalaropes, respectively (Beyersbergen & Duncan 2007, p. 2). Fall 1994 surveys at the Chaplin complex recorded a high of 6,192 Wilson's phalaropes (July, Beyersbergen & Duncan 2007, p. 30). Surveys in 2020 and 2021 of Chaplin Lake registered highs of 1,008 (Wilson's and unidentified phalaropes, July), and 14,535, respectively (Wilson's, July, Carle et al. 2022, p. 25-26, Carle et al. 2023, p. 18).

Manitou, Reflex, Luck, and Lenoire Lakes, located in this region, all had <100 Wilson's phalaropes during multiple surveys in 2021 and 2022, except Luck Lake, which had 2,500 birds (Wilson's and unidentified phalaropes, July 2021, K. Caruso, unpublished data). Thus, data are limited, but at least at Chaplin Lake contemporary numbers are less than half of the 1986 high count.

Salt ponds in southern San Francisco Bay are used by Wilson's phalaropes as a staging site during fall migration (Jehl 1988, p. 50). Jehl (1988, p. 50) reported a peak count of 40,000 from south San Francisco Bay in 1986. All sites included in the Jehl (1988, p. 50) study were at salt ponds or commercial salt works, and observations from San Francisco Bay were therefore presumably from commercial salt ponds. South San Francisco Bay historically approached Mono Lake in numbers of staging Wilson's phalaropes, supporting up to 5% of the adult world population (Jehl 1988, p. 50). Surveys of southern San Francisco Bay were carried out from June–September in 2021–2022, with partial pilot surveys of some sites conducted in 2019–2020 (Burns & Van Schmidt 2023, p. 4, Carle et al. 2023, p. 7). The peak count of Wilson's phalaropes during a single survey in that period was 735 individuals in 2022 (Carle et al. 2023, p. 18), indicating a 98% decline within the region compared to the 1986 count (Jehl 1988, p. 50. These declines were based on a limited number of recent survey years, but were corroborated by an analysis of eBird data from 1973-2022 (LaBarbera et al. 2023, p. 3), and a statistical analysis to model numbers for years with no data (Burns & Van Schmidt 2023, p. 12). The results showed strong agreement with the magnitude of decline observed in the scientific surveys, and that this decline exceeded the rate elsewhere in California (**Fig. 23;** Burns & Van Schmidt 2023, p. 2, LaBarbera et al. 2023, p. 21).

The drivers of this severe decline remain unclear. The decline in eBird counts mostly occurred prior to 2000, indicating that restoration of former salt ponds beginning in 2003 was not the primary driver of the historical decline (LaBarbera et al. 2023, p. 26). However, 2022 counts have declined another 77% since 2005–2007, when non-targeted waterbird surveys were conducted (Burns & Van Schmidt 2023, p. 14). That targeted surveys have detected fewer phalaropes than non-targeted surveys underscores the robustness of this result (Burns & Van Schmidt 2023, p. 2). This decrease largely occurred from 2007–2009, shortly after salinity was first reduced in hypersaline ponds to transition them away from salt production, but before most tidal marsh restoration had occurred (Burns & Van Schmidt 2023, p. 3).
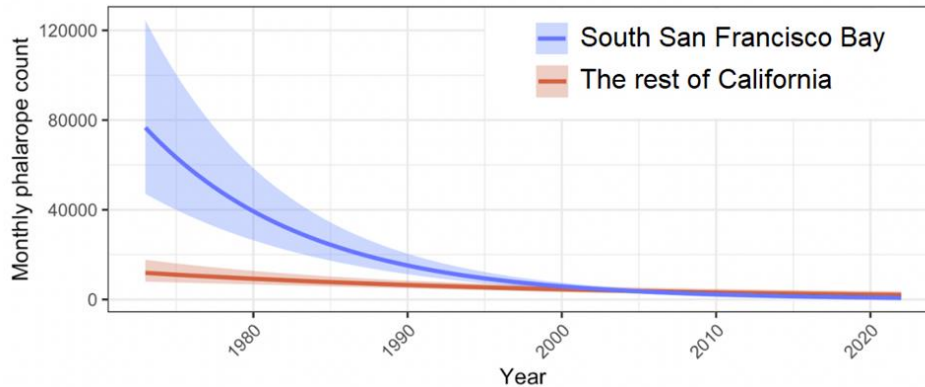
Figure 23: *Trends from eBird monthly counts/checklist of phalaropes within South San Francisco Bay (blue) versus the rest of California (i.e., primarily Mono Lake; red). Note that because surveys in San Francisco Bay are done by levees, they have higher detectability than elsewhere in California (including Mono Lake); only the relative trends should be compared, not the total counts. Adapted from LaBarbera et al. (2023, p. 21).*

In addition to the sites we have discussed, several other sites were identified by Jehl 1988 as having >10,000 Wilson's phalarope present in June-July of either 1986 or 1987 (p. 50-51). Since then, none of these sites have had standardized bird surveys that we are aware of. To check whether these sites still host large numbers of phalaropes, we reviewed eBird data for each site by 1) looking at high count records for Wilson's phalaropes for each eBird hotspot associated with the site (up to March 2023), and 2) looking at species-specific records for those areas to get a sense of the regular magnitude of flocks being reported. eBird data collection has many biases, including non-standardized counts, spatial biases associated with easy vantage points, untrained observer error issues, and lack of detail about coverage. However, these were the best available data and we presumed that taken as a whole they may reflect the orders of magnitude of flocks regularly present at these sites.

Benton Lake NWR, Montana had 20,000 Wilson's phalaropes present in June 1986 (Jehl 1988, p. 50). The highest eBird count for the species was 1,500 in July 2012 (Barnes 2012, *entire*). Only the year 2012 had reports of >1,000 birds. There were regular reports of flock sizes in the low hundreds of Wilson's phalaropes in other years.

Big Lake, Montana had 40,000 Wilson's phalaropes in July 1986 (Jehl 1988, p. 50). The high count on eBird for the species was 2,500 in June 2022 (Huggins 2022, *entire*) which was the only record of >1,000 Wilson's phalaropes. There were regularly records of flock sizes in the low hundreds.

Bowdoin NWR/Medicine Lake NWR, Montana had 28,000–33,000 Wilson's phalaropes in August 1987 (Jehl 1988, p. 50). The high count on eBird at Bowdoin NWR was 13,826 in 1999 (Dinsmore 1999, *entire*). Bowdoin NWR had regular records of >1,000 birds, including in recent years. Medicine Lake NWR's high count on eBird was 1,000 in 2022 (Fickle 2022, *entire*). There were regular records of Wilson's phalaropes in the low hundreds or dozens.

Horsehead Lake ND had 13,500 Wilson's phalaropes in July 1987 (Jehl 1988, p. 51). The high count on eBird was 250 in May 2018 (Neumann 2021, *entire*). Fall numbers on eBird were typically in the low hundreds or dozens.

45

The San Diego Bay salt ponds had 5,000-10,000 Wilson's phalaropes in July 1986 (Jehl 1988, p. 51). The largest record on eBird was 1,600 during June 2021 (Sadowski 2021, *entire*). Numbers were regularly in the low hundreds.

Together, eBird numbers suggest that flocks of the magnitude reported in Jehl 1988 (p. 50-51) no longer regularly occur at virtually all historically important sites. Only Bowdoin NWR had numbers in the same order of magnitude reported by Jehl (1988, p. 50-51). eBird numbers should be interpreted cautiously because of the aforementioned issues, but these results suggest that while Wilson's phalaropes continue to use these sites, the numbers of birds they are attracting is far fewer than those currently seen regularly at GSL, Mono Lake, and Lake Abert.

Several sites not surveyed by Jehl that are in the Central Flyway also have had significant numbers of fall- and spring-migrating Wilson's phalaropes, including playa and saline lakes in Texas and New Mexico, and wetlands in Kansas. The complex of playas and saline lakes in Texas and New Mexico had fall totals of 2,058 and 16,746 birds in two years, and spring totals of 652 and 3,138 birds (Andrei et al., p. 376). Cheyenne Bottoms, which once regularly had spring Wilson's phalarope counts in the tens of thousands, has had dwindling numbers with counts not exceeding 200 since 2020 (**Table 3;** Castellino et al. 2024, p. 20), likely related to drastically declining water levels there (Zimmerman 1990, p. 34).

Wintering range trends: Few data are available from non-breeding areas outside of the North American staging sites for an analysis of trends, but there is some information from specific sites in the two main non-breeding regions (Castellino et al. 2024, p. 18-19). In the Bolivian altiplano, there was a major decline in the 2000s compared with the 1980s (Castellino et al. 2024, p. 18). Four sites in Bolivia that had historical records of abundances ≥100,000 individuals in 1979 have had orders of magnitude lower counts in surveys since 1999 (**Table 3;** Hurlbert et al. 1984, p. 50, Castellino et al. 2024, p. 18, 22). Counts from 1999–2021 (n = 9 years) for the species at sites that had previous counts of ≥100,000 individuals in 1979 yielded the following abundance ranges (min-max): Laguna Pastos Grandes 0-205; Laguna Hedionda Norte 0-20,000; Laguna Kalina 16-30,000 (Castellino et al. 2024, p. 18). Laguna Chulluncani, which had 25,000 birds in 1979 had a range of 0-407 birds from 1999–2021 (Hurlbert et al. 1984, p. 50, Castellino et al. 2024, p. 18). Other potential phalarope sites in the region had similar ranges: Laguna Colorada 7-7,153, Laguna Kollpa 0-149 birds (Castellino et al. 2024, p. 18).

Periodic surveys have occurred since the 1990s at Laguna Mar Chiquita, Argentina. Aerial survey methods are used at Laguna Mar Chiquita due to the lake's large size (**Fig. 24**). From 2010 on, there are abundance data for each austral summer except during 2013 and 2016 (Castellino et al. 2024, p. 19). Abundances during this period ranged from a minimum of 6,000 (March 2018) to a maximum of 613,640 individuals (February 2021; Castellino et al. 2024, p. 19). Since 2019 there have been efforts at Laguna Mar Chiquita to monitor the species monthly, to better understand fluctuations in abundances. Unlike at other sites, contemporary totals at Laguna Mar Chiquita are relatively similar to historic high counts there, the largest of which was 500,000 (**Table 3;** Castellino et al. 2024, p. 19, 23).

Figure 24: Large flocks of phalaropes on Laguna Mar Chiquita in November 2022. Smaller dots are Wilson's phalaropes, larger are flamingos. Photo by Yanina Druetta.

At Laguna de Los Pozuelos, Argentina, the historic high count for Wilson's phalaropes was 500,000 birds in January 1988 (Castellino et al. 2024, p. 12). More recently, systematic surveys were conducted annually from 2006–2021 in February and July (Castellino et al. 2024, p. 18). During 8 of 16 seasons the abundance of phalaropes was below 1% of the global population estimation (i.e., 10,000; Castellino et al. 2024, p. 18). During the other 7 seasons it was below 4% of the global population estimation (i.e., 40,000; Castellino et al. 2024, p. 18). However, in February 2021 an estimated 449,108 Wilson´s phalaropes were observed at the site (Castellino et al. 2024, p. 18), representing almost 45% of the 2020 global population estimation. Though the reasons for this increase are unexplained, the 2021 numbers were much closer to historic counts than any other year since 2006 (Castellino et al. 2024, p. 18).

A handful of other sites with large historic counts also show decreases in abundance in more recent years. The Salar de Surire in Chile had 50,000 birds in the 1990s, and 13,590 birds in 2020 (**Table 3;** Castellino et al. 2024, p. 22). In Peru, Lago Junín had "tens of thousands" of birds in a count in 1979, 308 in 2014, and zero in 2020 (**Table 3**; Dineson et al. 2019, p. 90, Castellino et al. 2024, p. 21).

It is important to note that it is complex to assess population trends for the species in South America based on data from individual sites because habitat changes dynamically, with phalaropes' use of individual sites depending on seasonal habitat conditions (Castellino et al. 2024, p. 18). However, the pattern across sites in the wintering range indicates a major decline in numbers at most sites since the 1970s and 1980s (**Table 3**).

Table 3: Sites with counts of at least 30,000 Wilson's phalaropes (3% of the world population of 1 million; Castellino et al. 2024 p. 15). High count columns show the highest count in each period, followed in parentheses by the month and year of the count and the number of years in each period with counts in italics and bold. Where dates or number of count years are not listed, that information was unavailable. Table modified from Castellino et al. 2024, p. 19-23.

Country	Site	State/Province	High count prior to 2010	High count since 2010	High count since 2020	Sources (historic, since 2010, and since 2020, respectively)
Canada	Chaplin, Old Wives Reed Lakes	Saskatchewan	35,000 (Jul 1986)	7,036 (Jul 2018, *5*)[a]	14,535 (Jul 2022, *3*)[a, b]	Jehl 1988, A. McKellar unpublished data, K. Caruso unpublished data
	Coteau Lake	Saskatchewan	30,000 (Jul 1986)	No data	No data	Jehl 1988
USA	Great Salt Lake	Utah	603,000 (Jul 1991)	337,698[c] (Jul 2019, *4*)	122,850[d] (July 2020, *3*)	Jehl 1988, Carle et al. 2021, Carle et al. 2021
	Mono Lake	California	93,000 (Jul 1976)	45,143[e] (Jul 2021, *4*)	45,143[e] (Jul 2021, *3*)	Winkler 1977, Carle et al. 2021, Carle et al. 2021
	Lake Abert	Oregon	67,000 (1982)	230,000[f] (Jul 2013, *10*)	21,830 (Jul 2020, *3*)	Jehl 1999, Carle et al. 2021, Carle et al. 2021
	Lahontan Valley	Nevada	67,000	No data	No data	Neel and and Henry 1986
	Cheyenne Bottoms	Kansas	52,184 (1990)	26,835 (2014, *8*)	191	Robert Penner, *in litt,* Robert Penner, *in litt*
	Big Lake	Montana	40,000 (Jul 1986)	No data	No data	Jehl 1988
	San Francisco Bay	California	40,000	1,988 (July 2014)	767 (July 2020, *3*)	Jehl 1988, SFBBO unpublished data, Burns et al. 2023
	Moss Landing	California	350,000	No data	No data	ISS
	Bowdoin NWR	Montana	28,000-33,000 (Jul 1986)	No data	No data	Jehl 1988
	Walker Lake	Nevada	No data	100,000 (Sep 2016, *1*)	No data	R. Lowry, *in litt*

Mexico	Bahía de Santa María	Sinaloa	60,000	No data	No data	Vega et al 2006.
	Lago de Texcoco	Mexico	42,274 (May 2006)	No data	No data	WHSRN 2007
	Sistema Lagunar Ceuta	Sinaloa	15,000	877 (Aug 2015, *8*)	No data	Vega et al 2006., Medardeo Cruz Lopez *pers. comm.*
Ecuador	Piscinas artificiales de Ecuasal - Pacoa	Santa Elena	32,000 (Aug 1995)	42,525 (Aug 2016)	17100 (Sept 2022)	B. Haase *pers. comm*, A. Agreda *pers comm*, B Haase, *pers. comm.*
Perú	Lago Junín	Junín	*Tens of thousands* (Oct 1979)	308 (Feb 2014, *2*)	0 (Feb 2020, *2*)	Dinesen et al. 2019, Dinesen et al. 2019, WHSRN 2020).
Bolivia	Laguna Pastos Grandes	Potosí	±100,000	58 (Feb 2020, *10*)	58 (Feb 2010, *10*)	Blanco & Canevari 1998, WHRSN 2020, WHSRN 2020.
	Laguna Loromayu	Potosí	±100,000 (Feb 1979)	7,833 (Feb 2021, *1*)	7,833 (Feb 2021, *1*)	Hurlbert et al. 1984, Omar Rocha *pers. comm.*
	Lago Hedionda Norte	Potosí	±100,000 (Feb 1979)	20,000 (Apr 2012, *10*)	3,302 (Feb 2020, *2*)	Hurlbert et al 1984, Omar Rocha *pers. comm.* WHSRN 2020.
	Laguna Kalina (Busch)	Potosí	±100,000 (Feb 1979)	30,000 (Apr 2012, *10*)	8,812 (Feb 2020, *2*)	Hurlbert et al 1984, Omar Rocha *pers. comm.* WHSRN 2020.
Chile	Salar de Surire	15th Región	50,000	13,590 (Feb 2020, *2*)	13,590 (Feb 2020, *2*)	Bech & Brendstrup-Hansen 1992, WHSRN 2020, WHSRN 2020.
Argentina	Laguna Mar Chiquita	Córdoba	500,000	613,640 (Feb 2021, *11*)	613,640 (Feb 2021, *11*)	Scott & Carbonell 1986, Aves Argentinas, unpublished data, Aves Argentinas, unpublished data.

	Laguna de Pozuelos	Jujuy	500,000 (Jan 1988)	449,108 (Feb 2021, *16*)	449,108 (Feb 2021, *2*)	Moschione & San Cristóbal unpublished data, WHSRN 2021, WHSRN 2021.
	Lago Epecuén	Buenos Aires	No data	140,000 (Sep 2016, *6*)	120,000 (Feb 2020, *2*)	M. Castellino unpublished data, WHSRN 2020.
	Laguna del Palar	Jujuy	No data	143,649 (Feb 2020, *2*)	143,649 (Feb 2020, *2*)	WHSRN 2020, WHSRN 2020

a: counts of only Chaplin Lake

b: same day unidentified phalarope count was 1,200 individuals.

c: the number of unidentified phalaropes on the same date was 2,386 individuals.

d: the number of unidentified phalaropes on the same date was 107,990 individuals.

e: standardized point counts survey covering approximately 20% of the lake surface; same day unidentified phalarope total was 245.

f: recorded as "unidentified phalaropes," but July 24 date suggests likelihood of a high proportion of Wilson's.

IV. THREATS

A. Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range

Two of the three staging sites identified as the most important for the species by Jehl (1999, p. 40) were in danger of drying up or were already dry in fall 2022. Present and threatened habitat destruction, modification and curtailment are the norm for Wilson's phalaropes, not the exception. The threat of ecosystem collapse at North American saline lakes is an imminent near-term reality. These declines are primarily being driven by upstream diversion of freshwater inputs for consumptive use, rather than by climate variation (Wurtsbaugh et al. 2017, p. 1).

- In 2022, GSL, which holds the largest staging concentrations of the species, reached a historically low water level, reached salinities unhealthy for brine shrimp and flies, (USGS 2023, *entire*) and had widespread exposure of microbialites resulting in lost fly habitat (Frantz et al. 2023, p. 1).
- Virtually all habitat at Lake Abert was lost during both 2014–2015 and 2021–2022 when the lake became dry except for inches of water around a few freshwater springs in the lakebed (Larson et al. 2016, p. 389, Hall et al. 2023, p. 13).
- Mono Lake has the best protection of its water inputs of all these sites; however due to continuing regulated water diversion, the lake remains far below target management lake levels (McQuilkin 2023, p. 5, Hall et al. 2023, p. 16). The lake is susceptible to habitat loss and increasing salinity if the lake level decreases during extreme, long-term droughts

that are projected for the region during the 21st century (Hall et al. 2023, p. 16, Finklin et al. 2013, p. 111).

- Phalarope habitat in the Lahontan Wetlands in Nevada has declined as the site has received less water, with lake habitat shrinking by 90% in the Carson Sink since the 1980s (Donnelly et al. 2020, p. 2049). Phalarope numbers have declined by an order of magnitude there (Senner et al. 2021, p. 93).
- The majority of phalarope habitat in San Francisco Bay has been lost as salt ponds have been restored to salt marsh, and phalarope numbers have declined by 98% there since the 1980s (LaBarbera et al. 2023, p. 28).
- Smaller, more ephemeral wetlands in the Great Basin used by phalaropes are susceptible to water diversion and drought, eliminating back-up options to the large perennial saline lakes for staging phalaropes. As the climate is projected to become hotter and drier, hydrologic resilience of North American bird flyways is decreasing (Haig et al. 2019, p. 4, Donnelly et al. 2020, p. 2049).
- South American wintering habitat is threatened, especially in the altiplano region of the Andes (Gutierrez et al. 2022, p. 1, Marconi et al. 2022, p. 840). Rapid development of lithium mining, which uses large quantities of groundwater, has already impacted the region and is expected to continue to deplete aquifers (Martin et al. 2017, p. 171, Garcés & Alvarez 2020, p. 193). This, coupled with drying from climate change (Neukom et al. 2015, p. 8), puts saline lagoons of this entire region at risk of complete desiccation, a parallel to the situation in North American staging habitat.

We will discuss the threats to each site in detail, however, put together as a complete picture, the habitat during the staging part of the annual cycle of phalaropes is severely threatened by water diversion and climate change. Serious losses have already occurred at GSL, Lake Abert, San Francisco Bay, the Lahontan Wetlands, and the South American Altiplano, and greater losses are projected to occur in the foreseeable future. Below, we describe the status and threats to phalaropes at each major staging site and in the wintering range, and we discuss the mechanisms of how potential impacts of how current habitat loss is affecting the species and how predicted future habitat loss will affect the species.

1.   **Factors affecting Wilson's phalarope habitat utilization and persistence**

   a.   *The impacts of increasing salinity on saline lake invertebrates*

As stated by Herbst (2023, p. 2), "Halophytic organisms often do not so much love salt as they have evolved means to tolerate it." As saline lakes' water levels decline, various factors negatively impact the invertebrate prey of phalaropes, including: 1) increased salinities, 2) loss of submerged rocky substrates that provide habitat for alkali and brine fly reproduction, and 3) warming of shallower water causing reduced dissolved-oxygen. Although alkali flies, brine flies, and brine shrimp are evolved for hyper-saline environments, as saline lakes shrink the water can become too saline for even these species to tolerate (Herbst 2023, p. 5, Dana & Lenz 1986, p. 433, Larson 2023, p. 107–108). Consequences of increased salinity occur on a gradient: alkali

and brine flies and brine shrimp can tolerate elevated salinities, but experience increasing negative impacts on growth, reproduction, and survival as salinities increase (Herbst 2023, p. 5, Marden et al. 2020, p. 198). Past their maximum salinity tolerances, mortality and population crashes of flies and brine shrimp can occur (Dana & Lenz 1986, Marden et al. 2020, p. 198, p. 433 Herbst 2023, p. 5). Physiological stress from high salinities is exacerbated by lower dissolved-oxygen levels at high salinities (Sherwood et al. 1991, p. 27, Barnes & Wurtsbaugh 2015, p. 9), which has been the apparent cause of observed brine shrimp die-offs at Lake Abert (Larson 2023, p. 107-108).

Alkali and brine flies: Alkali flies (*C. hyans)* occur at Mono Lake and Lake Abert, and in lower salinity areas of GSL (**Fig. 25;** Herbst 1988, p. 146, Herbst 1999, p. 132). Brine flies (*E. gracilis*) are the dominant fly at GSL and are adapted to its chloride-rich waters (Herbst 1999, p. 133), and do not occur at Mono Lake or Lake Abert. For both fly species, increasing salinity at saline lakes increases the costs of osmoregulation at the expense of other developmental and reproductive needs (Dana & Herbst 1977, p. 68-69, Herbst & Bradley 1993, p. 198, Herbst 2023, p. 2).

Figure 25: Alkali flies feeding at Lake Abert, Oregon. Photo by Ron Larson.

Alkali flies: Herbst (2023, p. 1) found, based on extensive experimentation and field observation, that "the cost of osmoregulation to fitness from rising salinity was exhibited in slower growth rates of larvae, smaller size at maturity of pupae, reduced adult emergence success, and lower fecundity." Specifically, third-instar (the third stage of development between molts) fly larvae exposed to a range of salinity levels of Mono Lake water (10, 50, 100, and 150 g/L) showed slower average growth with increasing salinity, with significant decreases in growth rate at ≥100 g/L (Herbst 2023, p. 4). Third-instar larvae exposed to higher salinities had slower development time, did not achieve similar sizes to those exposed to less saline water, and had lower adult emergence success (Herbst 2023, p. 4). Experimental results showing these impacts of salinity were corroborated by field observations comparing adults and pupae at Lake Abert and Mono Lake (Herbst 2023, p. 7). At the time of observation, Lake Abert's salinity was 25 g/L and Mono Lake's was 90 g/L; both adults and pupae were significantly larger at Lake Abert (Herbst 2023, p. 7). Adults and pupae at Lake Abert and Mono Lake had similar nutritional values, but the larger larvae at the less saline Lake Abert had significantly more lipid and higher caloric content than the larvae at Mono Lake (Herbst 2023, p. 8). In addition to these effects,

food limitation, coupled with increased salinity, caused reduced survival and delayed maturation of alkali fly larvae and exposure of food-limited larvae to salinities >100 g/L caused development of the larvae to cease (Herbst 2023, p. 5). In both field and experimental settings, green algae, diatom, and cyanobacteria had growth rates reduced by half or more as salinity increased from 50 to 100 g/L (Herbst & Castenholz 1994, p. 590, Herbst & Blinn 1998, p. 773-774). Thus, increased salinities as lakes shrink are likely to result in an interacting impact on alkali flies of reduced food availability and direct negative physiological impacts. Herbst (2023, p. 5) identified a salinity management range of 25 to 100 g/L as the optimal range for the productivity of alkali flies, and approximately 50 g/L as the salinity level that maximizes their productivity as a food source for birds. The upper threshold of salinity for alkali flies was 150 g/L, the salinity at which larvae could not complete development (Herbst 2023, p. 03). At its 2022 elevation, the salinity of Lake Abert was approximately 250 g/L (Larson & Wood 2022, p. 15), well above the tolerance of alkali flies.

Brine flies: Most research on the effects of salinity on flies has been on alkali rather than brine flies, but in experimental treatments at GSL brine fly final biomass decreased 45% as salinity increased from 50 to 250 g/L (Barnes and Wurtsbaugh 2015, p. 6). GSL, which has typically been in the 100-150 g/L range since 2010, reached 185 g/L in fall 2022 as lake elevations reached record lows (Gilbert Bay station; Gilbert Bay and Saltair stations, USGS 2023, *entire*). Experimental observations from GSL indicate that at salinities of 250 g/L or greater, fly abundances were "very low," and that fly populations were inversely correlated to salinity (Collins 1980, p. 109). Preliminary data from summer 2022 indicated major decreases in fly populations at GSL, and those flies that were present were small and stunted (Larsen 2022a, *entire*).

Brine shrimp: The other key invertebrate prey species at phalarope staging sites are brine shrimp (*Artemia franciscana* at GSL and Lake Abert, *A. monica* at Mono Lake, and *A. salina* at Canadian sites).

Great Salt Lake*:* 160 g/L has been identified as the maximum salinity for a healthy population of *Artemia* in GSL (Marden et al. 2020, p. 197). This was based on experimental and field observations showing decreases in GSL *Artemia* density at >160 g/L, which was consistent with laboratory results in other *Artemia* species (Marden et al. 2020, p. 197, Abatzopoulos et al. 2003, p. 197). Marden *et al.* (2020, p. 197) found that "evidence from literature converges on 150-170 g/L as the maximum range for multiple *Artemia* species." Significant declines in *Artemia* survival were observed in salinities of >178 g/L after 3 weeks of experimental exposure in GSL water (Marden et al. 2020, p. 198). GSL brine shrimp exposed to salinities from 25 to 225 g/L had linearly decreasing growth rates, with 30% less growth in the highest salinity concentrations (Barnes & Wurtsbaugh 2015, p. 5). As salinities increased from 75 to 225 g/L, final *Artemia* mass decreased 60% and total biomass decreased 4-fold (Barnes & Wurtsbaugh 2015, p. 5). In other experiments, *Artemia franciscana* cysts from GSL did not hatch at salinities ≥250 g/L (Lindsay et al. 2019, p. 10-11). Brine shrimp at GSL have survived previous periods of heightened salinity at GSL, including at low lake levels in the 1960s when salinity was 260-280 g/L (Stephens 1990, p. 143). Though there are no data from this period, this was likely detrimental to *Artemia* populations (Marden et al. 2020, p. 199). *Artemia* are resilient and have

survived high and low salinity perturbations in GSL (Marden et al. 2020), but those previous perturbations to salinity were relatively brief.

Lake Abert: At Lake Abert, *Artemia franciscana* die offs occurred in 2010, when the lake exceeded 150 g/L salinity in July and rose to 170 g/L in August (**Fig. 26;** Larson et al. 2016, p. 395). During that event, *Artemia* turned red and died in nearshore windrows (Larson et al. 2016, p. 395, Larson 2023, p. 107). The causes of these mortality events appeared to be high salinities and associated low levels of dissolved oxygen (Larson 2023, p. 107). The red color was attributed to increased hemoglobin production as a result of hypersaline-induced hypoxia (Larson et al. 2016, p. 395, Larson 2023, p. 107). Similar die-offs of shrimp occurred at Lake Abert in 2013–2014 and 2015–2016 when the lake's level became very low and salinity exceeded 150 g/L (Larson 2023, p. 107). During 2014, when the lake was nearly completely desiccated and salinities exceeded 180 g/L, no shrimp were observed in Lake Abert (Larson et al. 2016, p. 395).

Figure 26: Masses of dying brine shrimp along the shore of Lake Abert in August 2010. The red coloration of the shrimp was due to hemoglobin, which is produced by the shrimp in response to low dissolved oxygen concentrations. Photo by Ron Larson.

Mono Lake: At Mono Lake, experimental exposures of *A. monica* to a range of salinities from 76 to 179 g/L showed that the lethal salt tolerance limit for subadults was between 159–179 g/L (Dana & Lenz 1986, p. 433). Many negative impacts occurred at high but sub-lethal salinities, including smaller adult sizes, slower growth, smaller brood sizes, greater female mortality during reproduction, and decreased percent of eggs hatching (Dana & Lenz 1986, p. 430-433). Life-time reproductive potential of females decreased linearly with increasing salinity treatments (Dana & Lenz 1986, p. 434-435). Those authors identified that extinction of *A. monica* was likely at salinities greater than 133 g/L, the threshold where reproduction and survival were severely reduced (Dana & Lenz 1986, p. 434-435). At Mono Lake, legal regulation of water kept the salinities from reaching levels with severe negative effects on brine shrimp. The current salinity of Mono Lake varies by annual lake levels but is approximately 81 g/L

([Mono Lake Committee 2023](#), *entire*). At the lowest lake levels while diversions were occurring unabated, the lake's maximum salinity was 99 g/L (Herbst 1988, p. 149). If diversions had continued unabated, salinities were predicted to fluctuate between 169–248 g/L by the mid-21st century (Vorster 1985, p. 231).

b. *Loss of prey resources due to exposure of submerged rocks*

A serious threat to phalaropes is the loss of alkali and brine fly prey resources when declining water levels cause exposure of submerged rocks at saline lakes. Submerged rocks (microbialites at GSL and tufa at Mono Lake) are important for saline lake ecosystem productivity because they provide a substrate for the reproduction and life stage development of alkali and brine flies (Herbst & Bradley 1993, p. 195, Wurtsbaugh 2009, p. 1). Submerged rocks are also important for primary production because they provide algae food resources for alkali flies and brine shrimp (Anderson et al. 2020, p. 2120). Microbialites at GSL are mats of benthic microbial communities that trap detrital sediment and act as the locus of mineral precipitation—essentially, the microbes build rocky structures (Lindsay et al. 2020, p. 88).

Great Salt Lake: Microbialites mostly exist in the shallow margins of GSL (Lindsay et al. 2020, p. 91). Microbial communities associated with microbialites provide a third to half of the primary production in GSL (Anderson et al. 2020, p. 2114, Frantz et al. 2023, p. 4). An estimated 88% of fly production in GSL originates from microbialite reefs (Collins 1980, p. 111), which make up only 20% of the lake bottom (Lindsay et al. 2020, p. 87). At both GSL and Mono Lake, densities of alkali fly larvae and pupae were much greater (10-20x greater) on hard, rocky substrates than on soft substrates (Herbst & Bradley 1993, p. 195, Wurtsbaugh 2009, p. 1). Abundance of adult brine flies was 1-2x greater over submerged microbialites at GSL compared with sand or mud substrates (Frank and Conover 2023, p. 407). Accordingly, phalarope abundance (Wilson's and red-necked lumped) was also highest over submerged microbialites (Frank & Conover 2023, p. 407). The shallow margins of GSL contain the majority of microbialite substrates hosting brine flies, which can quickly become dry as lake levels decline (Frantz et al. 2023, p. 10).

As GSL's water levels fell to record low levels in 2022, 40% of the lake's microbialites were exposed to air and desiccated (**Fig. 27;** Frantz et al. 2023, p. 5). Exposed microbialites became bleached and experienced weathering as surface chlorophyll was lost (Frantz et al. 2023, p. 18). However, when experimentally re-submerged, bleached microbialites quickly recovered at salinities ≤170 g/L salt content (Frantz et al. 2023, p. 14-15). At greater salinity levels, microbialites recovered more slowly and it was unclear to what extent microbial communities were re-established (Frantz et al. 2023, p. 19). Recovery at lower salinities appeared to be a result of primary seeding from microbes suspended in lake water, indicating that a lake with intolerably high salinity levels would not be able to re-seed healthy microbialite communities (Frantz et al. 2023, p. 19). Also, recovery occurred after short-term desiccation, and it remains unclear for how long microbialites can be desiccated and still recover (Frantz et al. 2023, p. 19). Aside from direct desiccation impacts, at salinities of 200 g/L or higher, there were also negative effects on the abundance of microbialite-associated biomass (Lindsay et al. 2019, p. 10).
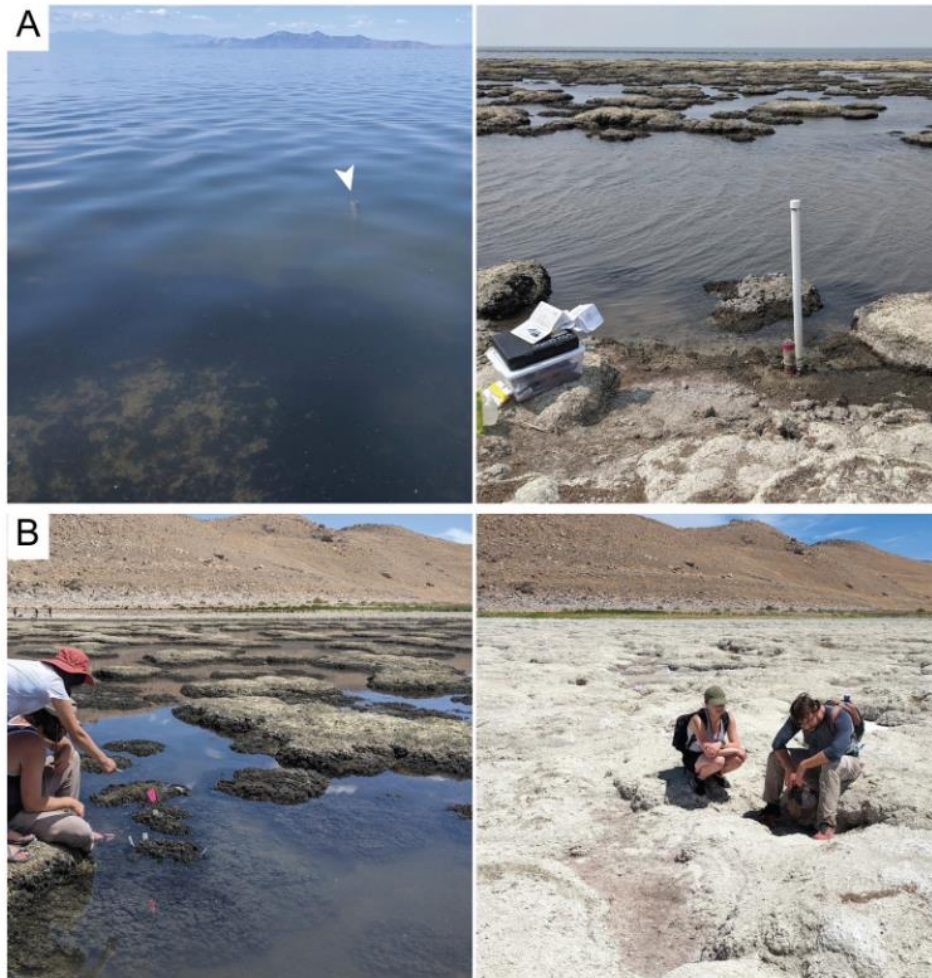
Figure 27: *Figure from Frantz et al. 2023 (p. 6) showing microbialite exposure with dropping lake levels at Great Salt Lake at one site in July 2019 and August 2021 (Panel A), and another site in July 2021 and July 2022 (Panel B).*

GSL's extremely low lake levels occurred very recently, and to date there are few data showing the impact on flies from the loss of submerged microbialites at GSL in 2022. However, preliminary observations from scientists indicated a major decrease in fly populations in 2022 (Larsen 2022, *entire,* Walter 2023, *entire*). Coinciding with these low fly populations, peak Wilson's phalarope numbers at GSL were the second lowest on record in 2022 (165,705 birds; only 1994 had a lower count at 118,000, n = 17 years with complete counts, 1992-2022; Jehl 1999, p. 40, Carle et al. 2023, p. 10). Loss of microbialite-related productivity and fly prey is an immediate threat to Wilson's phalaropes in relation to the drying of GSL, because Wilson's phalaropes rely heavily on brine flies during their staging period at GSL (Frank et al. 2021, p. 30).

Mono Lake: Tufa rocks at Mono Lake are restricted to shallow-water parts of Mono Lake (Herbst & Bradley 1993, p. 196, Keevil et al. 2022, p. 16). Alkali fly larvae and pupae in Mono

Lake are most abundant at depths of ≤1 m, occurring to a maximum depth of 10 m (Herbst & Bradley 1993, p. 195). At Mono Lake, alkali fly production was estimated to be maximized at a lake level of 6,380 feet, due to this elevation having the greatest abundance of shallowly submerged tufa (Herbst & Bradley 1993, p. 196). Nearly half the submerged rock habitat at Mono Lake would be lost at a level of 6,370' compared to 6,380' (Herbst & Bradley 1993, p. 196). At Mono Lake levels less than 6,372', fly abundance was predicted to be less than half of those at 6,380' based on the loss of that rocky habitat (Herbst & Bradley 1993, p. 196). Above an elevation of 6,380', rocks would be more deeply submerged, resulting in a decrease in fly habitat. However, the known positive effects on flies from lowered salinities at these higher levels (Herbst 2023, p. 5) was not modeled by Herbst & Bradley (1993). They did note, however, that decreased salinities would positively impact flies and that the optimal lake level might be near 6,385' if salinity were incorporated into the model (Herbst & Bradley 1993, p. 198). Also, they did not evaluate tufa presence between 6,390–6,392' elevations, so it is possible that tufa existing between those elevations could provide more habitat for flies (6,392' is the state-mandated lake management level; Herbst & Bradley 1993, p. 199).

In 2022, Mono Lake reached a level of 6,378' (McQuilkin 2023, p. 5). This elevation was below the optimal level for fly production, and in the elevation range in which submerged tufa habitat is rapidly lost with dropping lake levels (Herbst & Bradley 1993, p. 196). Had the wet winter of 2022/2023 not occurred, the level would have continued dropping (McQuilkin 2023, p. 5), with further desiccation of tufa and lost prey base.

c. *The importance of staging areas to shorebird survival and population decline*

A key question is how the loss of staging area habitat would impact populations of Wilson's phalaropes. Wilson's phalarope may have some flexibility in staging habitat choice, but smaller sites with lower invertebrate productivity cannot fill the place of large perennial hypersaline lake stop-overs. What would happen to phalarope populations if GSL and Lake Abert simultaneously collapsed, leaving Mono Lake as the only major saline lake staging site? Unfortunately, there are well-documented examples of shorebird species experiencing extreme population declines driven by loss of staging habitat. The link between staging area habitat loss and shorebird population declines is best documented for shorebirds using the Yellow Sea mudflats as a stop-over in the Austral-Asian flyway (Piersma et al. 2016, p. 488, Studds et al. 2017, p. 3-4, Murray et al. 2017, p. 873-874), and in red knots (*Calidris canutus rufa*) at staging sites in eastern North America, where they target specific prey (Baker et al. 2004, p. 875, Niles et al. 2009, p. 159, USFWS 2014, p. 73745). Both these cases have similarities to the situation with Wilson's phalaropes—just a few important staging sites with critical prey resources to fuel molting and migration, with bird populations concentrated at a small number of sites experiencing significant habitat loss.

Loss of key staging sites has been linked to population declines of red knots (*Calidris canutus piersmai*), great knots (*Caidris tenuirostris*), and bar-tailed godwits (*Limosa lappoica menzbieri*) in the Australasian flyway (Piersma et al. 2016, p. 488, Murray et al. 2017, p. 873-874). Declines occurred through reduced survivorship, specifically in the Yellow Sea staging

areas, and not during other parts of the annual cycle (Piersma et al. 2016, p. 488). This link has been confirmed by multiple studies, including Murray et al. (2017, pp. 873-874), who stated, "Our results support hypotheses that the extensive loss of tidal flat habitat at migratory staging sites plays a central role in the long-term declines of migratory shorebird populations in the EAA Flyway." Similarly, Studds et al. (2017, p. 3-4) stated, "Our analyses indicate that reliance on the Yellow Sea is a leading driver of population declines in migratory shorebirds of the EAAF… This result implies that population declines are driven by low survival during or soon after staging in Yellow Sea tidal mudflats, likely because birds are unable to refuel enough to meet the energetic demands of migration." Piersma et al. (2016, p. 487) found that shorebird populations that stage in the Yellow Sea declined suddenly, rather than gradually declining as habitat was reduced. They and others theorized that as habitat was lost, birds concentrated in smaller and smaller portions of remaining habitat, where they were subject to competition with each other (Piersma et al. 2016, p. 486-488, Mu et al. 2022, p. 2091). This density-dependent competition resulted in reduced feeding performance in remaining habitat (Piersma et al. 2016, p. 486-488, Mu et al. 2022, p. 2091). This theory of density-dependent competition for shorebird prey resources is supported by modeling predicting that non-breeding habitat loss leads to increased competition for prey, resulting in greater predator (e.g., shorebird) mortality and negative population impacts (Sutherland & Dolman 1994, p. 137). If GSL and Lake Abert become desiccated, it seems likely that a similar scenario could play out with staging Wilson's phalaropes concentrated at Mono Lake, which is a fraction of the size of GSL and cannot provide enough food for the entire population.

Similarly, food availability at key staging areas was linked to survival, reproduction, and rapidly declining populations of red knots in North America (Baker *et al*. 2004, p. 875, Niles *et al*. 2009, p. 153). Over-exploitation by humans of horseshoe crabs (*Limulus polyphemus*) in the key staging area of Delaware Bay led to greater proportions of red knots departing at lower masses, resulting in lower survival (Baker *et al*. 2004, p. 880-881, Niles *et al*. 2009, p. 158). Rapid population declines with trajectories leading toward extinction in the species were driven by conditions at key staging sites (Baker et al. 2004, p. 880-881). Reduced food availability at staging sites was considered an important factor in the red knot's decline and a justification for their listing as "Threatened" by the USFWS in 2014 (USFWS 2014, p. 73745). In reviewing the management of red knots at Delaware Bay, Niles et al. (2009, p. 159) noted "There can be no doubt that the Delaware Bay food supply has played a critical role in [*Calidris canutus*] rufa's decline."

There is reason to believe that the Wilson's phalarope is subject to the same population dynamics as described for shorebirds staging at the Yellow Sea and red knots at Delaware Bay, given the historic, current, and impending loss of habitat at GSL, Lake Abert, Lahontan wetlands, and San Francisco Bay (Senner et al. 2018, p. 327). Wilson's phalarope's dependence on just a few large staging sites with specialized prey resources makes them vulnerable to habitat loss (Senner et al. 2018, p. 327). The example of the red knot is of a spring-migration staging site; however, the summer-fall-staging site ecology of Wilson's phalaropes is also a case of 1) limited habitat with abundant and sufficiently high-calorie food, 2) major energetic needs for a complete molt and doubling of body weight, and 3) rapid habitat loss and degradation, including loss of the prey base, at those key staging sites. Should GSL's and Lake Abert's ecosystems

collapse, Mono Lake would likely still persist. However, even at Mono Lake's greatest recorded surface area since 1980, it was only 7% the size of GSL's historically low 2022 surface area (Hall et al. 2023, p. 16). Mono Lake is only approximately 3-4% the size of GSL at more average GSL lake levels (Conover & Bell 2020, p. 241).

In the absence of GSL and Lake Abert, Mono Lake and other smaller Great Basin sites cannot be expected to provide enough food to sustain the entire Wilson's phalarope population that uses GSL and Lake Abert. Based on observed population trajectories of shorebirds experiencing staging habitat loss elsewhere, it is reasonable to expect that phalarope populations would crash in this scenario, as high densities of birds competed for prey resources at limited remaining habitat. Furthermore, desiccation of GSL is most likely to occur during a region-wide drought, in which case smaller regional wetlands are also likely to desiccate. Similar to the mechanisms thought to be causing shorebird populations to crash in the Yellow Sea staging areas, phalaropes might concentrate at the few sites still providing habitat, where intense inter- and intra-specific competition for food could result in mass mortalities.

It is important to remember that should GSL's ecosystem collapse, large numbers of other bird species that depend on saline lakes would also need to seek out remaining habitat, including up to 5 million eared grebes (Conover & Bell 2020, p. 247) and hundreds of thousands of red-necked phalaropes (Conover & Bell 2020, p. 249, Carle et al. 2023, p. 12). Eared grebes rely on saline lakes in much the same way as Wilson's phalaropes, with large proportions of the world population using the abundant prey base to accomplish energy-intensive and rapid weight gain and molts (Jehl 1988, p. 1). Mono Lake and other smaller habitats cannot be expected to provide enough calorie-rich food to meet the energetic needs of all these birds, in the absence of GSL and Lake Abert.

2. Threats at distinct phalarope habitats

a. Great Salt Lake

"If these trends of increasing salinity and decreasing depth [at GSL] continue, it will be catastrophic for those avian species that are dependent on brine flies for food because other hypersaline lakes in North America are not large enough to support the populations of phalaropes, avocets, and stilts that currently depend upon GSL for their survival."
--Frank & Conover 2023, p. 411

GSL (41.115 N°, -112.477 W°) is the largest lake by area in the western US and the eighth largest saline lake in the world (Wurtsbaugh et al. 2017, p. 2). It is 10-20 times larger than Mono Lake or Lake Abert (**Table 4**). GSL is relatively shallow, with a 4.5 m average depth when at a level of 4,200'; Baskin 2005, p. 2, Null & Wurtsbaugh 2020, p. 2). Overall, the lake is hyper-saline, but has salinities ranging from less than 1%-34% because of uneven freshwater inflow into the lake's several arms and impoundments exacerbated by artificial barriers such as

dikes and causeways (**Fig. 28**; Gunnell et al., 2022, p. 12, Null & Wurtsbaugh 2020, p. 2).
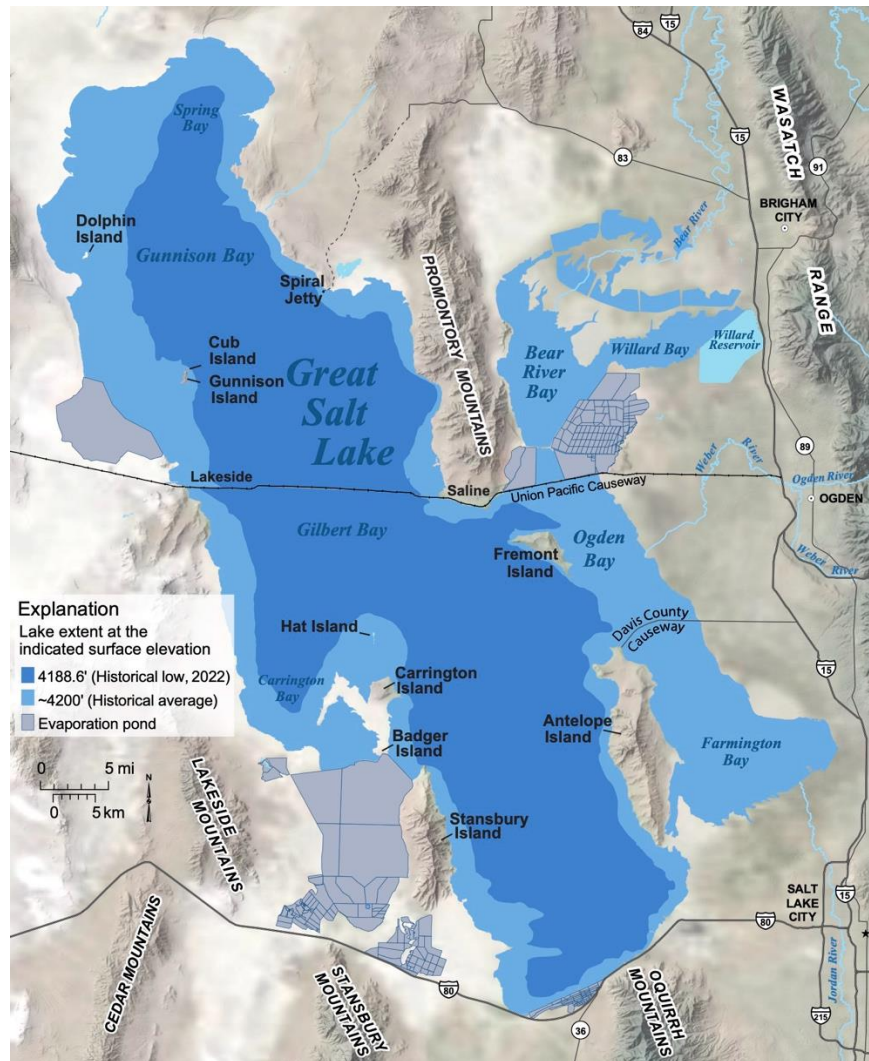
Figure 28: *The morphology of Great Salt Lake at two different stands (*<u>*Utah Geological Survey*</u>*).*

Like most saline lakes, Great Salt Lake has a large watershed crossing multiple jurisdictions (**Fig. 29;** Wurtsbaugh et al. 2017, p. 4). The 23-million-acre watershed is divided into four main basins, with the Bear, Jordan, and Weber watersheds contributing approximately 65% of the lake's inflows (Null & Wurtsbaugh 2020, p. 2). Of that 65% of inflow, the Bear River provides ~50%, the Jordan River provides ~25%, the Weber River provides ~20%, and the rest is from small streams (Great Salt Lake Strike Team, 2023, p. 9). Direct precipitation into the lake surface accounts for about 33% of inflows, with the remaining 2% of inflow from groundwater and ephemeral West Desert streamflows (Null & Wurstbaugh 2020, p. 3).
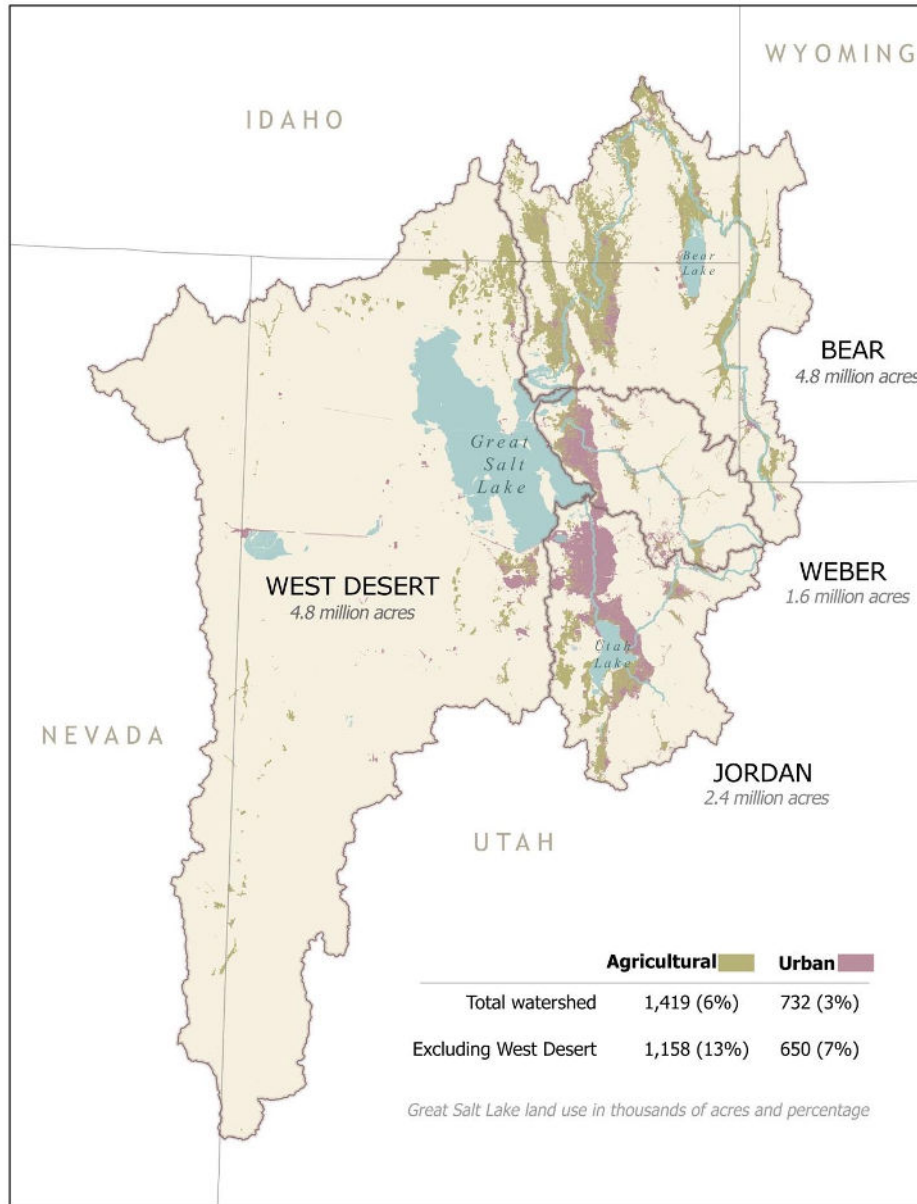
Figure 29: *Map of the Great Salt Lake watershed, including the most extensive land uses (agricultural and urban).*

Human water diversions began affecting GSL in the mid-1800s, shortly after European settlement (Wurtsbaugh & Sima 2022, ps. 1 and 6). The lake level averaged 4,204' from 1847 to 1900, close to its natural long-term mean of 4,207' (Wurtsbaugh et al., 2016, p. 3). However, in the 1900s, humans became the dominant force controlling the lake (Farmer 2014, p. 27, Wurtsbaugh et al. 2016, p. 2, Wine et al. 2019, p. 5). Federal and state construction of dams, canals, and pipelines allowed more of the watershed's natural runoff to be diverted for agricultural, industrial, and municipal use (Wurtsbaugh et al. 2016, p. 2, Wine & Laronne 2020, p. 1, Wurtsbaugh and Sima 2022, p. 4). These subsidized water projects led to unsustainable water consumption (Di Baldassarre et al. 2020, p. 1798, Fort 200, p. 125-126) and declining lake

levels through the 1960s (Wurtsbaugh et al. 2015, p. 2, Null & Wurtsbaugh 2020, p. 2). The construction of a railroad causeway 1959 divided GSL into a "North Arm" and "South Arm," isolating approximately 40% of GSL's surface in the North Arm from direct surface inflow of freshwater (White et al. 2015, p. 2). Because of this, the North Arm's current salinity range is 190–351 g/L, which is above the tolerance of brine shrimp, flies, and the primary producers they depend on (White et al. 2015, p. 13; Frantz et al. 2023, p. 18). Consequently, these invertebrates no longer occur in the North Arm (Marden et al. 2020, p. 184). Phalarope, which were abundant in the North Arm when it was less saline, have not been observed there on surveys from 2012-2021 (Frank and Conover 2023, p. 411). The North Arm of GSL provides a bleak example of what will happen to the South Arm if unsustainable water use continues (**Fig. 30**).
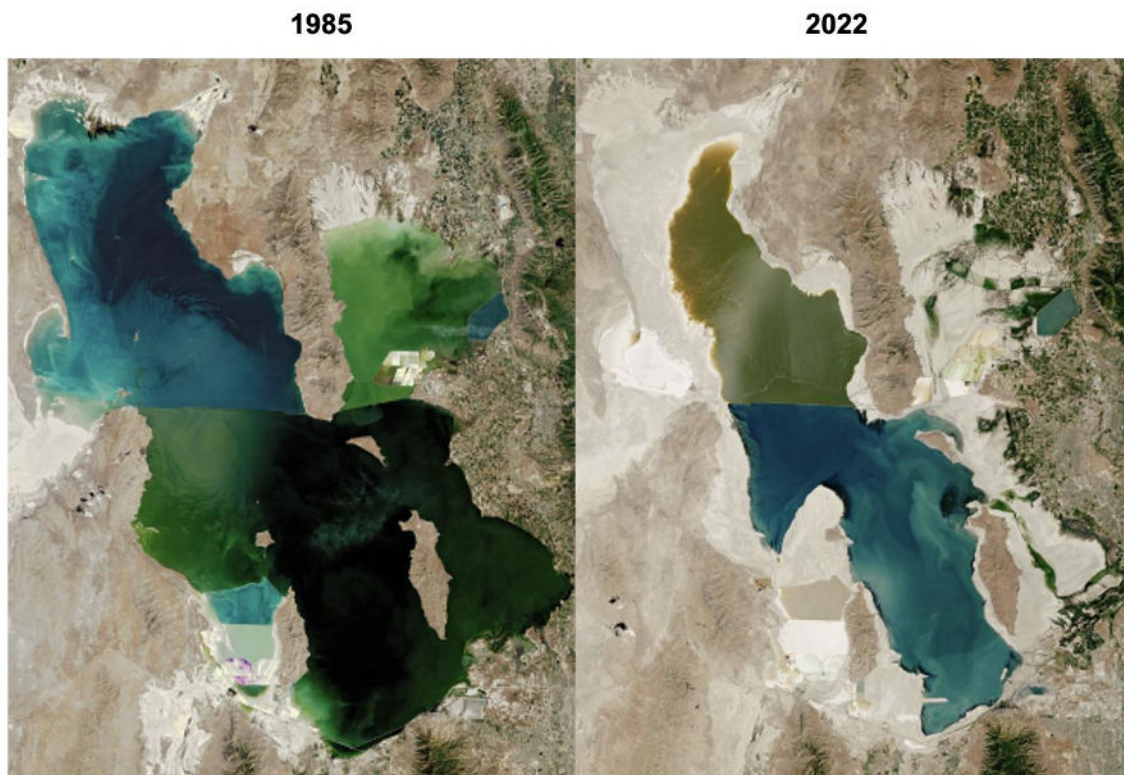
*Figure 30: Satellite imagery of the lake extent in 1985 and 2022 (NASA).*

An extremely uncommon wet period in the 1980s temporarily refilled the lake (Wine et al. 2019, p. 5, Trentelman 2020, p. 65), but since peaking in 1987, GSL's level has been in structural decline (**Fig. 31;** Hall et al. 2023, p. 16, Wurtsbaugh et al. 2017, p. 3). In November 2022, the lake level dropped to its lowest-ever recorded elevation of 4188' (USGS 2023, *entire,* Baxter & Butler 2020, p. 27; annual record-keeping began in 1847). This was below the lowest level on the state's contingency charts (Gaddis et al. 2012, p. 90, GSLAC 2019, p. 96, UDNR 2022a, *entire*). That record low represented a 60% decline in GSL surface area and a 73% decline in water volume compared to the mean estimated natural water level without water diversions (4,207'; Wurtsbaugh et al. 2016, p. 3, Abbott et al. 2023, p. 5).
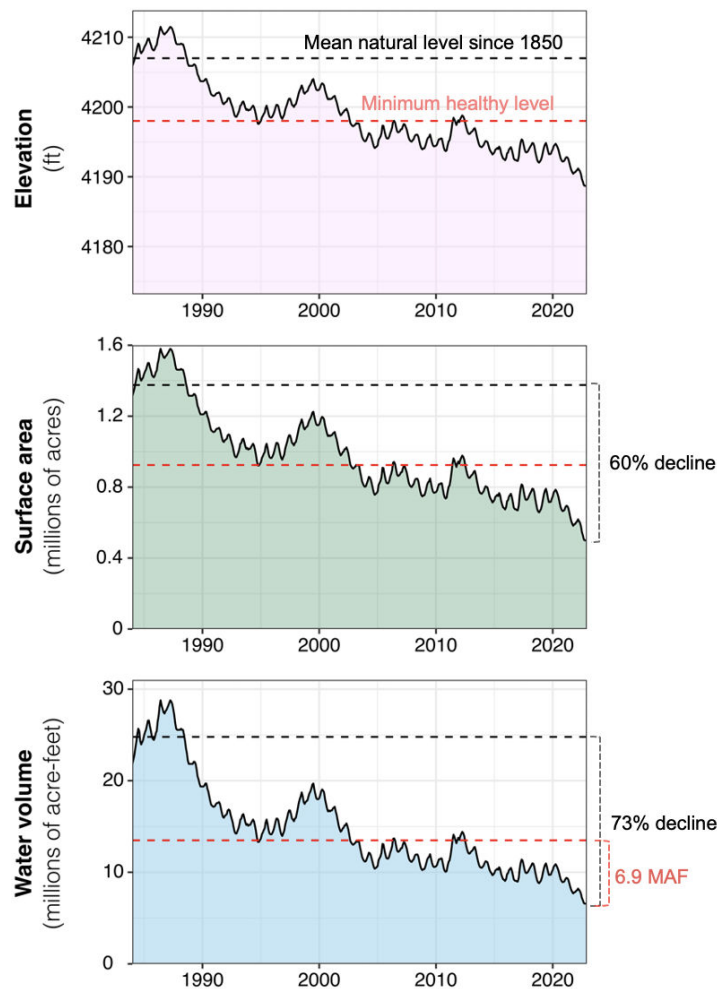
Figure 31: Elevation, extent, and volume of Great Salt Lake from 1985 to 2022. The mean natural values represented by the black dotted lines were determined from estimated 1850-2016 values without human water use (Wurtsbaugh et al. 2016). Data from USGS 2023, Wurtsbaugh et al. 2016, and Tarboton 2017.

GSL's extremely low recent lake levels have caused extensive negative ecological impacts that threaten the entire ecosystem, particularly seven bird species, including Wilson's phalarope and Eared grebe (Daniels et al., 2024, p. 27, Steed 2024, p. 21-22). In 2022, GSL's salinity (measured at Gilbert Bay in the South Arm) rose to 185 g/L (**Fig. 32;** USGS 2023, *entire*). This exceeded the 160 g/L salinity threshold beyond which the survival and reproduction of brine shrimp are compromised (**Fig. 32;** Marden et al. 2020, p. 197). Sustained salinities at this level threatened ecosystem collapse by causing brine shrimp to fail to hatch. The high salinities in 2022 also exceeded levels known to have negative impacts on brine fly and alkali fly growth, reproduction, and survival (Herbst 2023, p. 1, Barnes and Wurtsbaugh 2015, p. 6), as well as the primary producers upon which these invertebrates depend (Frantz et al. 2023, p. 18). At GSL's record-low 2022 level, 40% of the lake's submerged microbialite formations, were

exposed to air and desiccated (Frantz et al. 2023, p. 5, Frank & Conover 2023, p. 403). These microbialite formations support at a third of the lake's primary productivity, nearly 90% of its brine fly population, and the lake's entire summer brine shrimp population (Collins et al., 1980, p. 1, Barrett, 2021, p. 71, Frantz et al. 2023, p. 5). Furthermore, each 0.33-m decrease in GSL surface elevation results in a concomitant loss of 40–53 km^2 of surface area from GSL (Baskin 2005, p. 6, Frank & Conover 2019, p. 11)–with the simple result that there is less habitat area for phalaropes and their invertebrate prey. For example, Bear River Bay and Ogden Bay, once important phalarope habitats, have been largely dry since 2014–2015 (Frank & Conover 2019, p. 9, Frank and Conover 2023, p. 410). Other areas used by phalaropes, such as Farmington Bay, have shrunk in area, causing phalaropes to concentrate in the remaining narrow strip of water (Frank & Conover 2019, p. 9). Thus, the low lake levels and high salinity of recent GSL elevations have severely threatened, and already caused degradation of the Wilson's phalarope prey base and habitat extent.
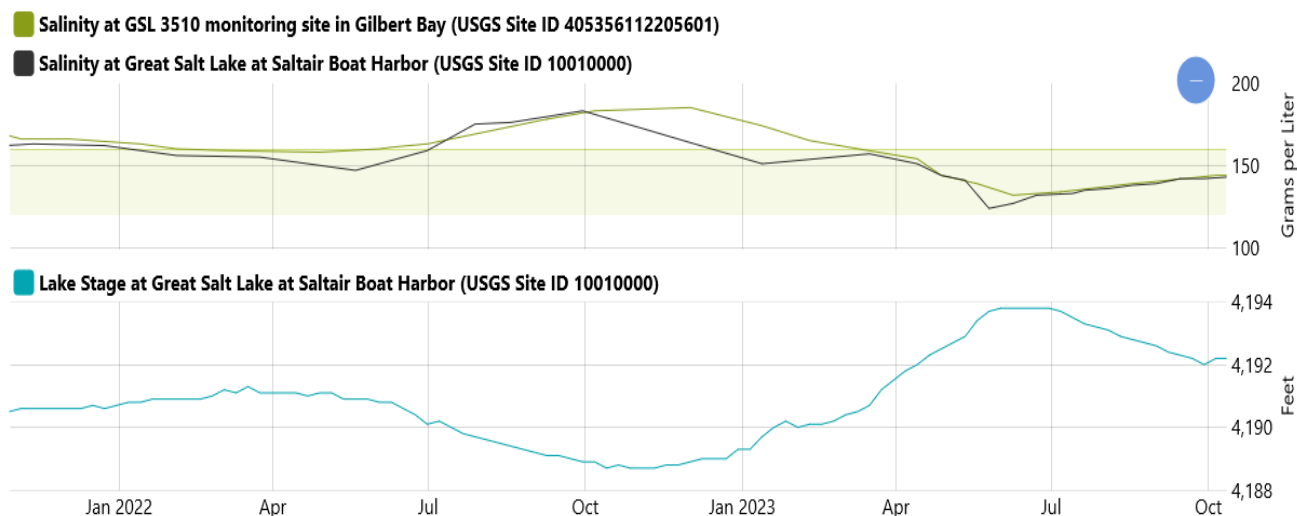
Figure 32: *Salinity (top) and lake elevation (bottom) of GSL from January 2022 to October 2023 from USGS GSL Hydro Mapper ([USGS 2023](#)). From late June 2022 to February 2023, GSL's salinity at Gilbert Bay (green line) exceeded the optimal range for brine shrimp (120-160 g/L, green shading). GSL's 2023 surface elevation peaked in July before again starting to decrease.*

The decline of GSL has been caused predominantly by human water consumption (Wurtsbaugh et al. 2019, p. 3, Wine et al. 2019, p. 1). Approximately 85% of the reduction in GSL's area as of 2019 was attributed to human consumption of water (Wine et al. 2019, p. 1). From 2019–2022, the lake received less than a third of its natural streamflow because of excessive water use (accountable for 67-73% of decline) plus drought and climate change (23-35% of the decline; UDNR 2022b, p. 3; Great Salt Lake Strike Team, 2023, p. 4). Most of this water diversion was for agriculture–irrigation of alfalfa and other crops directly accounts for 64-75% of total consumptive water use plus 5-10% indirectly through storage and transport losses such as reservoir evaporation (Wurtsbaugh et al. 2016, p. 4, Null & Wurtsbaugh 2020, p. 7, Wilson et al. 2022, p. 27, Great Salt Lake Strike Team, 2024, p. 13). Mineral extraction from the

lake itself represents another 7-9% of water use (Abbott et al. 2023, p. 7, Great Salt Lake Strike Team, 2024, p. 13). Cities and industry account for the final 9-16% of consumptive water use, of which 90% is outdoor water use (irrigation for lawns and other decorative plants; Abbott et al. 2023, p. 7, Great Salt Lake Strike Team, 2024, p. 13). The industrial portion of the consumptive water use includes mining, electricity production, and other industrial processes (Utah Division of Water Resources 2010, p. 7, Dieter et al. 2018, p. 8). The most recent water balance study found that without human water use, GSL lake elevation would likely be 4.6 m (15') higher (Merck and Tarboton 2024, p. 21), which would put it squarely in the state's established healthy range (Gaddis et al. 2012, p. 90, GSLAC 2019, p. 96, UDNR 2022a, *entire*).

Irrigated agriculture covers 1.4 million acres or 6% of the watershed. 63% of this agricultural land occurs in Utah, 31% in Idaho, 5% in Wyoming, and 1% in Nevada. Urban development covers 0.7 million acres or 3% of the watershed area, with 93% occurring in Utah (Abbott et al. 2023, p. 6). Indoor water use has less direct effect on lake level because ~95% is returned after wastewater treatment, though the storage, conveyance, and treatment of water used indoors does cause consumptive losses and degradation of water quality (Utah Division of Water Resources 2010, p. 4, Abbott et al. 2021, p. 23). However, an increase in permit applications in 2023 to recycle wastewater in the GSL watershed could reduce the amount of water returned to the lake (Johnston 2023, *entire*).

Another significant threat to the stability of the Great Salt Lake ecosystem comes from the Bear River Development Act passed by the Utah legislature in 1991 and still on the books (Utah Division of Water Rights, *entire*). This act authorizes and directs the Division of Water Resources to "…*develop the surface waters of the Bear River and its tributaries through the planning and construction of reservoirs and associated facilities*…" If completed, the water infrastructure project would divert up to 220,000 acre-feet of water annually from the Bear River–the major source of water for GSL (Utah Division of Water Rights, *entire*).

Climate change is a secondary contributor to the decline of GSL (Wurtsbaugh et al. 2019, p. 4, Wine et al. 2019, p. 1, Great Salt Lake Strike Team, 2023, p. 11). Human greenhouse gas emissions have caused ~4°F of warming in the GSL region since 1900 and exacerbated drought throughout the southwestern U.S. (Williams et al. 2020, p. 4, 2022, p. 1, Zhang et al. 2022, p. 1, Wilson et al. 2022, p. 35). This climate change has reduced runoff to GSL and increased evaporation, accounting for ~9% of the lake's decline based on current estimates (Wurtsbaugh et al. 2017, p. 4, Abbott et al. 2019, p. 3048, Wine et al. 2019, p. 2, Wilson et al. 2022, p. 35). Future climate change is projected to increase evapotranspiration more than precipitation, which will reduce runoff to the lake, though the magnitude of these trends are poorly constrained (Wilson et al. 2022, p. 35; Great Salt Lake Strike Team, 2023, p. 15). This decrease in runoff will make reversing the lake's decline even harder (Van Loon et al. 2016, p. 1, Wilson et al. 2022, p. 35, Cook et al. 2022, p. 9).

Based on the best available hydrological and societal-impacts data, Abbott et al. (2023, p. 13) recommended setting a minimum streamflow requirement of 2.5 million acre-feet per year to recover GSL. River flow at or above this threshold corresponds strongly with periods of lake elevation rise (**Fig. 33**). This is approximately one million acre-feet per year more than the recent annual average streamflow to the lake (1.6 million acre-feet per year since 2020; Abbott et al. 2020, p. 13, UDNR 2022b, p. 1). Depending on the near-future climate conditions, this could

require 0.7 to 1.2 million acre-feet per year of conservation (Great Salt Lake Strike Team, 2024, p. 17), representing a 30 to 50% reduction in consumptive water use in the watershed (Abbott et al. 2023, p. 13).
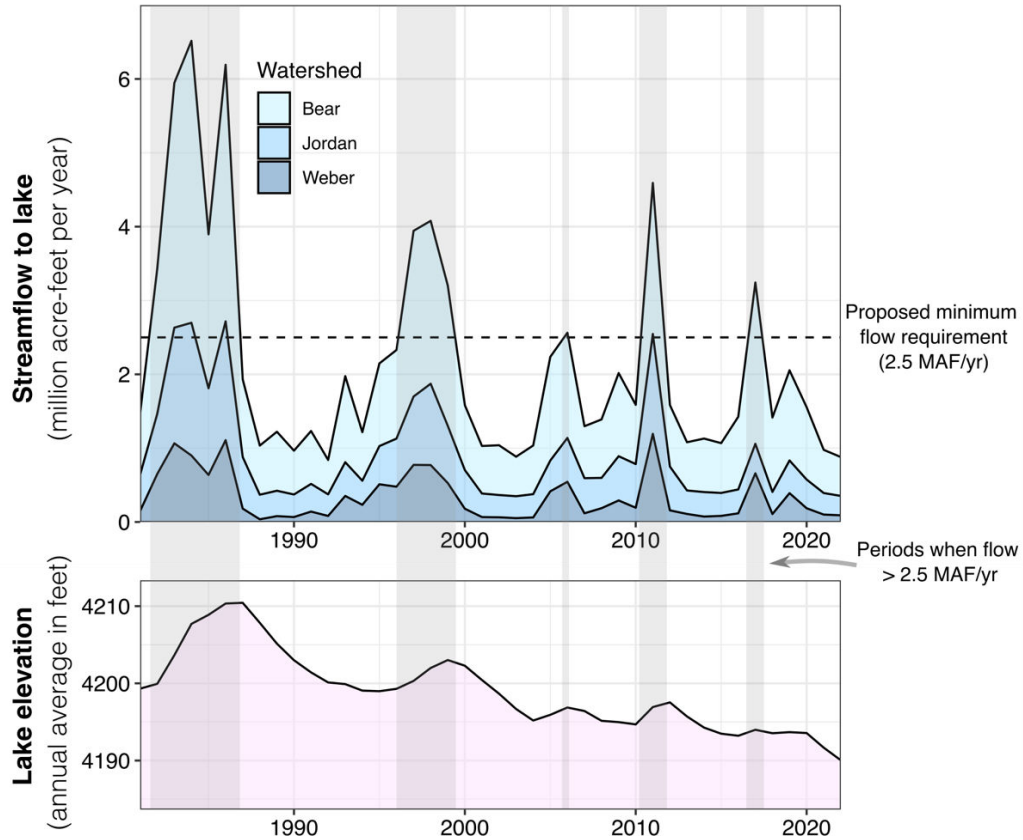
Figure 33: Streamflow to Great Salt Lake and annual lake elevation since 1981. The vertical shading shows years when streamflow equaled or exceeded the proposed minimum flow requirement of 2.5 million acre-feet per year.

While recent legislative changes to Utah's water management will likely contribute to water conservation in the coming decades, they are not adequate to help the lake through its current crisis. In fact, if legal, financial, and technical support is not provided for water users to implement these changes, the new policies could have little to no influence on Great Salt Lake in 2024 and beyond (Steed 2024, p. 14, Daniels et al., 2024, p. 9). For example, conservation in 2022 and 2023 increased streamflow to the lake by less than 100,000 acre-feet, with most conserved water held in reservoirs or consumed elsewhere in the watershed (Larsen 2022b, *entire*, Steed 2024, p. 10, 14). After the record-breaking wet winter of 2022–2023, the south arm of GSL rose to a peak in June 2023 of 4,193.8' and salinities in the south arm dropped briefly to 132 g/L (**Fig. 32;** USGS 2023, *entire*). The lake level began dropping again in August 2023, and by December 2023 levels had decreased to 4,192.1' and salinity had increased to 144 g/L (USGS 2023, *entire*). GSL's 2023 level was 7-8 feet below the 4,200 level proposed to keep the lake healthy, and 5-6 below the 4,198' management level at the bottom of the Great Salt Lake

Commissioner's target range of 4,198-4,205' in January of 2024; Larsen 2023, Steed 2024, p. 24). Utah's 2023 water year was 136% of normal (Williams 2023, *entire*) and cannot be expected to be replicated every year. Even with this record-breaking infusion of water to the lake, the year-over-year overall lake level rose only around 2' because of the capturing of runoff in reservoirs (Great Salt Lake Strike Team, 2024 p. 7; Steed 2024, p. 11). With increasing water demand and decreasing water availability due to climate change, the most likely GSL outlook is one of disappearance or extreme reduction unless there are major changes in water policy and use (Abbott et al. 2023, p. 2). Given the difficulty of implementing rapid change with state resources and political realities (Steed 2024, p. 4, Daniels et al. 2024, p. 9), GSL is likely to remain in crisis for the foreseeable future with low water levels and high salinities that would result in habitat loss for up to 60% of the world population of Wilson's phalaropes (Jehl 1999, p. 40, Castellino et al. 2024, p. 15).

Future outlook for invertebrates at Great Salt Lake:

The future outlook for GSL is of sustained extremely high salinities and low lake levels. Observed die-offs of *Artemia* in multiple years at Lake Abert are a preview of what could happen at GSL if persistent salinities of >160 g/L occur. Had the winter of 2022–2023 not been extraordinarily wet, those salinities would have occurred in GSL, with major consequences to *Artemia* and brine flies, carried up the food chain to phalaropes and other birds. Furthermore, in 2022, Lake Abert was nearly fully desiccated, leaving only Mono Lake as a suitable staging habitat.

The North Arm of Great Salt Lake, where *Artemia* are extirpated because of high salinity levels, is a good corollary of a scenario in which the South Arm salinity levels exceed the salinity tolerance of invertebrates. The construction of a railroad causeway in 1959 isolated the North Arm of the lake from nearly all freshwater inflow (White et al. 2015, p. 2). Because of this, the North Arm's current salinity range is 190-351 g/L, above the tolerance of brine shrimp and flies (White et al. 2015, p. 13). These invertebrates no longer occur in the North Arm, which composes approximately 40% of GSL (Marden et al. 2020, p. 184). This demonstrates the vulnerability of Wilson's phalarope populations to the effects of habitat loss from water diversion and climate change at hypersaline lakes.
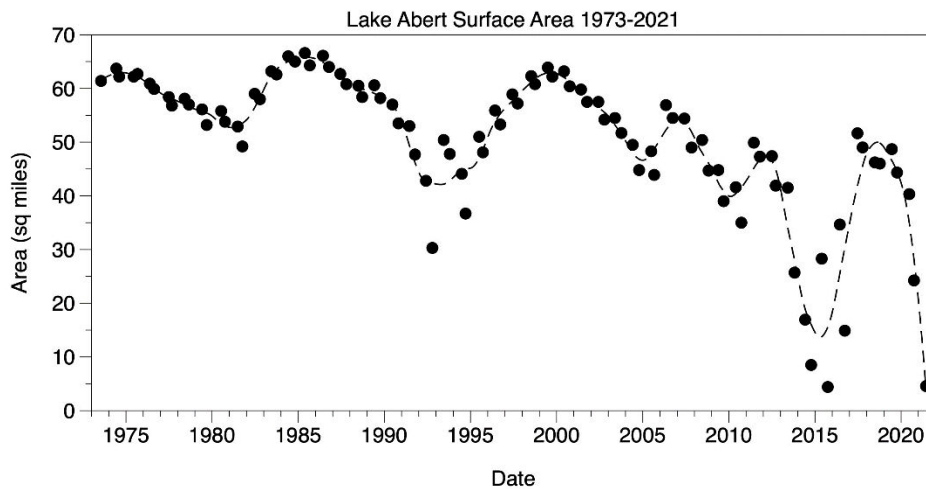
    b. *Lake Abert*

Lake Abert (42.66°N, 120.23°W) is located in the northwestern part of the Great Basin in south central Oregon (Larson 2023, p. 3). The lake gets most of its water from the Chewaucan River, which drains snow-covered mountain ranges to the west of the lake, as well as small tributaries in the lower part of the Chewaucan watershed (Larson 2023, p. 4). The Chewaucan River is a highly over-allocated system, with allocated water rights far exceeding the water in the river in most years (Moore 2016, p. 151). Water is diverted from the Chewaucan River for human use below Paisley, primarily for flood-irrigation of grass-hay, and upstream diversions annually remove approximately half or more of the flow of the river (Moore 2016, p. 151).

According to Moore (2016, p. 151) Essentially, all the surface water can be legally extracted from the river leaving nothing to flow into Lake Abert. The highest "recent" surface-water elevation of Lake Abert was 1298.6 m in 1958 (Larson 2023, p. 86). At the opposite extreme, Lake Abert was dry or nearly dry during the Dust Bowl era in the 1920s and 1930s, reaching a low of ~1294.0 m in 1937 (Larson 2023, p. 86). More recently, Lake Abert became desiccated in 2014–2015 and again 2021–2022 due to upstream water diversions and drought (**Fig. 34;** Moore 2016, p. 146, Larson et al. 2016, p. 389, Hall et al 2023, p. 16, Larson 2023, p. 87). Since around the year 2000, Lake Abert's water level has varied considerably but overall has had a steep declining trend (**Fig. 35;** Larson 2023, p. 86).

Figure 34: *Lake Abert in October 2002 and October 2022. The small amount of water on the right side of the lakebed in 2022 was from localized freshwater spring inputs. Satellite images from* [*NASA Earth Observatory 2022*](NASA Earth Observatory 2022).

Figure 35: *Lake Abert surface area, 1974-2021. Data from Larson 2023, p. 86.*

Between 2013 and 2014 the lake area shrank from 45 to 10 square miles and continued to shrink in 2014 until the lake was totally desiccated except for a ~1 square mile area of water coming from a nearby spring complex (Larson 2023, p. 86-87). The lake reached the same levels, with only localized spring water inputs, in 2021–2022 (Larson 2023, p. 86-87). Salinities spiked during these desiccation events, reaching >250 g/L during both 2014–2015 and 2021–2022 (Moore et al. 2016, p. 146, Hall et al. 2023, p. 4). In analyses of the 2014-2015 desiccation event, both Moore (2016, p. 149) and Larson et al. (2016, p. 400) concluded that (1) water withdrawals for agriculture substantially contributed to the desiccation event, (2) without water withdrawals the lake would have maintained a higher volume and lower salinity, and (3) such conditions would mostly have been within the salinity optimal range for brine shrimp and alkali flies, which is <150 g/L. Low flows in the Chewaucan River were a contributing factor to that desiccation event, but compared to historical flows they were not drastically low (Larson et al. 2016, p. 400). Besides agricultural diversions from the river, increased groundwater pumping since around 1990 may have contributed to reduced inflows into the lake (Larson et al. 2016, p. 400). As stated by Larson et al. (2016, p. 400):

> *Since about 1990, groundwater usage has increased in the basin, as evidenced by the proliferation of center-pivot irrigation for alfalfa production, which now occupies approximately 67 km². This shift from production of grass hay to alfalfa, which is a water-intensive crop, has likely adversely affected the aquifer, especially since 2001 as a result of the recent drought and higher evaporation rates. In fact, there has been a steady decline in the water table, based on data reported by OWRD for Lake County wells #1633 and #1719.*

The desiccation events at Lake Abert reduced the availably of habitat for Wilson's phalarope and also impacted the quality of the habitat by increasing the salinity to >250 g/L. The productivity of Lake Abert's brine shrimp, alkali flies, and algae is much reduced at salinities >100 g/L (Herbst & Castenholz 1994, p. 590, Herbst 2023, p. 1), and salinities >150 g/L or 160 g/L become lethal for flies and brine shrimp (Herbst 2023, p. 5, Marden et al. 2020, p. 197). Relative abundance of adult alkali flies was reduced during periods of high salinity and low lake levels at Lake Abert (Larson & Wood 2022, p. 16). The nearly 90% decrease in phalarope abundance during the two recent desiccation events was likely a result of less food being present, especially since the lake area was <1 mile² in 2021–2022 (**Fig. 36**).

Figure 36: *The dry bed of Lake Abert in July 2022. Photo by Ron Larson.*

The impact of staging habitat loss on phalaropes is most well documented at Lake Abert, because of the recent desiccation events, and it is clear that phalarope numbers correlate to lake levels, and that at desiccation levels phalarope numbers drop by around 80–90% (Senner et al. 2019, p. 326, Carle et al. 2022, p. 18). Unfortunately, water inflows remain over-allocated and there is no indication that government intervention or regulation is on the horizon. Indeed, the state of Oregon suspended its own efforts to monitor water in the Chewaucan river recently (Davis 2022, *entire*). Another troubling new development is a proposal by the company PacifiCorp for a pumped storage hydropower in the Chewaucan River watershed, which would involve construction of two new approximately 50-acre reservoirs that would be filled with water diverted from the Chewaucan (Parks 2022, *entire*). If this project were built, it would divert more water from the over-allocated Chewaucan River and likely further reduce the amount of water reaching Lake Abert.

Given the over-allocation of its freshwater inputs, continued groundwater pumping in the Chewaucan Valley, increasing aridification from climate change, and threats of new reservoirs on the Chewaucan, without major policy changes, Lake Abert can be expected to continue to experience increasingly frequent desiccation events in the near future (**Fig. 37**). Aside from the 1 km^2 area fed by local springs, in many years it can be expected that Lake Abert will provide no phalarope habitat. Lake Abert provides a vivid illustration of the potential impact that the loss of GSL would have on phalaropes—but losing GSL would impact an order of magnitude more birds and half the world population of the species.
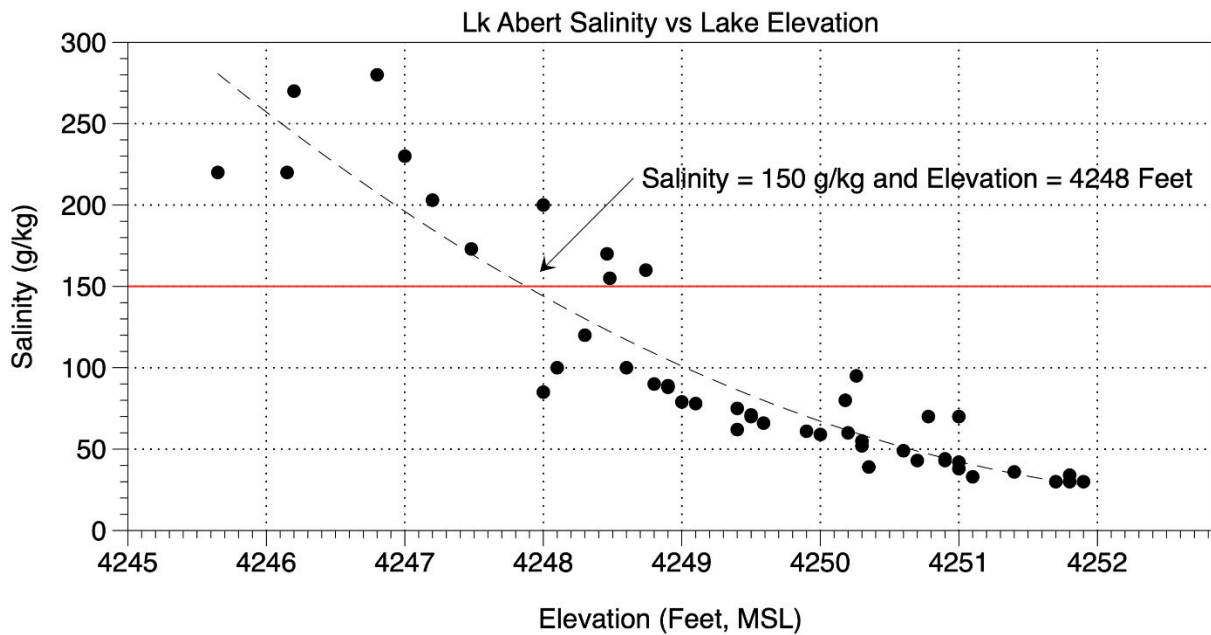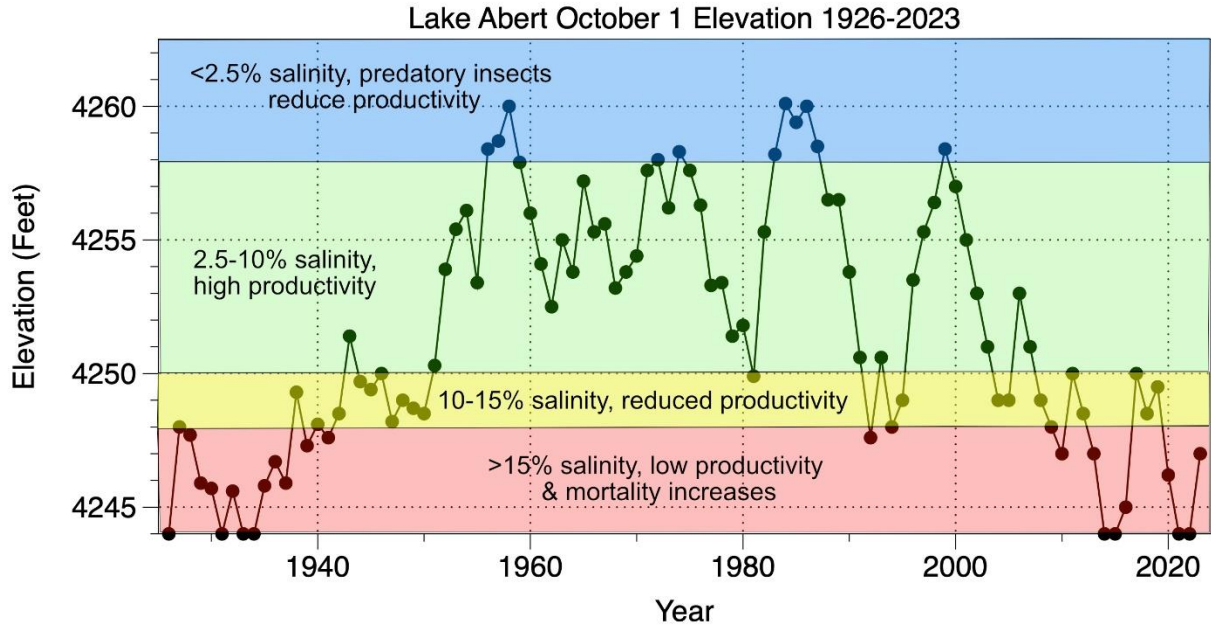
Figure 37: *Top: Approximate Lake Abert elevations 1925–2021 and how they affect shrimp and fly populations. At elevations of ~4250-4258 feet, salinities are optimal for the biota, shown in green (Herbst 1994). However, when the water levels drop below 4250 feet and salinity increases, the biota can become stressed, as shown in red, and that can lower productivity and cause death. Diagram source: David Herbst. Bottom: Relationship between Lake Abert elevation and salinity. The 15% salinity maximum for productivity is shown as a red line. Data from Phillips and VanDenburgh (1971), and Herbst, Kreuz, and Larson pers. com.*

c. Mono Lake

A refuge but not a replacement: Mono Lake is a hyper-saline lake located in east-central California. Mono Lake is a critical phalarope staging site (**Fig. 38**), but has only approximately 4-5% of the surface area of GSL and typically hosts <10% of the phalarope numbers of GSL (**Table 4;** Jehl 1999, p. 40). Beginning in 1941, four of Mono Lake's (38.028° N, -119.011° W) major tributary streams were entirely diverted for use by the City of Los Angeles, causing the lake to decline by an average of 1–2 feet a year until 1970 (Winkler 1977, p. 2). From 1940-1980, water diversions caused Mono Lake's surface elevation to decrease from 6,416' to 6,376', with the surface area shrinking from 85 to 63 square miles (Arnold 2004, p. 14). The salinity of the lake nearly doubled between 1941 and 1982 (from 48 to 93 g/L) because of diversions from freshwater streams (Dana & Lenz 1986, p. 428). The decrease in lake level caused many ecological problems relevant to phalaropes. If diversions had continued unabated, salinities were predicted to fluctuate between 169 and 248 g/L by the mid-21st century (Vorster 1985, p. 231), with extinction of the Mono Lake brine shrimp *Artemia monica* predicted if the lake reached salinities above 133 g/L (Dana & Lenz 1986, p. 435). Hydrologic modeling simulating continued diversions shows the lake would have fallen to 6,350' by 2016 (McQuilkin 2016, p. 24). At elevated salinity levels below those expected to cause ecosystem collapse, *A. monica* and alkali flies display decreased growth and productivity (Dana & Lenz 1986, p. 431, Dana & Herbst 1977, p. 68-69, Herbst & Bradley 1993, p. 198). As the lake level dropped, the amount of submerged tufa that alkali flies use as a pupal substrate was also reduced, reducing the amount of flies available to phalaropes (Herbst & Bradley 1993, p. 200).

Figure 38: *Tens of thousands of Wilson's phalaropes staging at Mono Lake in 2021. Photo by Andew Youssef.*
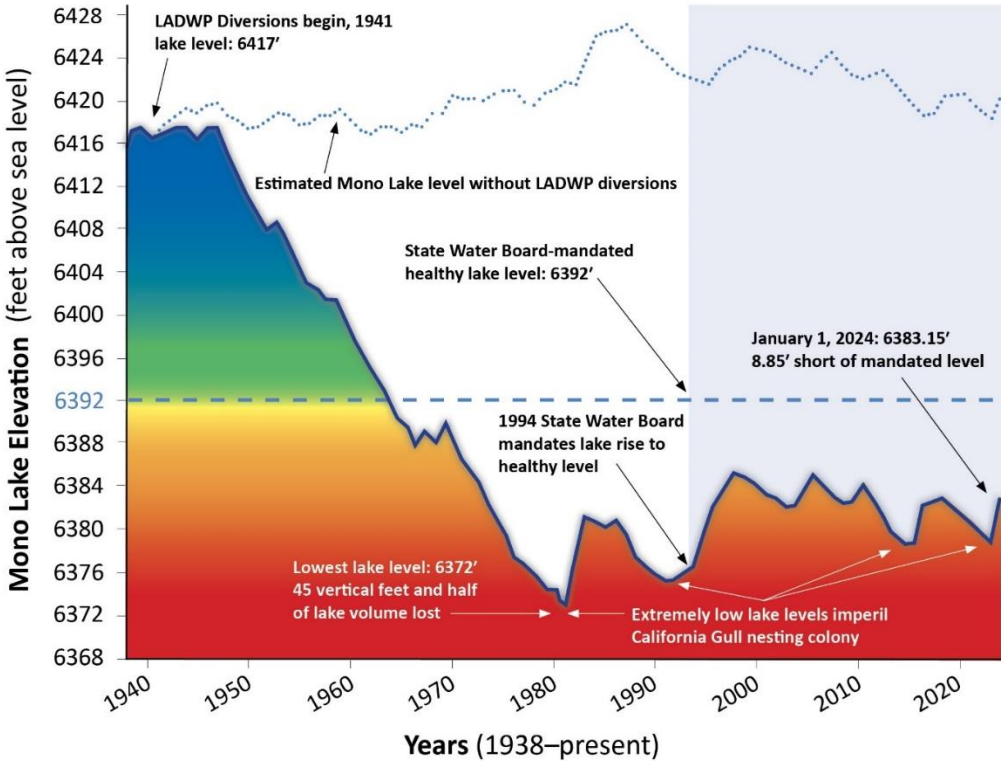
Fortunately, water diversions have been curtailed and Mono Lake's ecosystem did not collapse. Mono Lake was the subject of an intense legal battle over its water, which resulted in the California Supreme Court ruling that the lake's "public trust" values must be protected, writing that the public trust is, "an affirmation of the duty of the state to protect the people's common heritage of streams, lakes, marshlands and tidelands," (Blumm & Schwartz 1995, p. 707). In 1994, after years of analysis, the California State Water Resources Control Board amended Los Angeles' water right license in order to protect public trust values by mandating an average water elevation management level of 6,392' above sea level (Loomis 1995, p. 270-271). This lake level was decided upon in order to balance water use with protection of multiple resource values including lake ecosystem health, birds, air quality, and recreational and aesthetic values, and to ensure that the lake could be resilient during prolonged drought conditions (California State Water Resources Control Board 1994, p. 1). Since the 1994 decision, the lake level has risen from a low of 6,372' in 1982, but in 2022 the lake level remained 13 feet below the mandated management level (the lake was at 6,378.4 in December 2022; California State Water Resources Control Board 2023, p. 3, Mono Basin Clearinghouse 2023, *entire*). Since reaching a post-diversion high of 6,385.1 in 1999, the lake level has fluctuated in the range of 6,378'–6,384' for >20 years (Mono Basin Clearinghouse 2023, *entire*). Model-based predictions that the lake would reach the mandated 6,392' lake level in 20 years after the 1994 decision were not realized, probably due to mismatches between the 50-year period the models used (a relatively wet period) and the observed precipitation and evaporation in the last 30 years (McQuilkin 2023, p. 4).

Despite the failure of Mono Lake to reach the management level in the expected 20-year timeline, water diversions to Los Angeles have not been adjusted (McQuilkin 2023, p. 7). Modeling using updated data indicates that if diversions to Los Angeles had been completely halted, Mono Lake would now stand at approximately 6,390', just two feet shy of the mandated management level (McQuilkin 2023, p. 6). The same models indicated that given observed climate conditions, Mono Lake will not rise to the 6,392' elevation management level if the current volume of water diversions to Los Angeles continue at any level (McQuilkin 2023, p. 5). The city of Los Angeles continues to divert freshwater inputs to Mono Lake under its 1994 water license (16,000 or 4,500 acre-feet a year, depending on whether the lake is above or below >6,380', respectively; or zero acre-feet below 6,377' [California State Water Control Board 2023, p. 2-3]). The California State Water Board convened a public workshop in February 2023 to discuss the current status of Mono Lake levels, drought conditions, and ongoing diversions from its tributary streams, and the fact that the lake has not risen to the mandated management level in the timeframe that the board expected. There was public comment urging the State Board to hold its planned public hearing to modify Mono Lake diversion criteria to achieve the management lake level. To date, while the State Water Board has affirmed it will hold a hearing and that it has the authority to "reexamine past water allocation decisions" it has not committed to a date (California State Water Control Board 2023, p. 4)

With water diversions continuing at current levels, Mono Lake is expected to remain below the 6,392' management level (**Fig. 39;** McQuilkin 2023, p. 5), and will continue to be vulnerable to the extreme, prolonged drought periods that climate models predict will occur more frequently in the region with climate change (Seager et al. 2007, p. 1183-1184, Ficklin et al. 2013, p. 128). Current climate models predict that by the end of the 21st century, the Mono Basin will have significantly reduced streamflows, increased evapotranspiration, increased occurrence of drought years (Ficklin et al. 2013, p. 128), and increased air- and lake-temperatures (Schneider et al. 2009, p. L22402). There is a high level of uncertainty about changes to precipitation in the Mono Basin, and it's not clear that there will be a trend either upward or downward in total precipitation (Bedsworth et al. 2018, p. 25). However, precipitation patterns are predicted to be more volatile, with alternating periods of drought and unusually heavy precipitation (*Ibid*). Nonetheless, climate change factors combined are predicted to lead to less available water in the watershed (Ficklin et al. 2013, p. 128). During prolonged drought, the lake level could still drop substantially and degrade phalarope habitat due to increased salinity's impact on invertebrates and tufa exposure reducing substrates for alkali flies (Dana & Lenz 1986, p. 431, Dana & Herbst 1977, p. 69, Herbst & Bradley 1993, p. 198, Herbst & Bradley 1993, p. 200). Since the 1994 water board decision, Mono Lake has twice decreased to below 6,380' for extended periods, reaching 6,377' in late 2015 and early 2023 (McQuilkin 2023, p. 4-5). Below 6,380, alkali fly populations are expected to decrease, as salinity increases and as submerged tufa substrate is rapidly lost with surface drops between 6,380' and 6,370' (Herbst & Bradley 1993, p. 196). Major gains in lake elevation achieved during the extraordinarily wet winter of 2022-2023 are expected to be gradually lost if water diversions continue at current levels, but would be retained if water diversions are adjusted to assure timely achievement of 6,392' (**Fig. 40;** McQuilkin 2023, p. 5-6).

Figure 39: *Observed Mono Lake surface elevation (black line with color fill), and projected elevation without water diversions (dotted line). The court-mandated lake elevation of 6,392 is shown in a dashed line. Modeled lake levels are based on the Mono Lake Committee's modeling results. Figure from McQuilkin 2023, p. 6.*

Without DWP diversions, Mono Lake would be rising
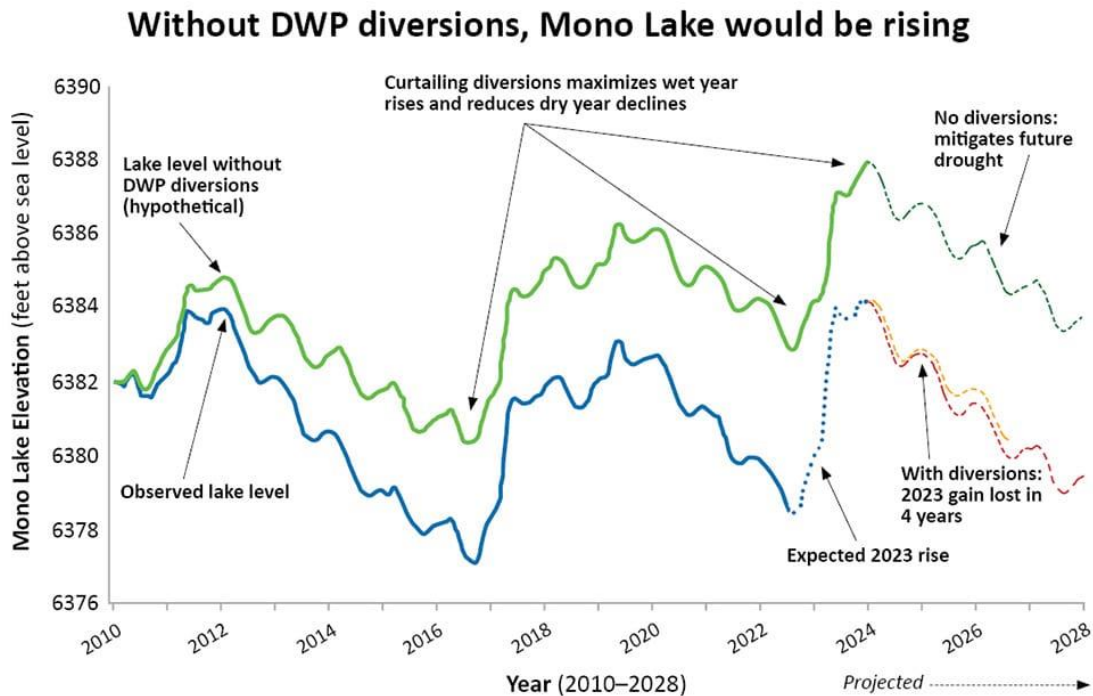
***Figure 40:** Mono Lake observed and projected lake levels, 2010-2028, from McQuilkin 2023, p. 5. Analysis shows that over time, the impact of DWP's stream diversions significantly affects Mono Lake's level. The wet winters of 2011 and 2017 resulted in dramatic lake rises, which were quickly erased by subsequent periods of drought and continued stream diversions. In summer 2023, a record lake rise was expected (blue dotted line), but this graph demonstrates how droughts similar to those experienced in the last decade (dashed red line: 2012–2015 drought; dashed yellow line: 2020–2022 drought) would drop Mono Lake back down to dangerously low levels. Green line shows hypothetical lake levels if Los Angeles DWP stopped all stream diversions beginning in 2010—demonstrating that curtailing diversions creates enough buffer to sustain the lake through extended drought.*

Another threat is that at its current unnaturally low lake levels, large inputs of freshwater into the lake during periodic wet winters can cause prolonged periods of fresh and salt-water "meromixis," i.e. reduced mixing between the lake's denser, saltier bottom and lighter freshwater surface. This can reduce primary productivity in the lake (typically affecting brine shrimp more than alkali flies; Jellison et al. 1998, p 708-710). At its current lake level, Mono Lake remains vulnerable to meromixis but at 6,392' is expected to be more resilient to large freshwater inputs (Jellison et al. 1998, p. 709).

Despite its precarious status, Mono Lake is the best protected saline lake staging site for Wilson's phalaropes, and the only one with legal protections guaranteeing a minimum water inflow, requiring a management level, and providing strong governance overseeing compliance (Blumm & Schwartz 1995, p. 711-712). Because of the protection of the inflow of water to Mono Lake, it will likely become increasingly important for Wilson's phalaropes as other sites decline (Carle et al. 2022, p. 17-18). For example, when Lake Abert desiccated in 2021, there was an increase of approximately 30,000 Wilson's phalaropes at Mono Lake that coincided with

the habitat loss at Abert (Carle et al. 2022, p. 17-18). However, even a healthy Mono Lake cannot be expected to provide enough habitat or food for the entire population of Wilson's phalaropes, because Mono Lake is only approximately 4-5% of the surface area of GSL (**Table 4**).

d. Other Staging Sites

Lahontan wetlands: Originally encompassing some 70,000 ha of wetland, the Lahontan Valley wetlands in central Nevada (39.593° N, -118.509° W) shrunk throughout the 20th century due to upstream water diversion, mainly for irrigated agriculture (Jehl 1994, p. 262, Lemly et al. 2000, p. 489). From 1905–1987, the complex lost 84% of its wetlands and experienced significant water quality declines (Hoffman et al. 1990, p. 5, 29). Before agricultural development in the 1980s, 60,703 ha was a "representative" value of the extent of the wetlands, but wetlands shrank to 10,119 ha by 1993 (Kerley et al. 1993, p. 11-13, Senner et al. 2021, p. 96). In 1990, the Truckee-Carson Settlement authorized acquisition of enough water to maintain an average of 10,118 ha of wetland in the Lahontan Valley, but to date water rights acquired are sufficient to maintain only ~4,856 ha (Senner et al. 2021, p. 96). Despite these water acquisitions, the wetlands at the Lahontan Wetlands complex have been shrinking as the site has received less water due to upstream uses and drought (Jehl 1994, p. 262, Senner et al. 2021, p. 96). Comparing the periods 1984–1999 and 2000–2018, the extent of surface water in the Carson Sink (one part of the Lahontan wetlands) declined 43% in wetlands and 90% in lakes (Donnelly et al. 2020, p. 2049).

The Lahontan Wetlands are a case in which a historically important site for Wilson's phalaropes appears to have been degraded by wetland loss and degradation of water quality (Hoffman et al. 1990, p. 5, Lemly et al. 2000, p. 489) to the point that an order of magnitude fewer phalaropes now uses it compared with the 1980s (Neel & Henry 1996 p. 17, Senner et al 2021, p. 93). There have been recent calls for improved wetland management for shorebirds in the Lahontan Valley (Senner et al. 2021, p. 108-109). However, observed and predicted climatic trends are toward warmer temperatures and increased aridity in the Great Basin region (Dettinger et al. 2015, p. 2088, Snyder et al. 2019, p. 6, Hall et al. 2023, p. 13-14), so it is questionable that in the future the Lahontan Valley will receive sufficient water to restore greater amounts of wetland acreage. The State of Nevada recently gained ownership of Carson Lake, and it is now designated as a Wildlife Management Area and a management plan is in development (Senner et al. 2021, p. 108). It is unclear what priority will be given to the seasonal water needs of shorebirds, including phalaropes, versus management for waterfowl and, especially for waterfowl hunting. (Senner et al. 2021, p. 108).

San Francisco Bay: Since the 1850s, development caused the loss of 79% of San Francisco Bay's (SFB's) tidal wetlands and 42% of tidal mudflats, including widespread impoundment of wetlands into commercial salt production ponds (Monroe et al. 1999, p. 35, Goals Project 2015, p. 14). The salt ponds were used extensively as foraging and roosting habitat by waterbirds, including migratory phalaropes (Jehl 1988, p. 50, Warnock et al. 2002, p. 89., De La Cruz et al. 2018, p. 84). The ponds created hyper-saline wetland habitat on which phalaropes specialize, and hosted alkali flies and brine shrimp (Warnock et al. 2002, p. 87). Beginning in 2003, government

agencies began restoring 6,110 ha of former salt ponds to a mosaic of tidal marsh and ponds managed for wildlife (De La Cruz et al. 2018, p. 5). This restoration will likely reduce phalarope foraging habitat because lower-salinity wetlands in restored salt marsh do not host invertebrates like brine shrimp that phalaropes specialize on, and restored areas can become more vegetated, discouraging shorebird use (Warnock et al. 2002, p. 90).

Phalarope numbers in SFB declined by 98% from the 1980s to present but the drivers of this severe decline are unclear (LaBarbera et al. 2023, p. 2, 26-27). The decline mostly occurred prior to the year 2000 (LaBarbera et al. 2023, p. 20), indicating that restoration of former salt ponds that began in 2003 was not the primary driver of that historical decline. However, 2022 phalarope counts have further declined 77% since 2005–2007 (Burns & Van Schmidt 2023). This decrease largely occurred from 2007–2009, shortly after salinity was first reduced in hypersaline ponds to transition them away from salt production, but before most tidal marsh restoration had occurred (LaBarbera et al. 2023, p. 28).

Taken together, available evidence suggests the collapse of a staging area for Wilson's phalarope that once rivaled Mono Lake in importance. South San Francisco Bay as a staging area represents a unique habitat type that faces different set of challenges than the remaining primary staging areas in North America, which are mostly large perennial saline lakes. If plasticity in choice of staging area exists, then restoring and maintaining quality phalarope habitat in South San Francisco Bay could be a refuge for migrating birds to create long-term resiliency.

Saskatchewan Saline Lakes: Land ownership and protection at the Chaplin complex is a patchwork of private and public lands, with various conservation and protection statuses (**Fig. 41**). IBA and WHSRN designations bring attention and management plans, but neither provides direct protection. Old Wives Lake is designated a Migratory Bird Sanctuary, which does not directly protect habitat, but protects birds from hunting (Schmutz 2000, p. 6). Chaplin Lake is designated a Heritage Marsh and its water is managed collaboratively between a mineral company and conservation groups (Schmutz 2000, p. 8). Reed Lake's natural source of water inflow is developed upstream for irrigation and only excess flows that cannot be stored and irrigation runoff returns feed the lake (Schmutz 2000, p. 13).

Figure 41: *Breeding habitat of Wilson's phalaropes in Saskatchewan, Canada, characterized by agricultural lands interspersed with small ponds. Photo by Ryan Carle.*

Though source waters are managed and habitat is not completely protected, it appears that the main threat to these sites is desiccation due to climate change and agricultural pollution impacting water quality, rather than water diversion (Schmutz 2000, p. 55-57). The Chaplin complex is dynamic, with water levels changing based on naturally variable conditions (Beyersbergen & Duncan 2007, p. 5). Old Wives Lake was dry during the 1940s, and Chaplin, Reed, and Old Wives Lakes were dry or mostly dry during droughts in the late 1980s (summarized by Beyersbergen & Duncan 2007, p. 5). Climate models indicate that in the region of southern Saskatchewan, temperatures will increase significantly in the 21st century (Ballard et al. 2014, p. 25, Shepherd & McGinn 2003, p. 305). For the entire Prairie Pothole region of the north-central U.S. and southern Canada, temperatures are predicted to rise by 3-4 °C by 2060 (Ballard et al. 2014, p. 20) and for southern Saskatchewan maximum temperatures under a doubling in atmospheric CO_2 are predicted to rise by 2.9–5.2 °C (depending on the model type; Shepherd & McGinn 2003, p. 305). Average temperatures from 1963-1998 in southwestern Saskatchewan rose by 2.9 and 2.2 °C (winter and spring, respectively; Cutforth et al. 1999, p. 346).

Most models also predict climate change to result in increased precipitation in the region (Hammer 1990, p. 325, Shepherd & McGinn 2003, p. 301, Ballard et al. 2014, p. 25). In the entire Prairie Pothole region, maximum increases in precipitation were projected to be 10% in winter and spring, with no increase in summer (Ballard et al. 2014, p. 25). However, increased temperatures will increase evapotranspiration, which is projected to result in negative changes in summer surface water (Ballard et al. 2014, p. 25).

Saline lakes in Saskatchewan increase in salinity as water levels decrease, whether over the course of the summer or during drought (Hammer 1990, p. 321). Thus, lakes would become more saline if increased evapotranspiration results in a net negative water balance. A study of

paleolimnology in the region concluded that the historical meteorological record does not capture the full range of climatic variability in southern Canadian prairies, and that significant periods of greater and lesser aridity have occurred in the last 2,000 years compared to the last 100 (Lemmen et al. 1997, p. 127). That study found that a period between 5,000–7,000 BP was the best historical analog to projections of future climates (Lemmon et al. 1997, p. 127). During that period, the regional water table was 4 m below ground and the vast majority of surface water was lost (Lemmen et al. 1997, p. 127). While the saline lakes of Saskatchewan are not imminently threatened by water diversion, their future is by no means ensured, because of the impacts of climate change (Schmutz 2000, p. 55). These lakes also cannot act as replacement habitat for GSL and Lake Abert. They collectively provide only ~7% of the surface area of GSL, and the numbers of phalaropes that currently use them is far less than those that use GSL and Lake Abert (**Table 4**).

Table 4: Surface areas, threats, and percentages of Wilson's phalarope population at North American staging sites.

Site	Surface area (km^2)[a]	Year of max (bolded) and min (italics) surface area on record	% of surface area compared to GSL[g]	Threats[h]	Max. % adult WIPH population at site[i]	Max % world WIPH population at site[j]
GSL	4,402, *6,474*	**1987[b]**, *2022[c]*	NA	Water diversion, mainly for agriculture, climate change	81%	60%
Lake Abert	**1-165**, 148, *259*	**1958[d]**, *1926, 1931, 1933, 1934, 2014, 2015, 2021, 2022*	0-6	Water diversion for agriculture, climate change	31%	23%

Mono Lake	179, *210*	**1941[e]**, *1982*	4-5	Water diversion for urban use makes lake vulnerable to drought and climate change	13%	9%
Chaplin complex	339	No data, *mostly dry during drought in the late 1980s*[f]	7	Stable but vulnerable to climate change	5%	4%
Lahontan Wetlands	**101**-**607** *1,943*	1993	2-44	Habitat mostly already lost to water diversion mainly for agriculture	9%	7%
San Francisco Bay salt ponds	**140**	No data	3	Salt pond habitat already mostly lost to salt marsh restoration & development	5%	4%

[a] Non-italicized values are from Conover & Bell 2020, p. 241-242, summarizing "commonly reported" values. Italicized values are from Wilsey et al. 2017, p. 29., who used a method that resulted in larger values than commonly reported. Bolded range value for Lake Abert is from Larson 2023, p. 87. Bolded range for Lahontan Wetlands is from Senner et al. 2020, p. 96 and Kerley et al. 1993, p. 11-13. Bolded value for San Francisco Bay salt ponds is from De La Cruz et al. 2018, p. 5. Note that saline lake levels tend to fluctuate dynamically, and these values are not constant.

[b]Wurtsbaugh et al. 2017, p. 3
[c]USGS 2023, *entire*
[d] all values from Larson 2023, p. 87.
[e] all values from McQuilkin 2023, p. 6
[f] Beyersbergen & Duncan 2007, p. 5
[g] Percents are based on dividing high and low ranges for each site by 4,402 km^2 value of GSL
[h]See Threats section for each site for summary and citations of threats
[i] Based on 741,000 adults, Jehl 1988, p. 50-51; see maximum totals of phalaropes for each site in **Table 3** of this document
[j] Based on 1,000,000 individuals estimated in 2020 survey of South American habitat, Castellino et al. 2024, p. 15; see maximum totals of phalaropes for each site in **Table 3** of this document

e. *Central flyway and Spring migratory sites*

Major spring migratory stops such as Cheyenne Bottoms and Quivera National Wildlife Refuge in Kansas, and freshwater playas and saline lakes in northwest Texas and New Mexico are also experiencing declining water levels because of human water use. At Cheyenne Bottoms, 20,000–70,000 acre-feet of water from the Arkansas River were transferred to the wetland in the 1960s, but this dropped to around 4,000 acre-feet in the 1980s and only 483 in 1985 (Zimmerman 1990, p. 34). This decline was driven by withdrawal of water for irrigated agriculture (Duram 1995, p. 5). Since the year 2000, Cheyenne Bottoms has repeatedly dried up completely during drought periods (Aber et al. 2016, p. 33). Quivera NWR has experienced similar problems, with reduced water availability and competition for water with irrigated agriculture, which has also overdrafted the groundwater aquifer (**Fig. 42;** Llopes-Jensen 2023, *entire*). In 2023, USFWS agreed to continue talks about solutions rather than mandate reduced groundwater pumping (Llopes-Jensen 2023, *entire*).

Figure 42: *Dry wetland at Quivera NWR, Kansas in November 2022. Photo by Celia Llopis-Jepsen/Kansas News Service, from (Llopes-Jensen 2023).*

Freshwater playas and saline lakes in Texas and New Mexico are also threatened by reduced water availability. Wilson's phalaropes use both habitats during spring and fall migration but occur more frequently at the saline lakes (Andrei et al. 2006, p. 381). The approximately 45 saline lakes in this region are sustained by discharge from springs fed by the Ogallala Aquifer, whereas freshwater playas in the region are fed only by precipitation (Brunne 1981, p. 38, Andrei et al. 2009, p. 138). The Ogallala Aquifer is heavily overdrafted (Brunne 1981, p. 42, Terrell et al. 2002, p. 3, Steiner et al. 2021, p. 2), and by the 1970s flows of these springs were reduced and many lakes were perpetually dry (Brunne 1981, p. 38, 45). In the 2000s, springs flows continued to decline because of declining aquifer levels, resulting in increasing salinization of saline lakes and shorter hydroperiods (Andrei et al. 2008, p. 251). Saline lakes in the region often hosted Artemia and *Ephydra* flies, but lakes with salinities of >120 g/L had no invertebrates present (Andrei et al. 2008, p. 251). Andrei et al. (2009, p. 146) noted that, "Declining water tables, reduced freshwater spring flow and increased salinity could reduce the importance of saline lakes in the SGP as migration stopovers because the foraging strategies used by shorebirds may become ineffective due to reduced availability of invertebrates." The best solution for conserving these habitats would be by conservation of the waters of the Ogallala Aquifer (Andrei et al. 2009, p. 146).

Can fresher terminal lakes provide phalarope habitat if they shrink and become hyper-saline?

Less saline terminal lake habitats such as Walker Lake, Nevada (38.701°N, -118.721°W) and the Salton Sea, California (33.351°N, -115.870°W) are also declining in water levels and stand out as major water-bodies that could become hyper-saline in the future, providing conditions capable of supporting alkali flies and/or brine shrimp (Herbst et al. 2013a, p. 881, Hart et al. 1998, p. 150, Bradley et al. 2022, p. 13). In 2007, Walker Lake, located approximately 580 km east of GSL, had approximately 2–3% the surface area of GSL (4,402 and 6,474 km^2; GSL from Conover & Bell 2020, p. 241, Walker Lake from Herbst et al. 2013, p. 63). Walker Lake's salinity in 2012 was 19.73 g/L, far less saline than most important phalarope habitats (Herbst et al. 2013a, p. 878). To date, the largest Wilson's phalarope count on record for Walker Lake was approximately 100,000 birds in September 2016 (**Table 3;** Castellino et al. 2024, p. 21). Such high numbers of phalaropes do not regularly occur at Walker Lake, which does not host *Artemia* or large populations of alkali flies (Herbst et al. 2013b, p. 66). As Walker Lake's water level declines and salinity becomes more ideal for alkali flies, the substrata will increasingly be composed of fine sediments and mud, which are unfavorable for alkali fly pupal attachment (Herbst et al. 2013, p. 70). This could mean that even if Walker Lake were to become hyper-saline, it may not be a highly productive source of brine flies for phalaropes. Furthermore, managing Walker Lake as a hyper-saline lake would come at the sacrifice of many species that rely on the lake in its naturally less-saline state, including endemic fishes (Jellison & Savon 2005, p. 137, USFWS 2013, p. 3). Permanent solutions to manage Walker Lake's water level would likely prioritize the species reliant on its natural less-saline waters (USFWS 2013, p. 7, Herbst et al. 2013a, p. 883). If Walker Lake becomes hyper-saline, it would be because management solutions fail to maintain sufficient freshwater inputs (Jellison & Sevon 2005, p.

139) meaning that in the long-term Walker Lake also would be threatened with becoming too saline or with drying up completely (Donnelly et al. 2020, p. 2053)–thus it cannot be considered "safe" future phalarope habitat.

The Salton Sea is approximately 14–21% of the surface area of GSL and is located 890 km southwest of GSL (Conover & Bell 2020, p. 241, Wilsey et al. 2017, p. 29). Its salinity level is approximately 44 g/L (UCANR 2023, *entire*). To date, the greatest count of Wilson's phalaropes that we are aware of at the Salton Sea is 3,065 birds in August 1999 (Shuford et al. 2002, p. 264). In 2018, freshwater inputs were terminated to the Salton Sea, and it began to rapidly increase in salinity, but alkali flies and *Artemia* were not present as of 2022 (Bradley et al. 2022, p. 12). Increasing salinity had not resulted in abundant *Artemia*, which may be limited by population increases of predatory water boatmen (*Trichocorixa* sp.) following rapid decline of *Tilapia* spp. fish populations (Bradley et al. 2022, p. 13). Increasing salinity at Salton Sea could eventually result in colonization by alkali flies and brine shrimp (Shuford et al. 2002, p. 272), but it would be at the loss essential habitat for fish-eating birds and island-nesting bird habitat (Cohn 2000, p. 295, Shuford et al. 2002, p. 255, Jones et al. 2019, p. 6-9). With many competing wildlife values at the Salton Sea, the needs of phalaropes would have to be balanced with those of many other species (Jones et al. 2019, p. 9). Furthermore, water quality is a major concern at the Salton Sea, which receives most of its water from agricultural runoff (Schroeder et al. 2002, p. 23). The Salton Sea has excessive levels of selenium (Schroeder et al. 2002, p.43, Vogl & Henry 2002, p. 53), which can be carried up the food-web by invertebrates, including brine flies (Wurtsbaugh 2009, p. 6, Ohlendorf et al. 2009, p. 28). When ingested by birds, selenium can cause serious physiological damage, reproductive impairment, and mortality (reviewed in Ohlendorf & Heinz 2011, p. 696). Given these factors, even if in the future the Salton Sea became hyper-saline, it would not be safe long-term replacement habitat for Wilson's phalaropes. Like Walker Lake, the Salton Sea cannot be expected to sustain Wilson's phalarope populations if its transition to hyper-saline conditions is a symptom of insufficient water inputs (Doede & DeGuzman 2020, p. 8, Jones et al. 2019, p. 4).

f. *Wintering habitat—South America*

The threat of habitat loss to Wilson's phalaropes is not limited to staging habitat. Their wintering habitat and breeding habitat are also threatened with current and/or impending habitat loss, primarily from water use in mining activities, and from climate change.

In South America, the core wintering habitat of the species falls within the "lithium triangle" in the high Andes of Argentina, Bolivia, and Chile (**Fig. 43;** Hurlbert 1984, p. 49, Gajardo & Redón 2019, p. 1-2, Heredia et al. 2020, p. 222, Castellino et al. 2024, p. 33-34). This area could host up to 68% of the world's lithium salt brine reserves (Sticco et al. 2021, p. 7). Lithium mining development has increased rapidly in the region as demand for lithium-ion batteries for electric cars increases (Martin et al. 2017, p. 171, Garcés & Alvarez 2020, p. 193). World lithium production has increased more than 8-fold from 1995–2017 and is expected to increase by 10% per annum or more in the next two decades (Maxwell & Mora, 2020 p. 60).

The primary threat to phalarope habitat from lithium mining is that the process uses immense quantities of groundwater, which can result in the lowering of aquifers in the extremely

arid altiplano region (Alam & Sepulveda 2022, p. 183). To mine lithium, brine is pumped from beneath the surface of salares (dry ancient lake basins) to evaporating pools where the mineral is concentrated (Flexer et al. 2018, p. 1191). Approximately 105,000–132,000 gallons of brine are evaporated to produce 1 ton of lithium carbonate, and an average extraction facility producing 20,000 tons of lithium per year is estimated to evaporate approximately 7.6 million cubic meters of water per year (Flexer et al. 2018, p. 1200, Marconi et al. 2022, p. 838). Depending on the technology used and lithium concentrations, approximately 21,000–37,000 gallons of freshwater are used per ton of lithium production (Marconi et al. 2022, p. 838). This water use occurs in the extremely arid Andean altiplano region, which annually receives only 300-600 mm (11.8-24.6 inches) of rain in its eastern region and 60-150 mm (2.4-6.2 inches) in its western region (Lupo et al. 2018, p. 78). The high arid plateaus of the altiplano have a naturally negative water balance, where annual evaporation exceeds annual precipitation and groundwater discharge often exceeds recharge (Marconi et al. 2022, p. 830). There is evidence that inter-basin groundwater flow is important in the modern hydrologic balance and that fossil groundwater (i.e. from 100 to >10,000 years age) is the dominant water discharge in wetlands in the region (Marconi et al. 2022, p. 830). Thus, water balances in the region are delicate and surface water that provides habitat for phalaropes is likely to be affected by the intensive water use from mining. For example, sustainable brine water pumping (e.g., that does not dry up wetlands) by mining in the altiplano basin of the Salar de Huasco was estimated at 52.8 gallons per second for 25 years (Acosta & Custodio 2008, p. 33). However, current brine water use by mining in the nearby Salar de Atacama is estimated at over ten times that, at 565.8 gallons per second daily (Gajardo & Redón 2019, p. 4).

Figure 43: The Salar de Pujsa in the Chilean altiplano. The Salar de Pujsa is a Ramsar site located in a Chilean National Reserve, whose water is nevertheless threatened by lithium mining development. Photo by Ryan Carle.

The full scope of the impacts of lithium mining in the altiplano is not yet understood, and there is a lack of baseline information about mining's effects on aquifers, despite the active development of mining projects (Marchegiani et al. 2019 p. 36-37, Gajardo & Redon 2019, p. 6, Marconi et al. 2022, p. 845-846). Observed negative impacts to date include a direct negative correlation between lithium mining and abundance of two species of flamingos, and a decline in winter surface water in the Salar de Atacama in Chile (Gutierrez et al. 2022, p. 6-8). Evaluation of the precise impacts of lithium mining on phalaropes is also difficult because there are limited data on phalarope use of the area, and on the number and scope of lithium development projects. However, approximately 25% of all Wilson's phalaropes recorded in a survey across South American wintering habitat in South America were in the "lithium triangle" (Castellino & Lesterhuis 2020, p. 8-9, Castellino et al. 2024, p. 34). Sites within the lithium triangle with records of >1% of the world population of Wilson's phalaropes are the Salar de Surire (Chile), Laguna de Los Pozuelos and Laguna Palar (Argentina), and Lago Poopó, Lago Hedionda Norte, Laguna Uru Uru, Laguna Alalay, Laguna Saquewa, Laguna Chulluncani, Lagunas Pastos Grandes, and Laguna Loromayu (Bolivia; **Table 3;** Castellino et al. 2024, p. 20-24). However, phalarope use of the area is dynamic and there have been few surveys of the region (Castellino et al. 2024, p. 18, 34), so other unidentified sites may also be important to phalaropes within the lithium triangle. A preliminary analysis of overlap of lithium development and important shorebird areas within the lithium triangle used satellite imagery and GIS methods to identify 18 active lithium mining projects in the region (FARN, Fundación Tantí, Fundación Yuchán, WHSRN & Comunidad Sustentable unpublished data). However, many more projects are approved and in early-phases of development; a literature review identified 101 approved projects in the region, many of which are likely to be actively developed in the near future (FARN, Fundación Tantí, Fundación Yuchán, WHSRN & Comunidad Sustentable unpublished data). The same analysis identified 22 wetlands hosting >1% of world populations of a shorebird species (FARN, Fundación Tantí, Fundación Yuchán, WHSRN & Comunidad Sustentable unpublished data). Three of those shorebird areas overlapped directly with active lithium mining operations, and a majority of shorebird sites were only a short distance from lithium claims in various stages of development (FARN, Fundación Tantí, Fundación Yuchán, WHSRN & Comunidad Sustentable unpublished data). Among active lithium mining operations, mining is in the prospecting stage at the Salar de Surire (PCM 2023, *entire*), which has the highest counts of Wilson's phalaropes in Chile (**Table 3;** Castellino et al. 2024, p. 22). The Salar de Atacama, Chile, hosts one of the largest active lithium mines in the world, as well as at least three areas with surface water with high counts of 300 to >700 Wilson's phalaropes each (Lagunas Barras Negros, Chaxa, and Puilar; CNAA Chile 2010a, 2010b, 2011, *entire*). Lithium exploration is occurring also at Laguna Negro Francisco in Chile, which has had counts of >100 Wilson's phalaropes (GCFA 2010, *entire*), but has been seldomly surveyed for shorebirds (Castellino et al. 2024, p. 15). In Argentina, lithium mining is being developed at Salinas Grandes (Jujuy province; Justicia Ambiental y Derechos 2020, *entire*), adjacent to Laguna Guayatayoc, which has high counts of >6,700 Wilson's phalaropes (Moschione 2020, *entire*).

Despite the information gaps, many observers have concluded that the environmental impacts to the region's wetlands and avifauna are likely to be severe in the near future, given the expected rapid growth of lithium mining in the altiplano and the absence of better governance,

regulation, and mitigation of the industry (Gutierrez et al. 2022, p. 8, Izquierdo et al. 2015, p. 55, Marconi et al. 2022, p. 845).

Laguna Mar Chiquita and lowland Argentina

The importance of wetlands in lowland Argentina has recently become better known, especially the outsized importance of Laguna Mar Chiquita (Castellino et al. 2024, p. 12). Laguna Mar Chiquita can be seen as the South American corollary to GSL, in size and numbers of phalaropes present (**Table 3;** Castellino et al. 2024, p. 12). Laguna Mar Chiquita was designated an Argentinian National Park in 2022, a major advancement in its conservation (Castellino et al. 2024, p. 35). Park status will improve management and awareness of conservation issues at Laguna Mar Chiquita, but it does not come with a water right (Castellino et al. 2024, p. 41). Laguna Mar Chiquita's water level has fluctuated greatly over time, and its water balance is not well understood (Bucher & Curto 2012, p. 128).

Currently in 2023, Laguna Mar Chiquita's water level has been falling, likely related to sustained drought in the region and diversions from its freshwater inputs (Castellino et al. 2024, p. 36). It is fed by three rivers, the largest of which is the Rio Dulce, which flows through a major metropolitan area (Córdoba) and a major agricultural region before reaching the lake (Castellino et al. 2024, p. 36). Water diversions from the Rio Dulce are not well measured or managed but are increasing (Bucher & Curto 2012, p. 136). Agricultural pollution impacting water quality is also a concern, with high levels of DDT, PCBs and other chemicals present in the lake (Ballesteros et al. 2014, p. 79, Castellino et al. 2024, p. 38). Overall, the conservation outlook for Laguna Mar Chiquita is positive in terms of management of the actual site, but like the other saline lakes used by Wilson's phalaropes, its water future is precarious (Castellino et al. 2024, p. 36). Other sites in lowland Argentina are known to be important for phalaropes, such as Lago Epecuén (Castellino et al. 2024, p. 12), but there are few data or information available on their water situations and management.

B. Disease or Predation

There is no evidence of disease or predation negatively affecting Wilson's phalarope populations.

C. Overutilization for Commercial, Recreational, Scientific, or Educational Purposes

Overutilization of Wilson's phalarope for commercial, recreational, scientific, or educational purposes is not known to be a threat at this time.

D. Inadequacy of Existing Regulatory Mechanisms

Despite its historic and current habitat loss, 75% decline in population since the 1980s, and foreseeable catastrophic future habitat loss at GSL and Lake Abert, the Wilson's phalarope is presently without federal protection under the U.S. Endangered Species Act and receives minimal protection under the Migratory Bird Treaty Act ("MBTA"). The MBTA protects against

the direct take of birds, nests, and eggs, but the act provides no authority for protection of habitat or food sources (16 U.S. Code Chapter 7 Subchapter II). Furthermore, the legislative history of the MBTA supports differing interpretations of the nature and scope of the MBTA's taking prohibitions, and consequently, courts have been split on when and if to apply the MBTA (Congressional Research Service 2022, p.1).

Wilson's phalarope was included in the U.S. list of Birds of Conservation Concern 2002 due to widely reported declines (USFWS 2002, p. 73). However, the species has not been included in the more recent Birds of Conservation Concern lists in 2008 or 2021 (USFWS 2009, p. 70-71, USFWS 2021, p. 19). It is listed as a species of Least Concern in the U.S. Shorebird Conservation Plan (U.S. Shorebird Conservation Plan Partnership 2016, p. 7). Wilson's phalarope is identified as a priority species in the Intermountain West Regional Shorebird Plan, where its use of that region is ranked as "critically important" to the species (Oring *et al.* 2013, p. 45). In the Northern Plains/Prairie Potholes Regional Shorebird Conservation Plan, Wilson's phalarope is identified as a "species of concern" and ranked 4 (out of a scale of 1–5) for both regional and national conservation priority (Skagen & Thompson 2013, p. 12). However, these designations do not provide binding legal mechanisms for conservation of the species or its habitat.

At the state level in the United States, Wilson's phalarope is listed as Threatened in Minnesota (Minnesota Department of Natural Resources 2021, Appendix B, p. 5). Minnesota's Threatened designation provides legal protection against take of the species but does not cover sufficient geographical areas to address the range-wide threats to Wilson's phalaropes. Wilson's phalarope is listed as a Level I Species of Conservation Priority in North Dakota, South Dakota, and Wyoming (Butler et al. 2014, p. 27), but those listings are used to prioritize making conservation strategies for species and do not provide legally binding protection.

Designations in other countries: In Canada, Wilson's phalarope is included in the Canadian Shorebird Conservation Plan, where is listed as a species of Priority for Conservation in six management regions, but is not listed as Special Concern, Threatened, or Endangered at a national level ([Environment and Climate Change Canada 2019](#)). Thus, there are not legally binding mechanisms for the species' protection or protection of its habitat in Canada. Even if there were, the Canadian Species at Risk Act would not be able to address critical issues with staging sites in the U.S. or wintering sites in South America.

In South America, Wilson's phalarope is mentioned in the National Shorebird Conservation Plan for Argentina among shorebird species requiring special attention when planning and executing actions of habitat conservation and management at the site scale (Ministerio de Ambiente y Desarrollo Sostenbile de Argentina (MADS) et al. 2017, p. 25). However, Wilson's phalarope was not included as a focal species in Argentina's plan because it did not meet the criteria used, including not being listed as Threatened by the IUCN (MADS et al. 2017, p. 25). Nationally, in Argentina the species is considered Not Threatened (MADS et al. 2017, p. 25). In Peru, Wilson's phalarope is listed at national level as Least Concern (SERFOR 2023, p. 25). Wilson's phalarope is not listed as a threatened species in Chile (Ministerio de Medio Ambiente 2022, p. 24-25). No specific information about the species at national level is available for Bolivia and Paraguay.

Wilson's phalarope is listed as Least Concern in IUCN, and the population trend for the species identified as Increasing (BirdLife International 2023, *entire*). However, this may not reflect the current state of the population and needs review based on updated information from recent years. For example, that IUCN assessment does not include the 70% decline in abundance since 1980 reported by Smith et al. (2023, p. 8), the declining trends in the most recent range-wide analyses of BBS data (Ziolkowski et al. 2023, *entire*), declines observed since the 1980s at North American staging sites (Jehl 1988, p, 50-51, Jehl 1999, p. 40, Carle et al. 2023, p. 18), or the threat of habitat loss at GSL, Lake Abert, and the altiplano in the near future. The Wilson's phalarope could be qualified for Endangered listing by the IUCN based on the criteria of "an observed, estimated, inferred, projected or suspected population size reduction of ≥50% over any 10 year or three generation period..." (I.e. the reported past decline from Smith et al. 2023, p. 8) and "A population size reduction of ≥50%, projected or suspected to be met within the next 10 years or three generations..." (i.e. if GSL's ecosystem collapses, staging habitat for >50% of the world population would be destroyed; IUCN 2000, p. 18). Reevaluation of the IUCN status of the species is currently underway.

Wilson's phalarope was included as a focal species in the Midcontinent Shorebird Conservation Initiative (MSCI), the current development of which is led by the US Fish and Wildlife Service, the Conservation of Arctic Flora and Fauna through the Arctic Migratory Bird Initiative, and Manomet –WHSRN (MSCI 2024, *entire*). Focal species status in MSCI will provide a strategic framework for conservation of the species and potentially future funding for research and conservation. However, the MSCI is presently in a preliminary stage and has yet to be funded or implemented. Further, like the other initiatives listed here, the MSCI would not provide legally binding regulatory mechanisms for conservation of the species or its habitats.

Regulatory mechanisms for protection of specific sites:

Key sites for phalaropes have varying levels of conservation designation, but only Mono Lake has guaranteed water rights and a strong legal mechanism for protection of its water. Some key phalarope staging and wintering sites have been designated as Western Hemisphere Shorebird Reserve Network (WHSRN) sites of International (≥100,000 shorebirds annually or ≥10% of a species' world population) or Hemispheric Importance (≥500,000 shorebirds annually or ≥30% of a species' world population). Phalarope sites designated as of "Hemispheric Importance" to shorebirds by WHSRN are GSL, the Chaplin Lake complex, the Lahontan Wetlands, San Francisco Bay, and Laguna Mar Chiquita (WHSRN 2024, *entire*). Mono Lake is designated a site of "International Importance," specifically because of its importance to Wilson's phalarope (WHSRN 2024, *Mono Lake account*). WHSRN designations bring international attention to sites and encourage local conservation initiatives because landowners must agree to prioritize shorebird conservation and protect and manage shorebird habitat at the site as part of the designation process (WHSRN 2024b, *entire*). However, conservation efforts are voluntary, there is no legal mechanism for enforcement of agreements to protect shorebirds at each site, and WHSRN designations provide no legal authority to protect Wilson's phalaropes or their habitat.

Various sites of importance to phalaropes are also designated as Wetlands of

International Importance under the Ramsar Convention (Ramsar 2024, *entire*). These include San Francisco Bay/Estuary in the U.S., Laguna Mar Chiquita and Laguna de los Pozuelos in Argentina, and Salar de Aguas Calientes IV, Salar de Pujsa, Salar de Tara, Salar de Surire, and the Soncor hydrologic system of the Salar de Atacama in Chile (Gajardo & Redón 2019, p. 4, (Ramsar 2024, *entire*). The Ramsar Convention is an international treaty in which signatory countries agree to manage designated wetlands with "wise-use" principles aimed at conservation and are encouraged to adopt national wetland strategies and implement laws protecting wetlands (Ramsar Convention Secretariat 2010, p. 7). However, like WHSRN designation, Ramsar designation does not come with legally binding mechanisms for protecting wetlands or species. In the case of the South American altiplano, impacts to Ramsar sites are routinely ignored in environmental impact assessments for lithium mining (Marconi et al. 2022, p. 840).

Another symbolic conservation designation pertaining to some of important phalarope sites is that of Important Bird Area (Birdlife International 2024, *entire*). Sites listed as IBAs include the Mono Lake Basin, Lake Abert, GSL, and Lahontan Valley. However, like WHSRN and Ramsar designation, IBA designation provides only symbolic protection and increased visibility for conservation but provides no legal mechanism to protect habitats.

Various parts of GSL are protected as state parks, state wildlife management areas, waterfowl management areas, or migratory bird refuges. However, none of these designations encompasses the entire lake, nor do they provide legal protection of the freshwater inputs to the lake. Despite greatly increased public and legislative attention to GSL in 2022, GSL still has no legal water right and legally defined target management lake level. A Utah state legislative proposal introduced in 2023 to set a target lake elevation level of 4,198 feet was rejected in committee (Keddington 2023, *entire*). Measures that were enacted in the 2022 legislative session include creation of the Great Salt Lake Watershed Enhancement Trust (GSLWET) with an appropriation of $40 million to purchase temporary leases of water for the lake (GSLWET 2024, *entire*). Since its creation, the GSLWET has leased ~64,000 acre feet of water (https://www.gslwatertrust.org/our-impact). These efforts fall far short of procuring and sustainably maintaining the input of 2.5 million acre feet a year of water estimated to be needed to restore GSL and prevent its further collapse (Abbott et al. 2023, p. 13). The water trust is a positive step but currently provides insufficient funding to purchase sufficient water rights or modify water delivery infrastructure to GSL (there is currently $25.5 million left in the original $40 million fund; GSLWET 2024, *entire*). In addition, the state has created a quasi-governmental agency, the Utah Inland Port Authority, which is using public money to subsidize industrial development destroying wetlands, uplands and playa within the Great Salt Lake Basin (Burrage et al. 2023, p. 2-5).

Lake Abert is designated an Area of Critical Environmental Concern by the Bureau of Land Management. Though this designation provides management attention from the BLM, it does not protect Lake Abert's water inputs. Lake Abert has no legal water right, and the state of Oregon has declined to study water use in the Chewaucan Basin to understand the issues (Davis 2022, *entire*). Currently efforts are focused on discussion with upstream water users and potential purchase of water rights for the lake. There are no legally binding regulatory mechanisms for its protection.

Mono Lake is protected as a California State Natural Reserve and surrounded by the

federal Mono Basin National Forest Scenic Area. These designations protect the entire lake from development and other negative uses, but they do not protect the water inputs to Mono Lake, which are under the jurisdiction of the California State Water Resources Control Board. Based on the 1994 California State Water Board Decision, Mono Lake has legal protection including a mandated sustainable management level. There is a strong legal foundation for protection of Mono Lake's waters, which is incorporated into the licenses to divert held by the primary water rights holder, the City of Los Angeles. Despite this regulatory mechanism, Mono Lake remains far below the designated management level of 6,392 feet (McQuilkin 2023, p. 4). The California State Water Board has affirmed that it has the authority to "reexamine past water allocation decisions" and plans to hold a hearing to consider changes to diversions but has not scheduled a date at this time (California State Water Control Board 2023, p. 4).

Western Water Law:

Many of the drivers of declines of saline lakes today can be traced back to the development of water law in the western United States and the prior appropriation doctrine (PAD). The PAD, adopted in 1885, allows a person to receive individual water rights when they are the first to use the water for an apparently beneficial purpose (Benson 2012, p. 676, MacDonnell 2015, p. 229-230). First set at a federal level, the PAD was enshrined as law in many western states and state constitutions (Benson 2012, p. 676). PAD governs the control and uses of water in California, Utah, and Oregon among many other states (Gopalakrishnan 1973, p. 61). The PAD encouraged development of water and by the early 20th century most western rivers were fully appropriated—and often over-appropriated, with people holding rights to use more water than the typical flows (Benson 2012, p. 676, MacDonnell 2015, p. 232-233). An example of this is the over-appropriation of the Chewaucan River, in which, "essentially all the surface water can be legally extracted from the river, leaving nothing to flow into Lake Abert" (Moore 2016, p. 151). The PAD awards water rights to those with a specific beneficial use of the water that the law regards as "useful," which traditionally has included irrigation, mining, manufacturing, and hydropower–and not environmental uses (Gopalakrishnan 1973, p. 68, Benson 2022, p. 680). An example of the impact of this on the current day situation at saline lakes is that letting water flow into Great Salt Lake was not legally considered a beneficial use of water in Utah until 2022 (Miller 2022, *entire*). Water rights holders in Utah operated under a "use it or lose it" doctrine in which, if they did not use their water, they were at risk of losing their water right (Miller 2022, *entire*). In 2022, the Utah Legislature changed the law to consider water for Great Salt Lake a "beneficial use" (Miller 2022, *entire*). In the case of Mono Lake, the city of Los Angeles held the water right to the tributary streams, with no water allocated for the health of the lake. In 1983, the California Supreme Court ruled that the public trust doctrine's mandate to conserve the lake as a public resource allowed for a revision of the city of Los Angeles' water licenses to ensure protection of the lake (California Supreme Court 1983, *entire*). Though the public trust doctrine has been applied successfully in California to save bodies of water, the states of Nevada and Idaho have ruled that it does not dislodge established appropriative water rights (Ryan 2020, p. 200). A recent lawsuit filed in Utah is arguing that the Great Salt Lake falls under the Public Trust Doctrine and should be saved (Center for Biological

Diversity 2023, *entire*). That lawsuit has not yet had a ruling.

In South America, Laguna Mar Chiquita was protected as an Argentinian National Park in 2022 (Castellino 2022, *entire*). The majority of the water of Laguna Mar Chiquita is in the national park (Parques Nacionales de Argentina 2023, *entire*). Park designation provides protection of the area of the lake included in the park from direct threats and protects phalaropes while they are at the lake. Park status also provides increased visibility and symbolic leverage for conservation of the site, but it does not provide legal mechanisms for maintaining sufficient inflow of water to the lake, or for preventing upstream pollution from rivers that run into the lake.

Various sites used by phalaropes in the Andes are protected in national parks in Chile and Argentina, including Reserva Nacional Los Flamencos in Chile and Monumento Nacional Laguna de Los Pozuelos in Argentina. These park designations protect wetlands from direct development or destruction, but do not protect their aquifers from over-drafting for mining.

To conserve key habitat and prevent extinction Wilson's phalarope needs a comprehensive, coordinated recovery effort that is consistent across jurisdictional boundaries and can be used to protect water inputs to key staging sites. Given this need, listing under the ESA is appropriate and necessary.

E. Other Factors

1. Climate Change

"Our results point to a remarkably drier future that falls far outside the contemporary experience of natural and human systems in Western North America, conditions that may present a substantial challenge to adaptation."
Cook et al. 2015, p. 6

"Large and relatively stable salt lakes such as Mono and Great Salt probably take on added importance to grebes and phalaropes during droughts, when alternative habitats are unavailable."
Jehl 1988, p. 54

"The health of bird populations that use unstable habitats is to a large extent dependent on the availability of back-up sites that can be used when conditions change. Unfortunately, there is not much redundancy left in the saline and alkaline lakes of the west."
*Jehl 1994, p. 270*

Saline lakes are sensitive to even small changes in their water budgets, especially changes in evaporation and precipitation (Williams 2002, p. 161). The impacts of global climate warming, which are manifested through changes in temperature, evaporation and precipitation, are therefore expected to affect saline lakes more than freshwater bodies (Williams 2002, p. 161). Generally, when saline lakes experienced increased aridity, ephemeral saline lakes stay dry for longer and permanent saline lakes become smaller and more saline (Williams 2002, p. 161,

Wurtsbaugh et al. 2017, p. 1). Globally, observed trends and future predictions are that saline lakes are shrinking in size, and that in many cases declines in water level have sped up since the year 2000 (Williams 1996, p. 38, Williams et al. 2002, p. 164, Wurtsbaugh et al. 2017, p. 3, Foroumandi et al. 2022, p. 1-2). In most cases globally, the decline of saline lakes is driven by a combination of water diversion and increased aridity associated with climate change, with water diversion usually the larger and more immediate threat (Wurtsbaugh et al. 2017, p. 3-4).

In the interior west of North America, climate change is observed and predicted to result in increased temperatures, increased evaporation, decreased precipitation, and more frequent and longer droughts in the 21st century (Seager et al. 2007, p. 1181, 1183-1184, Hall et al. 2023, p., Snyder et al. 2019, p. 4-5. Cook et al. 2015, p. 6). The drying trend underway in western North America is predicted to be unlike any climate state seen in the instrumental record, and to result in severe future droughts longer than the multi-decadal droughts that occurred in the region from 900–1300 AD medieval warm period (Seager 2007, p. 1181, Cook et al. 2015, p. 6). During that period, the interior west region experienced a series of extreme multi-decadal droughts (Cook et al. 2014, p. 1016), which caused declines in water levels of Mono Lake of 15.5 m (Stine 1994, p. 546). A recent study found a >80% chance of muti-decadal drought during 2050–2099 that was robust across multiple models and moisture metrics, and that the mean state of drought will exceed the most severe megadrought periods in the medieval climate anomaly (Cook et al. 2015, p. 4-6). Indeed, megadroughts are already underway; the 2000–2021 period in southwestern North America had the driest 22-year soil moisture anomaly of the last 1,200 years, which was linked to anthropogenic climate change (Williams et al. 2022, p. 232).

The climate of the Great Basin region has already experienced significant warming, and a shortening of the annual hydroperiod (Hall et al. 2023, Schneider et al. 2009, p. 5, Haig et al. 2019, p. 3). Loss of saline lake water volume and wetlands in the Great Basin has accelerated in the 21st century (**Fig. 44;** Haig et al. 2019, p. 3, Donnelly et al. 2020, p. 2052). Overall, climate modeling for the interior west of the United States indicates a trend toward increased warming and drought resulting in major declines in surface water on wetland and lake systems already stressed by water diversion (Haig et al. 2019, p. 3, Donnelly et al. 2020, p. 2054). Across the region, this is likely to result in disappearance of smaller, more ephemeral wetlands used by phalaropes and to strain the hydrologic resiliency of the network of staging sites (Haig et al. 2019, p. 4, Donnelly et al. 2020, p. 2054). As stated by Donnelly et al. (2020, p. 2054), "Lake and wetland declines we identify in snowmelt watersheds may signal a loss of plasticity in migratory networks." When smaller and more ephemeral sites become dry, large perennial saline lakes are likely to increase in importance to phalaropes, for lack of other options (Jehl 1988, p. 54, Haig et al. 2019, p. 6). Furthermore, the loss of water volume at these lakes due to water diversion has eliminated natural buffers against climate extremes (Wurtsbaugh et al. 2017, p. 4, Haig et al. 2019, p. 6), such that increased warming, evaporation, and drought may put them past their ecological breaking points (as is actively happening already at Lake Abert, Lahontan Wetlands, and GSL).
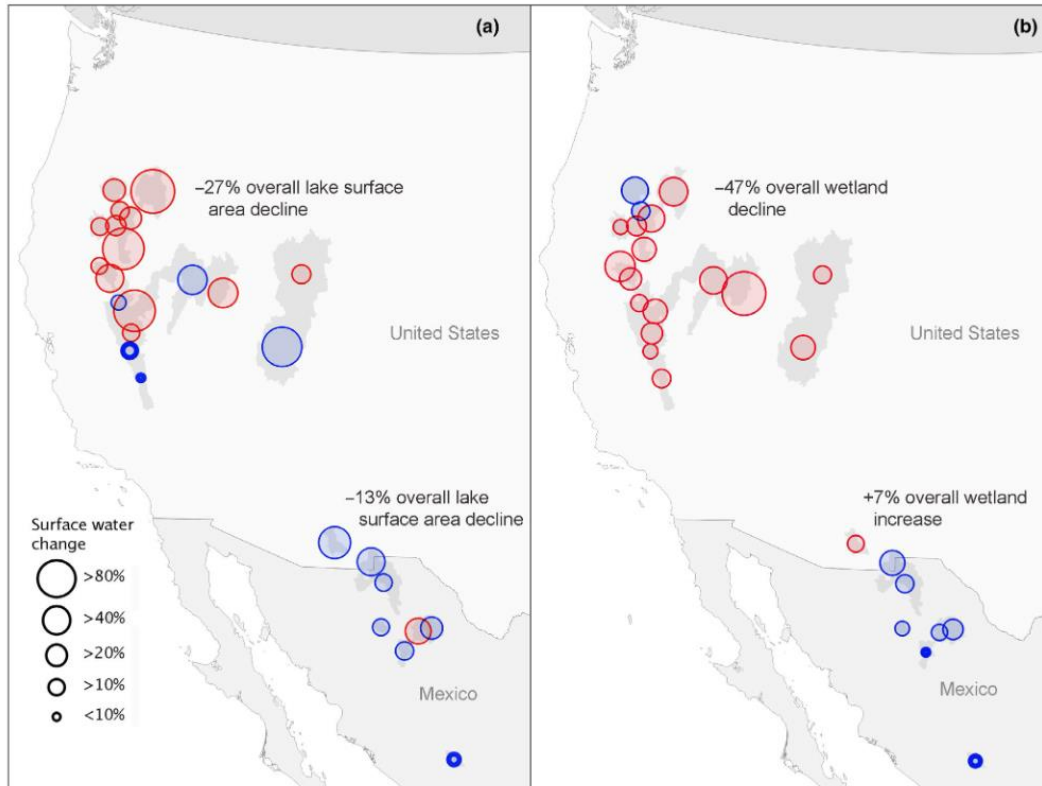
Figure 44: *Magnitude of surface water change for 26 endorheic watersheds between 1984–1999 and 2000–2018. Change is partitioned by lakes (a) and wetlands (b). Statistically significant (p < .05) declines are shown in red and insignificant declines shown in blue. Increases to surface water area are shown in bold blue outline. Figure from Donnelly et al. 2020, p. 2049.*

Climate models predict for the period 2021–2050 that precipitation, temperature, and evapotranspiration will increase in the GSL basin due to climate change (Hassan et al. 2022, p. 2714). Depending on the model, that study predicted a 1.1-1.3 or 1.4-1.7 °C increase in temperature, a 4.0-4.4 or 4.1-4.7% increase in evapotranspiration, and a –2.0 to +9.0 or –1.5 to +6.7% in precipitation (Hassan et al. 2022, p. 2714). Despite potential increases in precipitation, increased evaporation due to higher water temperatures, early snowmelt, and lower ice cover result in a predicted overall decline in GSL surface water elevation levels (Hassan et al. 2022, p. 2715). Climate modeling predicted that GSL surface water elevation resilience (i.e., the ability to recover after a low water period) will decline significantly and that recovery times will increase following ecosystem "failure" events (Hassan et al. 2022, p. 2715). In the 2001–2021 period, there was a trend toward fewer days of snow cover and strongly increasing day and night-time land surface temperatures in the GSL watershed (Hall et al. 2023, p. 8). During the same period, there was a mean increase in evapotranspiration of +28.1 mm and a statistically positive trend in evapotranspiration in many areas of the GSL basin (Hall et al. 2023, p. 12). There was a 1.4 °C increase in daytime surface water temperature at GSL with a 0.7 °C rate of change per decade (Hall et al. p. 15). There is debate about the role of climate in the current decline of GSL's water levels (Meng et al. 2019, p. 1, Wine et al. 2019, p. 1), but it has been convincingly shown that

water diversion, rather than climate, is the main driver of currently declines (Wine et al. 2019, p. 1, Wurtsbaugh et al. 2017, p. 4). However, studies agree that in the long-term, increased temperatures and evapotranspiration will negatively impact the water levels and freshwater inputs to GSL (Wurtsbaugh et al. 2017, p. 4, Wine et al. 2019, p. 5, Meng et al. 2019, p. 1).

The broad trends toward climate aridification, extreme drought, and increased temperatures and evaporation are also impacting Lake Abert (Hall et al. 2023, p. 13). Oregon's average temperature has warmed by 2.2°C since 1895 and is expected to warm by up to 5°C by 2050 and 8.2°C by 2080 (Fleishman 2023, p. 7). The frequency of droughts in Oregon, especially snow droughts, is also expected to increase (Fleishman 2023, p. 7). Continuing increased temperatures will likely result in reduced streamflows to Lake Abert, as well as greater demand for water from irrigators, in order to offset increased evapotranspiration, and greater evaporation from Lake Abert's surface (Larson 2023, p. 293).

In the Mono Lake Basin, climate models predict significant increases in annual temperature (2.5-4.1°C depending on emissions scenarios), with small decreases (1-3%) in precipitation by the end of the century (Ficklin et al. 2013, p. 111). Evapotranspiration in the Mono Basin is predicted to increase by 10 mm by the end of the century (Ficklin et al. 2013, p. 111). The combined effect of these changes is expected to reduce annual streamflow by 15% compared to historic conditions, to result in a 10-15% decreased occurrence of wet hydrologic years, and a 7-22% increased occurrence of drought conditions (Ficklin et al. 2013, p. 111). Those authors conclude that these changes, "will lead to less water available for aquatic ecosystems and water resources" and that, "Decreases in streamflow coupled with increases in Mono Lake evaporation from increased temperatures will further reduce the Mono Lake volume," (Ficklin et al. 2013, p. 138). Observed trends since the 1990s are consistent with these predictions: from 1992–2009, Mono Lake's nighttime water temperatures rose 0.15 ± 0.03°C per year for a total of 2.6°C over the study period (Schnieder et al. 2009, p. 4-5). Since 2001, the Mono Basin trended toward fewer days of snow cover and increased temperatures, with a loss of 28.4 days of snow cover per year and a 2.9°C increase in mean temperature comparing the decades of 2001–2011 and 2012–2021 (Hall et al. 2023, p. 10, 14). Comparing the time periods of 1984–1999 and 2000–2018, Mono Lake gained surface area (Donnelly et al. 2020, p. 2048-2049), due to regulatory action decreasing water diversions beginning in 1994. However, the lake rose much less than expected, standing 3.3 m below the intended management level in 2022 (Hall et al. 2023, p. 17).

Wilson's phalaropes breed in some areas that overlap with staging areas in the interior west of North America. The impacts expected in these areas are similar to that described above for staging areas—a hotter climate is expected in the 21st century, likely to result in loss of wetland habitat (Shepherd & McGinn 2003, p. 305, Ballard et al. 2014, p. 25). Various climate envelope modeling exercises by Audubon resulted in differing predictions of breeding habitat lost, from 46%–100% of breeding range habitat lost. Based on the model predicting 100% of breeding range habitat loss by 2080, Audubon considered Wilson's phalaropes "climate endangered" (Audubon 2014, *entire*). In a more recent model by Audubon, 46% of breeding habitat was predicted to be lost under a 3°C degree warming scenario, with virtually all breeding

habitat in the Great Basin and Great Plains regions of the United States lost as suitable breeding habitat (Audubon 2023, *entire*). This scenario also included predictions of relatively stable habitat in the Canadian plains and habitat gains in northern Canada, presumably as boreal forest is converted to grasslands (Audubon 2023, *entire*). Using that model, temperature increases of 1.5 and 2°C resulted in increases in breeding habitat loss of 22% and 30%, respectively (Audubon 2023, *entire*).

The wintering region of Wilson's phalaropes in the Andes has a complex precipitation regime influenced by complex local topography, with the main sources of precipitation coming from the east from the Amazon Basin (Neukom et al. 2015, p. 2). The altiplano region used by Wilson's phalaropes, located in the rain shadow of the Andes, is naturally exceptionally dry (Lupo et al. 2018, p. 78). The region has experienced an increasing trend in temperature during the 20th and 21st centuries that is projected to continue (Vuille et la. 2015, p. 3755, Urrutia & Vuille, 2009, p. 10). Precipitation is predicted to decrease by up to 33% in the region by 2100, driven by increased westerly winds (Minvielle & Garreaud 2011, p. 4582, Neukom et al. 2015, p. 8). However, predicting precipitation in the altiplano is complex (Urrutia & Vuille, 2009, p. 13), and some regions and model scenarios predict potential increases in precipitation (Blin et al. 2022, p. 16). There were observed increases in precipitation in the Bolivian altiplano from 1981–2018, which were characterized by more intense rainfall but a shorter overall rainy season (Torres-Batlló & Marti-Cardona 2020, p. 12, Torres-Batlló et al. 2020, p. 15). The same areas experienced increasing evapotranspiration over the same period (Torres-Batlló et al. 2020, p. 15). In other parts of the region, lake-area extent is currently at a 600-year low (Morales et al. 2015, p. 1147), and southwestern South America region experienced a long-term drying trend from the early 20th century to the early 21st century (Morales et al. 2012, p. 663, Rivera et al. 2020, p. 12). In Central Chile (latitude 30ºS, including the southern edge of Wilson's phalaropes' range in Chile), 2010–2019 was the longest sequence of drought years in the historical record (since 1918), and was considered a "mega-drought" (Garreaud et al. 2020, p. 435). This drying is expected to continue in the future because of anthropogenic climate forcing (Gerroud et al. 2020, p. 436). Overall, the altiplano is expected to continue to experience warmer temperatures (Minivielle & Garreaud 2011, p. 4582, Vuille et al. 2015, p. 375), increased evaporation rates (Torres-Batlló et al. 2020, p. 15), and regional decreases in precipitation (Neukom et al. 2015, p. 8). This, combined with the impacts of lithium mining, is likely to result major losses of altiplano wetlands habitats that phalaropes rely on.

V. REQUEST FOR CRITICAL HABITAT DESIGNATION

The Center formally requests that the Service designate critical habitat for Wilson's phalarope (*Phalaropus bicolor*) concurrently with listing, as required by the ESA. 16 U.S.C § 1533(a)(3)(A). Critical habitat, as defined by Section 3 of the ESA, consists of:

> *(i) the specific areas within the geographical area occupied by a species, at the time it is listed…. on which are found those physical or biological features (I) essential to the conservation of the species, and (II) which may require special management*

considerations or protections; and (ii) the specific areas outside the geographical areas occupied by the species as the time it is listed… upon a determinization by the Secretary that such areas are essential for the conservation of the species.
Id. at § 1532(5).

The effectiveness of the ESA depends on the designation of critical habitat; specifically, if a species' survival depends largely on protection of its habitat, then listing a species, on its own, would not accomplish recovery. H. Rep. No. 94-887 at 3 (1976). Accordingly, Congress has long recognized that "classifying a species as endangered or threatened is only the first step in ensuring its survival," and that protecting essential habitat is just as important to a species' survival and recovery as listing. *Id.*

Without a critical habitat designation, Wilson's phalarope has limited chances of survival and recovery. Significant portions of the species' current habitat face utter destruction due to dewatering. Wilson's phalarope populations have already declined up to 80%. The remaining populations therefore require a critical habitat designation in order to avoid further losses. The Center thus requests that the Service propose designating critical habitat for Wilson's phalarope concurrently with its proposed listing.

Critical habitat should include all existing breeding, staging, and winter habitat areas as appropriate, in addition to areas where recovery could potentially occur, as well as any areas otherwise determined to be important to the survival and recovery of the species.

VI. CONCLUSION

The Wilson's phalarope relies heavily on just a few key sites during a crucial part of its annual cycle. The staging period, in which Wilson's phalaropes congregate in large numbers at Great Salt Lake, Lake Abert, and Mono Lake, is a key energetic pinch-point in which they must rapidly double their weight and molt feathers for the subsequent long-distance migration. These large perennial saline lakes are key habitats because they provide extraordinary abundances of the specific prey on which phalaropes specialize (alkali flies, brine flies, and brine shrimp). There are clear examples in which other shorebirds that migrate long-distance have declined sharply toward extinction when similar key staging habitats have been lost (e.g., multiple species in the Yellow River Delta and the red knot in eastern North America).

Although Wilson's phalaropes have a large range and relatively large population, the dependence of large portions of the world population on just a handful of major staging sites means they could rapidly decline if those sites disappear. Unfortunately, under the status quo of water use and policy, Great Salt Lake and Lake Abert are on the foreseeable trajectory toward ecosystem collapse and disappearance as functional habitat for phalaropes in the near future. Great Salt Lake nearly experienced the beginnings of ecological collapse in 2022, saved only by an extraordinarily wet winter. Lake Abert has already entered a cycle in which the lakebed will likely be dry more often than it is wet. The third major staging site, Mono Lake, is better protected but remains vulnerable with ongoing water diversions and impacts of climate change. The situation at Great Salt Lake and Lake Abert currently threatens the destruction of habitat

throughout a significant portion of the range of the Wilson's phalarope.

In addition to these threats at North American saline lakes, the rapid expansion of lithium mining in the Andean Altiplano region poses an equally serious threat to Wilson's phalarope habitat. Lithium mining requires immense amounts of groundwater pumping, in a region where the water balance is naturally negative and most surface water is "fossil" water >100 years old from aquifers. There is a real threat that lithium mining may dry up surface water across much of this region, where >25% of all Wilson's phalaropes spend the winter.

An exacerbating layer on the habitat situations in both North and South America is the aridification of the climate due to anthropogenic climate change. Resiliency and redundancy of wetland networks in both continents have and are predicted to continue to be lost. Smaller and more ephemeral wetlands are the first to disappear because of rising temperatures and evaporation rates, and more frequent and severe droughts. This increases the importance of the large perennial saline lakes—habitats whose resilience to climate change has been severely degraded by water diversions. "Backup habitats" cannot be expected to save the Wilson's phalarope from habitat loss at saline lakes, because the intermountain west and the altiplano are experiencing general aridification and wetland loss.

Thus, the Wilson's phalarope faces multiple serious, near-term threats that threaten its habitat and survival. Existing regulatory mechanisms are inadequate for addressing these threats—Wilson's phalaropes do not have legally binding levels of protection in any country or state in their range, except the state of Minnesota where only a small population nests. Many important Wilson's phalarope habitats hold symbolic statuses of protection, such as WHSRN and RAMSAR site declarations—but these statuses do not confer legally binding water rights. Even State Park and National Park designations do not help solve the fundamental threat of water diversion for places like Great Salt Lake and Laguna Mar Chiquita. Of all the habitats we describe in this petition, only Mono Lake has legal protection of its water levels —and even so, it remains threatened by regulated water diversion.

Despite growing media attention and political awareness of these issues, especially at Great Salt Lake, the legal actions to protect these habitats have fallen far short. Legislative action on behalf of Great Salt Lake has so far yielded only 0.03% (64,000 acre-feet) of the 2.5 million acre-feet estimated to be needed annually to restore Great Salt Lake and prevent its further collapse. The program that produced that 0.03% currently has $40 million in funding, far short of the financing needed to obtain sufficient water to restore the lake. The Utah legislature has failed to pass a bill to set a target lake level for Great Salt Lake. Meanwhile, a water development proposal for the Bear River tributary to Great Salt Lake would further reduce water for the lake. There has been no legislative effort on behalf of Lake Abert to date, but there are new proposals to further develop the Chewaucan River watershed with new dams that would reduce available water for the lake. In South America, lithium mining development is rapidly proceeding with little governmental oversight or regulatory compliance.

In addition to the threatened destruction of habitat across a significant portion of the Wilson's phalarope range, and lack of regulatory mechanisms to address it, multiple lines of scientific evidence indicate that the species' world population has steeply declined—up to 70%– since the 1980s. Its relatively large population (currently estimated at 1 million individuals) does not make it immune to extinction—shorebirds are declining faster than other birds, and

migratory shorebirds are declining faster than other shorebirds. Shorebirds that rely on very specific habitats with specialized prey are more vulnerable still. The Wilson's phalarope meets the definition of Threatened under the ESA—it is at risk of becoming endangered in all or a significant part of its breeding range, in the foreseeable future.

The Wilson's phalarope is a remarkable bird in many ways—it is a shorebird that swims; it flaunts traditional male-female roles in breeding; it uses hyper-saline lake habitats too harsh for many other species; it is the only species that links saline lakes as a network throughout North and South America. Losing the Wilson's phalarope would mean losing an iconic species of saline lakes, losing a crucial ecological link in these ecosystems, losing an awe-inspiring migration over the length of our American continents, losing part of the soul that animates saline lakes throughout the hemisphere.

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