Conservation Plan for Wilson's Phalarope

(Phalaropus tricolor)

Version 2.0
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Executive Summary

Wilson’s Phalarope (*Phalaropus tricolor*) is a medium-size shorebird best known for congregating in huge flocks at a few key sites across the western hemisphere, for its highly aquatic habits, and for its reversed sexual dimorphism, with females being larger and more colorful than males. During the migration and non-breeding seasons, Wilson’s Phalarope prefer larger bodies of saline and alkaline water, which support a more abundant invertebrate fauna than small bodies of freshwater. When swimming, phalaropes often are seen whirling about in circles on the water, spearing prey at or near the surface in a very distinctive behavior.

Unlike Red Phalarope (*P. fulicarius*) and Red-necked Phalarope (*P. lobatus*), which have Holarctic breeding distributions and spend the non-breeding season at sea, Wilson’s Phalarope is a temperate breeder and spends the non-breeding season inland. The species is confined to the Americas where it breeds in central Canada and central and western United States and winters on saline lakes in the Andes of South America and the Southern Cone lowlands.

Despite being relatively well-studied in its breeding areas and at pre-migratory staging sites in North America, less is known about the ecology of the species during their migration and non-breeding season in South America. Monitoring population trends of Wilson’s Phalaropes is challenging, due to the remote and difficult-to-access habitats that they prefer, dynamic of use of these habitats, aquatic habits, and the fact that they are a highly gregarious species (flocks can number up to several hundred thousand individuals at a single site during migration and non-breeding seasons). Until very recently, the population size and trends of the Wilson’s Phalarope remained relatively uncertain. New efforts and analyses have indicated both a lower world population than previously thought and serious declining trends in North America.

Although the species has a relatively large population, most recently estimated at 1,000,000 individuals, Wilson’s Phalaropes warrant urgent conservation planning based on serious current threats to their habitat and declining population. The species’ population is believed to have undergone a significant decline during the early 20th century due to loss of breeding habitat, and a recent analysis of ISS data (Smith *et al*. 2023) indicated the species may have declined by approximately 75% in its North American range since 1980. Currently, many saline lake habitats used by Wilson’s Phalarope are existentially threatened by a combination of water diversion, drought,

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and other climate-related changes (e.g., reduced snowpack, altered precipitation patterns, increased evapotranspiration). The following are serious contemporary threats to habitat requiring urgent conservation measures:

1. Rapid desiccation of North American staging habitat, particularly at saline lakes, related to water diversion and climate change. Great Salt Lake, Utah, hosts the largest concentrations of Wilson's Phalaropes in North America (up to 60% of the world population at once). In late 2022, Great Salt Lake's surface elevation had dropped to 4188', its lowest level in recorded history (1847-present; USGS 2023). At that level, the lake had lost >50% of its volume compared to the volume at an average lake elevation over the historical time-series. It also had reached salinity levels exceeding 160 g/L, exceeding the threshold of reproduction and survival of brine shrimp (USGS 2023). Lake Abert, Oregon, another major staging site, became almost completely dry in 2014-2015 and again in 2021-2022. During these periods, there were major declines in phalarope numbers at that site. Mono Lake has a guaranteed legal water right, but in 2022 its surface elevation was 13' below the mandated target level. Water levels at each of these lakes improved due to an exceptionally large snowpack in the boreal winter of 2022-2023, but these gains are likely to be short-term because the drivers of water level declines—water diversion and an overall drying climate—have not been mitigated. Smaller and more ephemeral wetlands used by phalaropes in the Great Basin are likewise threatened by desiccation from upstream water diversion and drought related to climate change.

2. Habitat loss and water availability in the High Andean and Southern Cone lowlands saline lakes. The majority of the population spends the non-breeding season in the saline lakes of the altiplano in Argentina, Bolivia, Peru, and Chile, where water use mainly for mining activities is affecting the quality and extent of these lakes. Large concentrations also spend the austral summer in saline lakes on the Pampas and lowlands of Argentina, where ever-expanding agriculture and water use are significant threats. Major sites such as Laguna Mar Chiquita, Argentina (which can host >50% of the world population), do not have guaranteed water rights and are susceptible to the same threats of water diversion and drought as North American lakes.

3. Continuing loss of breeding habitat in North American prairie grasslands and wetlands. During the 20th century, conversion of most of the North American prairie grasslands and associated wetlands to agriculture destroyed a significant area of appropriate breeding habitat for this species. This is believed to have been a major factor in its apparent decline during the 20th century and will contribute to further declines if no conservation measures are taken and sustained. A decision in 2023 by the United States Supreme Court to end Clean Water Act protection of a seasonal wetlands could be detrimental to phalarope breeding habitat in the U.S. The species is also vulnerable to the effects of climate change on the breeding range, as parts of the prairies are predicted to become drier, and drought has already been documented among the main reasons for breeding areas being abandoned.

Overall, there is a clear need for urgent conservation action to protect the habitat of the Wilson's Phalarope, as well as a need for further research to better understand the species' population status, trends, and responses to rapid habitat changes. We conclude this Conservation Plan with outlines of priority conservation and research needs for the species. As a long-distance migrant, the Wilson's Phalarope exemplifies the need for conservation efforts to be coordinated regionally and internationally. As a saline lake specialist, the unique and charismatic Wilson's Phalarope can also bring attention to the urgent conservation needs of interior saline wetlands throughout the western hemisphere.

Purpose

This conservation plan defines the conservation status of Wilson's Phalarope throughout its range, describes current threats, identifies research and management needs, and outlines recommended conservation actions on the basis of current knowledge of the species. There have been significant advancements in our understanding of the conservation status, threats, and priority conservation actions for the species since the first version of this plan (Lesterhuis & Clay 2010). The goal of this version (2.0) is to update the Plan with current knowledge and reevaluate conservation priorities, to help guide management and research, identify knowledge gaps, and develop conservation strategies to benefit this species in both the short- and long-term and prevent future declines.

Natural History


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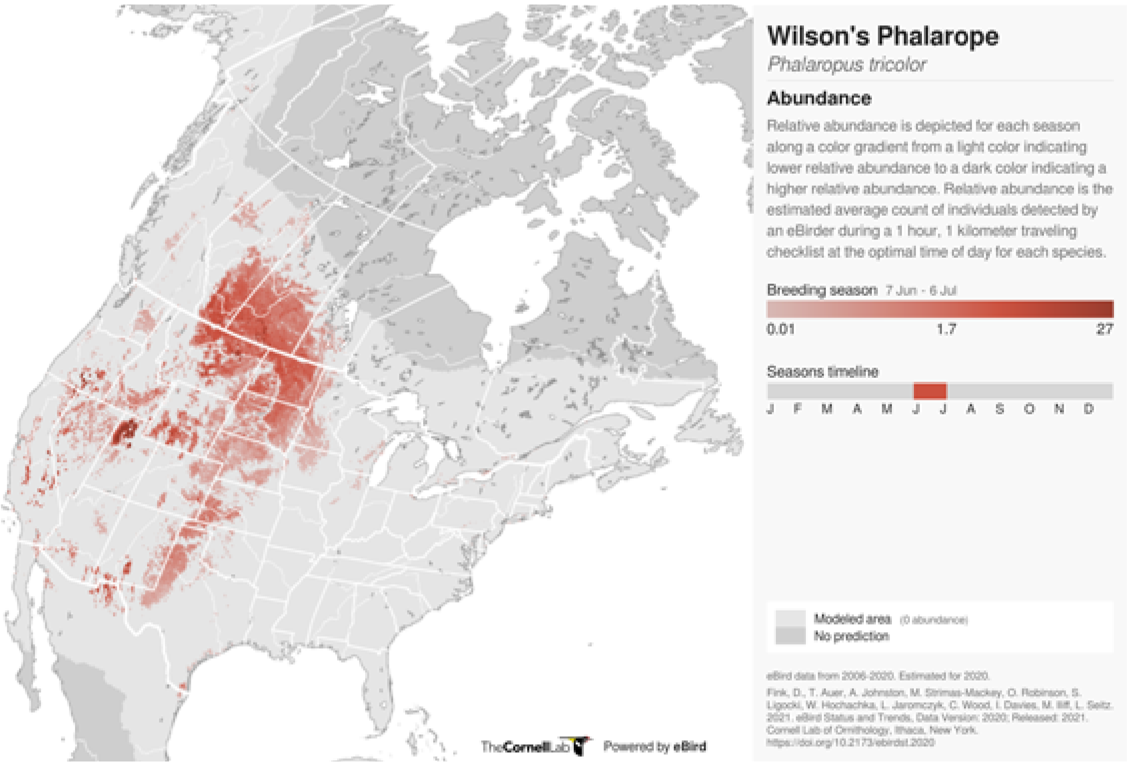
Wilson's Phalarope (*Phalaropus tricolor*) has been relatively well-studied on its breeding and pre-migratory staging sites in North America. Research has focused on particular aspects of the species' breeding biology such as courtship, nest site selection, nest densities, and nest success (e.g., Höhn 1967; Howe 1975; Murray 1983; Colwell 1986, 1987, 1992; Colwell & Oring 1988a, b, c, d, e; Delehanty & Oring 1993). Diet and foraging strategies and habitat use have also been the focus of several studies (Siegfried & Batt 1972, Skagen & Oman 1996, Laubhan & Gammonley 2000, May *et al*. 2002, Andrei *et al*. 2009, Frank & Conover 2019, 2021a, 2021b). Other studies have attempted to unravel the origin and phylogenetic relationship among phalaropes (e.g., Dittman & Zink 1991, Ericson *et al*. 2003). Given that Wilson's Phalaropes are commonly found in saline habitats throughout their annual lifecycle, a number of authors have focused their studies on the species' adaptations to those habitats (e.g., Mahony & Jehl 1985, Jehl 1988). During the non-breeding season, the species has been extensively studied at staging sites in North America (Jehl 1981, 1987, 1988, 1997, 1999). On the other hand, very little information is available regarding the species on its South America non-breeding grounds; among the exceptions are studies on feeding and behavior (Burger & Howe 1975, Gutiérrez & Soriano-Redondo, 2020) and on interactions with Chilean Flamingos (Phoenicopterus chilensis) (Hurlbert *et al*. 1984, Gutiérrez & Soriano-Redondo, 2018). Unlike most shorebirds, Wilson's Phalaropes are highly aquatic, foraging principally while swimming. They mainly feed on small invertebrates like dipterans and crustaceans, particularly brine flies, chironomids (Chironomidae) and Daphnia (Daphnia magna) (Frank & Conover 2021a), but also occasionally on seeds of aquatic plants (Colwell & Jehl 2020, O'Brien *et al*. 2006). During foraging, Wilson's Phalaropes sometimes spin in circles, which creates a vortex that draws invertebrates to the surface (Jehl 1988, Obst *et al*. 1996, Frank and Conover 2021b). They often forage among flocks of other species such as American Avocet (Recurvirostra americana), Northern

Shoveler (Anas clypeata), Blue-winged Teal (Anas discors), and Chilean Flamingos (Phoenicopterus chilensis), allowing them to feed on invertebrates that those species stir up (Williams 1953, Siegfried & Batt 1972, Hurlbert *et al*. 1984, O'Brien *et al*. 2006, Gutierrez & Soriano-Redondo, 2018). Wilson's Phalaropes can also be observed foraging on land, rapidly chasing and pecking prey from the ground, often disturbing other shorebirds (O'Brien *et al*. 2006). Courtship, which begins during northbound migration, is mostly characterized by female-female aggression (Höhn 1967, Kagarise 1979, Colwell & Oring 1988). After pairs are formed, females initiate nest site selection, which may be in vegetation on the edge of a lagoon or in upland areas, but always within 100 meters of wetlands (Höhn 1967, Colwell & Jehl 1994). Nests are usually situated in mixed vegetation that is generally taller and denser than that used by other prairie-breeding shorebirds (Colwell & Oring 1990). After clutch completion (usually four eggs), females leave the task of incubating and taking care of young entirely to the males (Colwell & Jehl 2020). As in other phalaropes, polyandry has been documented for Wilson's Phalarope, whereby a female will lay clutches with multiple males (Colwell 1986).

Distribution and Habitat

Breeding

The breeding range of the Wilson's Phalarope covers much of interior western North America (Fig. 1). In Canada, it breeds in southern Yukon, British Columbia, Alberta, Saskatchewan, Manitoba, and locally in southern Ontario and Quebec (Gauthier & Aubry 1996). In the United States, core breeding-range states include Montana, Wyoming, Idaho, North Dakota, South Dakota, Utah, northeastern California, and Oregon (Fig. 1). There are localized nesting records from Arizona, New Mexico, Nevada, Colorado, central to southern California, Nebraska, Texas, Minnesota, Illinois, Indiana, Ohio, Massachusetts, Kansas and Alaska (Johnsgard 1981, DeGraaf & Rappole 1995, Sutton & Arcilla 2018, Stanley



Relative abundance is depicted for each season along a color gradient from a light color indicating lower relative abundance to a dark color indicating a higher relative abundance. Relative abundance is the estimated average count of individuals detected by an eBirder during a 1 hour, 1 kilometer traveling checklist at the optimal time of day for each species.

eBird data from 2006-2020. Estimated for 2020.

Fink, D., T. Auer, A. Johnston, M. Strimas-Mackey, O. Robinson, S. Ligocki, W. Hochachka, L. Jaromczyk, C. Wood, I. Davies, M. Iliff, L. Seitz. 2021. eBird Status and Trends, Data Version: 2020; Released: 2021. Cornell Lab of Ornithology, Ithaca, New York. https://doi.org/10.2173/ebirdst.2020

Figure 1[1]: Range and relative abundance of Wilson's Phalarope during its breeding season, based on modeled eBird records from 7 June-6 July, 2006-2020. (Fink *et al.* 2020).

Senner *in litt*.). A contraction of the breeding range (and presumably population numbers) during the early 20th century was likely driven by the loss of prairie wetlands (Colwell & Jehl 2020). In the 1990s, there was indication of potential range expansion or recovery based on greater nesting effort at the edges of the breeding range (e.g., Alaska, British Columbia, and Quebec; Colwell & Jehl 2020). However, it is also possible that that pattern was driven by shifting reproductive effort away from the core breeding range in response to drought during the 1990s (Colwell & Jehl 2020). The most recent Breeding Bird Survey results (1966-2019) indicate negative trends in all North American regions except the Prairie Pothole region in the core breeding range, suggesting declines, rather than expansion, in breeding effort on the edges of the range (Sauer *et al*. 2019).

With the replacement of much of the native prairies of Canada and the United States by agriculture, the current primary breeding habitat of Wilson's Phalarope is shallow water bodies in disturbed mixed-grass prairies and agricultural areas (DeGraaf and Rappole 1995). The species nests semi-colonially near shallow ponds and lakes, ranging from fresh to highly saline, and preferably close to wet-meadow vegetation. However, it is also found nesting in swales along streams, shallow sloughs fringed with short grasses, and hay meadows or pastures, up to 100 m from water (Johnsgard 1981, Colwell & Jehl 2020, DeGraaf & Rappole 1995). Additional breeding habitats include taiga interspersed with moist, grassy muskeg, and aspen-grove parklands (DeGraaf & Rappole 1995). Of 438 nests found by Johnsgard (1981) in North Dakota, 50% were on semi-permanent ponds ranging from fresh to sub-saline; 40% on seasonal ponds and lakes; and the remaining 10% on alkali ponds or lakes and other types of wetlands, such us fen ponds. 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.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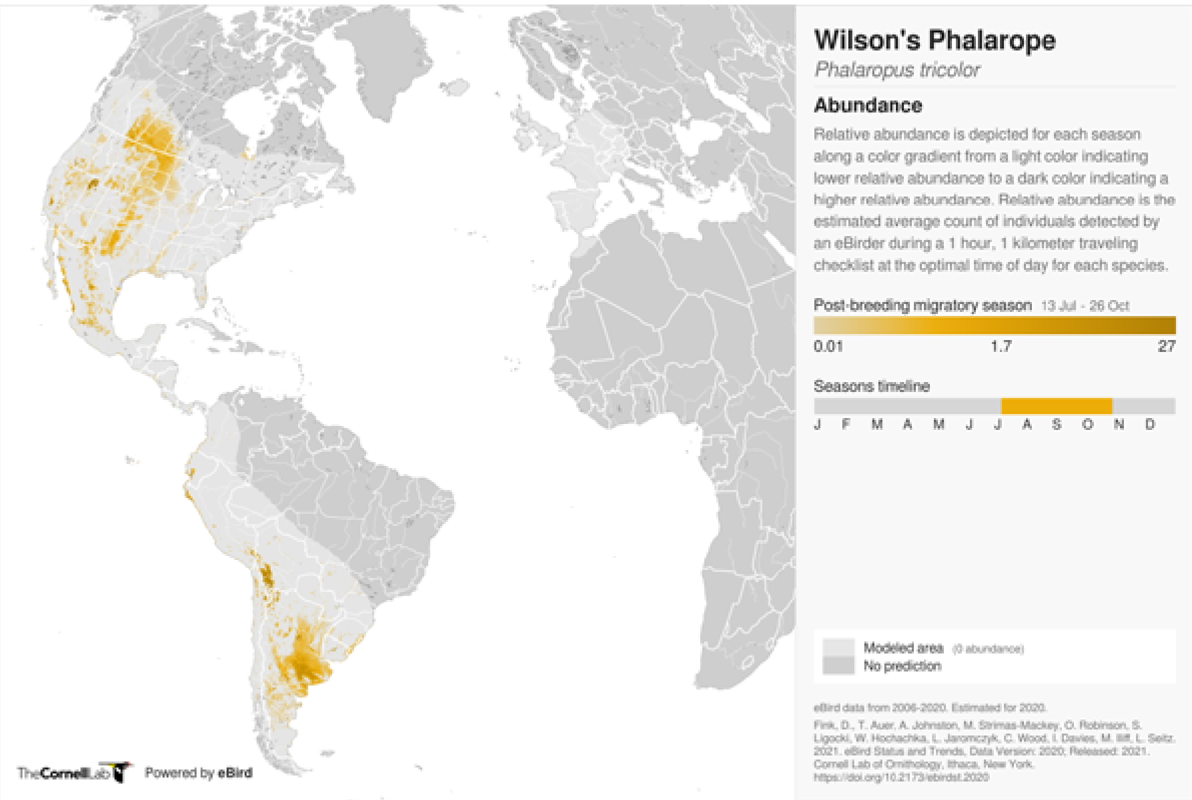
Migration

Southbound Migration

Wilson's Phalarope is a long-distance migrant, with birds undertaking annual movements from breeding grounds in central North America to non-breeding ("wintering") grounds in central and southern South America. Of the three major flyways identified for the western hemisphere (https://shorebirdflyways.org/) the species is most concentrated in the Mid-continent Flyway, crossing the interior of North America (Fig. 2). Brown *et al*. (2001) identified five distinctive flyways in North America (Pacific-Asiatic, Intermountain West, Central, Mississippi, and Atlantic), of which this species mainly uses the Intermountain West (up to 90% of adults) and Central Flyway. During migration, Wilson's Phalarope uses a variety of wetland habitats, from coastal wetlands and lagoons to freshwater wetlands and hypersaline lakes.

The southbound migration starts with females departing the breeding grounds and arriving at staging areas in western North America by mid-June (Jehl 1988; Fig. 2). Males generally arrive two or more weeks



Relative abundance is depicted for each season along a color gradient from a light color indicating lower relative abundance to a dark color indicating a higher relative abundance. Relative abundance is the estimated average count of individuals detected by an eBirder during a 1 hour, 1 kilometer traveling checklist at the optimal time of day for each species.

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Figure 2[1]: Range and relative abundance of Wilson's Phalarope during its post-breeding season corresponding to its southbound migration, based on eBird records from 13 July-26 October, 2006-2020 (Fink et al. 2020).

later (Jehl 1988) due to the species' reversed sex roles where males remain at the breeding site longer than females to care for the eggs and young. Notably, Wilson's Phalarope is one of only two shorebird species known to undergo a molt migration (Jehl 1988). Large numbers of Wilson's Phalaropes stage at hypersaline lakes in western North America where abundant food enables a rapid molt and fattening prior to a presumed non-stop flight to South America (Jehl 1988). At staging sites, adults amass loads of up to 54% of total body mass; for most shorebirds, fat loads of 45% of body mass is the maximum (Jehl 1997). This gain results in a brief period during which some phalaropes would be too heavy to fly, a phenomenon that is unknown for other species of shorebirds (Jehl 1997).

During June-August, huge flocks of Wilson's Phalaropes gather at saline lakes in western North America, including Great Salt Lake (Utah), Lake Abert (Oregon), Mono Lake (California), Lahontan Valley (Nevada), and Old Wives' and Chaplin Lakes (Saskatchewan; Jehl 1988). Typically, hundreds of thousands of birds gather annually at Great Salt Lake (Paul & Manning 2002), whereas other important staging sites (e.g., Mono Lake, Lake Abert) often have flocks numbering in the tens of thousands (Jehl 1988). Major staging sites tend to have an abundant supply of invertebrate prey. At these sites, alkali flies (*Ephydra hians*) and brine flies (*Ephydra gracilis*) in their various life stages are a major diet item of Wilson's Phalaropes. Brine shrimp (*Artemia spp.*), though abundant at many staging sites, were not recorded in phalarope diets at Great Salt Lake, Utah (Frank and Conover 2021a), but were at Mono Lake (Jehl 1988, Colwell and Jehl 1994). Other prey taken at Great Salt Lake included chironomids and Daphnia, which were associated with lower-salinity areas of the lake (Frank & Conover 2021a). During southbound migration, smaller numbers of phalaropes also visit wetlands across a broad geographic front in western North America in habitats including flooded meadows, alkaline ponds, coastal estuarine marshes, and sewage ponds.

Departure from the staging areas is from late July to August, and nearly all migrants have left by mid-September (Jehl 1988). Birds are presumed to then undertake a rapid and direct non-stop flight from the staging areas to coastal South America, crossing the Pacific Ocean (Jehl 1988). The theory that the southbound migration is direct and non-stop is based on the low number of fall records for Central America and northern South America, and the short interval between timing of departure dates from North American staging areas and first arrival dates in South America (Jehl 1988). However, in recent years eBird has greatly expanded observer coverage in Mexico and Central America, showing an abundance of Wilson's phalaropes records in Mexico and Central America during July–November (https://ebird.org/map/wilpha). Most eBird records in that region during those months are of single birds to groups of low hundreds at coastal wetlands and interior lakes. Preliminary data from birds tagged at Tule Lake, California in 2023 also indicated that at least some birds make more frequent stops on their way south (USGS, Margaret Rubega, & Oikonos unpublished data; https://motus.org/dashboard/#e=profile&d=species&s=4990). Tagged birds made intermediate stops at wetlands in southern California, coastal lagoons and estuaries in Baja California, Mexico, and lakes in interior central Mexico (USGS, Margaret Rubega, & Oikonos unpublished data). These tagging results were from a small sample of birds (n = 15) tagged at just one site, so more research is needed to more fully understand the scope of phalarope migratory behavior.

The first migrants arrive on the west coast of Ecuador and Peru by early August (Fig. 2). The artificial salt lakes of Ecuasal, loca-

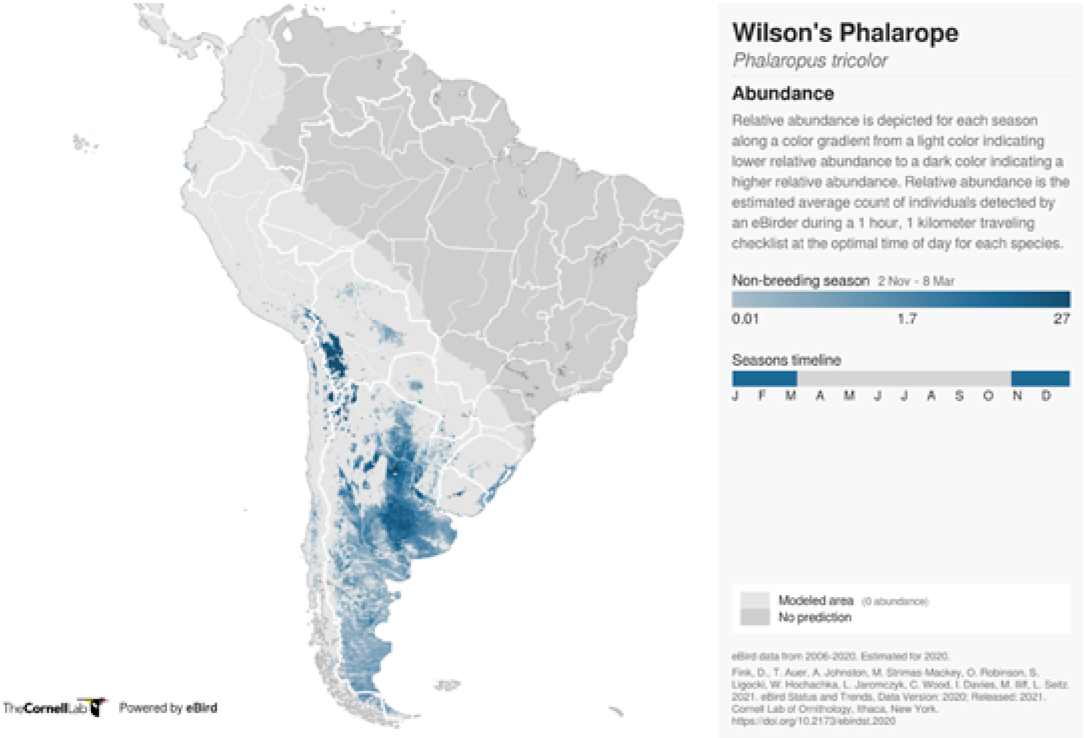


Figure 3[1]**:** Range and relative abundance of Wilson's phalarope during its non-breeding season, based on eBird records from 2 November - 8 March, 2006-2020 (Fink et al. 2020). Some records from Central America are not shown on this figure.

ted along the southern coast of Ecuador, are an important stopover site (Agreda *et al*. 2009). Numbers of Wilson's Phalaropes at Ecuasal tend to peak between August and September, with numbers in the range of 15,000-44,300 individuals (Agreda *et al*. 2009, A. Agreda, *in litt*.), though only a few hundred are seen in some years (Ridgely and Greenfield 2001).

From the west coast of South America, birds move south and east to sites in the high Andes (including parts of Peru, Chile, Bolivia, and Argentina; Fig. 2). Many individuals ultimately continue to sites in the Argentinean lowlands and Patagonia (Fig. 2). During migration, flocks of up to 25,000 birds are sometimes observed in the Paraguayan Chaco (Lesterhuis & Clay 2001). Some individuals reach as far south as Santa Cruz and Tierra del Fuego, Argentina (Hurlbert *et al*. 1984, Roesler & Inverti 2015).

Non-breeding (wintering)

During the non-breeding season, Wilson's Phalarope can be found over a wide area extending from northern Peru diagonally to Uruguay and south to Tierra del Fuego (Colwell & Jehl 2020; Fig. 3). Within this area, two distinct regions are of primary importance: the Andes (including northern Argentina, Chile, Bolivia, and southern Peru), and the central lowlands of Argentina (Hurlbert 1984, Colwell & Jehl 2020; Fig. 3). The highest abundances in the non-breeding range have been recorded in Bolivia and Argentina.

Occasionally, birds can be found during the non-breeding season as far north as northern Mexico and the southern U.S. (Sauer *et al*. 2008, Howell & Webb 1995). Rarely, small numbers of non-breeding birds have been observed in Hispaniola and Barbados (Raffaele *et al*. 2003), Colombia (Hilty & Brown 1986), Venezuela, the Guianas (Hilty 2003), and Brazil (Belton 1994, Mauricio & Diaz 1996, Silva e Silva & Olmos 2007, Scherer-Neto *et al*. 2008, Sick 1993). Also, there are records of small numbers of Wilson's Phalaropes (around 1,000 individuals) that stay in the pools of Ecuasal (Ecuador) during the southern summer and do not complete the migration to southern South America (Haase 2011).

The main non-breeding habitats used by Wilson's Phalarope in the high Andean region are mudflats and shallow, open-water habitats of ephemeral or permanent saline lakes. Further south in the Southern Cone lowlands (i.e., Chaco and Pampas), the species occurs mainly in lowland saline lagoons. In that region, smaller numbers of birds also occur along rivers and bays, freshwater lagoons, pools, cattle ranching reservoirs, marshes, and rice fields (Blanco *et al*. 2006).

Within specific countries in the non-breeding area, two sites were considered important for the species in Peru: The Reserva Nacional Salinas y Aguada Blanca (RNSAB), with abundances reaching up to 20,000 birds (Ramsar 2003), and Lago Junín, with "tens of thousands" (Harris 1981). However, these records are more than 30 years old, and in recent years Wilson's Phalarope was present in very low numbers or absent at these sites. No phalaropes were present during a 2020 survey of Lago Junín, and the site was not surveyed in 2021. In RNSAB, only 456 birds and 537 birds were recorded during surveys in February 2020 and 2021, respectively.
Large congregations of Wilson's Phalaropes have been reported 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). A total of 500,000 – 1,000,000 birds were estimated to be present in the southern Bolivian altiplano during the 1970s, and this area was considered the core wintering region for the species (Hurlbert *et al*. 1984). However, in recent years numbers in these lakes have

been much lower (Omar Rocha, *in litt*; see Trends section below).

Wilson's Phalarope is a regular visitor to the altiplano of Chile (Jaramillo *et al*. 2003), though the numbers are usually lower than in neighboring areas of Argentina and Bolivia. The highest counts in Chile are from Salar de Surire in the 15th Region, where there were observations of 13,590 birds in February 2020 (Castellino & Lesterhuis 2020), and 50,000 in the 1990s (Bech & Brendstrup-Hansen 1992). Wilson's Phalarope also occurs in the Patagonian region of Chile, often in scattered small groups but sometimes in flocks of >1,000 birds. For example, in the 12th Region (Magallanes), up to 5,525 Wilson's Phalaropes have occurred at Laguna de los Palos (Matus 2018) and there are records in the range of 1,000 birds at that site (Matus 2005), Laguna Toro (Saiter Villagrán 2020), and Laguna Blanco (Imberti 2020).

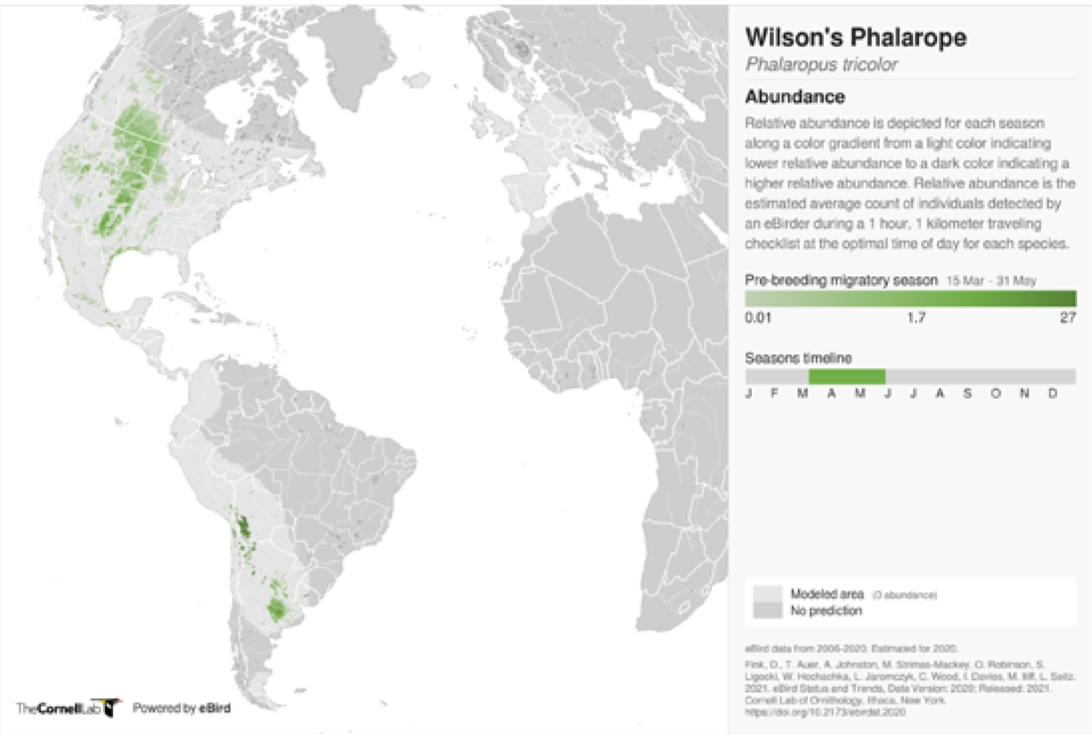
Some of the largest congregations of Wilson's Phalaropes are found in Argentina. In the northwest of the country, in the altiplano region, the most important site identified for the species is the Monumento Natural Laguna de los Pozuelos, in Jujuy province. The largest flock recorded at this site was 500,000 birds in January 1988 (Moschione & San Cristóbal, unpublished data), and 449,108 birds in February 2021 (WHSRN unpublished data). Laguna del Palar, a small and shallow lagoon that is part of the Ramsar site Lagunas de Vilama, also had a record of 143,649 birds in February 2020 (Castellino & Lesterhuis 2020) but only 418 birds were present in February 2021. Further south on the Argentinean Pampas is Laguna Mar Chiquita (Córdoba Province), a large saline lake which could be the most important non-breeding site for Wilson's Phalarope due to the high abundances recorded. In this lake, in almost every austral summer, ≥ 400,000 birds are recorded simultaneously (Pablo Michelutti, *in litt*). Close to Laguna Mar Chiquita is the Salinas Grandes, a very shallow ephemeral lake, which has records of Wilson's Phalaropes ranging from 7,300 in 2010 (Blanco 2010) up to approximately 20,000 birds in 2020 (M. Castellino, pers. obs.). Another important site in Argentina is Lago Epecuén, a hypersaline lake located in western Buenos Aires province.

The highest Wilson's Phalarope count at Lago Epecuén was 140,000 in September 2016 (M. Castellino, pers. obs.), and flocks of tens of thousands have occurred annually at this site since surveys began in 2015 (M. Castellino, pers. obs.). There are no data available for this site prior to that year. Wilson's Phalarope can also occur throughout the Patagonia region of Argentina. As in all their non-breeding range, Wilson's Phalaropes are highly mobile and abundances in Patagonia vary among sites and years. Although abundances are lower than in the Pampas, unpublished records and eBird data indicate counts of thousands of birds. Some notable counts are 5,000 at Bajo Giuliani (Minuet 2020) and 4,500 at Laguna Guatraché (ISS 1990), both in La Pampa province, and up to 20,000 at Reserva Provincial Caleta Olivia (Avalos 2020) in Santa Cruz province.

Northbound Migration

The return migration of Wilson's Phalaropes starts in March, and the first birds arrive back on the breeding grounds in late April to early May. Little is known of the return migration route or stopover locations, 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. 4). Large congregations of northbound birds have been recorded from a number of wetlands in Mexico, with flocks of up to 42,274 birds observed at Lago Texcoco (WHSRN 2007) and 60,000 at Bahía Santa María (Vega *et al*. 2006).

During northbound migration, Wilson's Phalaropes can be found in inland marshes, flooded fields, salt works, and sewage ponds, and coastal lagoons and estuaries



Relative abundance is depicted for each season along a color gradient from a light color indicating lower relative abundance to a dark color indicating a higher relative abundance. Relative abundance is the estimated average count of individuals detected by an eBirder during a 1 hour, 1 kilometer traveling checklist at the optimal time of day for each species.

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Figure 4[1]: Range and relative abundance of Wilson's Phalarope during its pre-breeding migration season, based on eBird records from 15 March - 31 May (Fink et al. 2020).

(Colwell and Jehl 1994, Hayman *et al*. 1986, Van Gils and Wiersma 1996).

After arrival in North America, most migrants appear to pass through the Intermountain West and the south-central Great Plains (Fig. 4). Spring migrants use shallow wetlands and coastal marshes in the south-central United States (Colwell & Jehl 2020). First-year birds return to the breeding grounds, but it is unclear what proportion actually breeds (O'Brien *et al*. 2006).

Over-summering birds

Since 2010, there have been records of thousands of birds spending the breeding season (austral winter/boreal summer) at several Argentinian sites, in what is normally considered the "non-breeding" range (Table 1).

Date	Site Name	State	High Count	Source
August 2010	Laguna Mar Chiquita	Córdoba	182,260	P. Michelutti, *in litt.*
August 2013	Laguna Mar Chiquita	Córdoba	18,000	
August 2015	Laguna Mar Chiquita	Córdoba	6,200	
June-July 2016	Lago Epecuén	Buenos Aires	8,000	Castellino & Bucher 2017
June 2017	Salinas Chicas	Buenos Aires	15,000	Castellino & Bucher 2017
July 2018	Salina Santa Inés	La Pampa	4,500	M. Castellino *pers. obs*
August 2018	Laguna Mar Chiquita	Córdoba	40	P. Michelutti *in litt*

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[1] The Figures 1,2,3 and 4 were obtained from https://ebird.org/science/status-and-trends. This material uses data from the eBird Status and Trends Project at the Cornell Lab of Ornithology, eBird.org. Any opinions, findings, and conclusions or recommendations expressed in this material are those of the author(s) and do not necessarily reflect the views of the Cornell Lab of Ornithology.

Population Estimate and Trend

Population Estimates

Estimating the population size and trends of the Wilson's Phalarope is complicated by multiple factors, including that they are small, often frequent remote areas, mix with other phalarope species at staging sites, and may form immense flocks. Observer error may be substantial when estimating the size of flocks of hundreds of thousands of birds. Also important, unlike most shorebirds, phalaropes spend much of their time swimming, often making them inaccessible to shore-based observers. Even when Wilson's Phalaropes are most concentrated at migratory staging sites, they may be far from shore on large bodies of water, making counting them challenging. The mixing of Wilson's and Red-necked Phalaropes at these staging sites also presents a challenge for species identification of distant birds. Finally, Wilson's Phalarope tends to be distributed over a large geographic area in all its life stages, making regional or even continentally coordinated efforts necessary for evaluating the global population. Because of these and other challenges, the population size and trends of the Wilson's Phalarope remain relatively uncertain, though new efforts have begun with the goal of better quantifying these metrics.

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), 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 in fall, 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). Other assumptions of this estimate were that there was no movement of birds between staging areas during July 1986 (Jehl 1988), and that all major sites were covered.

Subsequent reports of Wilson's Phalarope global population estimates (e.g, Morrison


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et al. 2006, Andres *et al*. 2012, Lesterhuis & Clay 2010), have continued to use Jehl (1988)'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; Skagen *et al*. 1999, ISS and MSS data cited in Morrison *et al*. 2001). It is noteworthy that the difference between Jehl (1988)'s 1.5 million estimate and the 900,000 estimate from other sources is relatively large; the difference of 600,000 birds is more than one third of the 1.5 million estimate by Jehl (1988). Accordingly, most authors have listed the 1.5 million estimate as of low accuracy. The Canadian population was estimated at 680,000 birds (Morrison *et al*. 2001), based on the estimated percentage of the species' breeding range that lies in Canada (45.3%) and Jehl (1988)'s global population estimate of 1.5 million. Several other global population estimates have been published: Hurlbert *et al*. (1984) cited 1,000,000–3,000,000 birds (an estimate provided by J. Jehl as a personal communication, with no supporting data cited), while Rose and Scott (1997), gave 100,000–1,000,000 individuals (which was a number based on Morrison *et al*. 1994). These estimates appear no more certain, and to have less evidence-based support, than the 1.5 million estimate.

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. Surveys were coordinated across Peru, Chile, Bolivia, and Argentina, most during a ten-day window in early February (Castellino & Lesterhuis 2020). A number of sites not covered by the simultaneous survey but surveyed for the Neotropical Waterbird Census (NWC) during the same period, and eBird additional records for the species from outside the scope of the survey area but within the same survey period were also included in the analysis. Unlike Jehl's 1988 surveys at migratory staging sites in North America, these surveys on the non-breeding grounds should have included all demographic groups, including juveniles. The total number of birds counted in the non-breeding range was 854,673. To estimate the species' population size, all surveyed sites were mapped with the coverage compared to habitat availability for the species within its core non-breeding area. The population size was estimated at 1,008,884 individuals, but based on coverage calculations and error estimations, it was suggested to set the estimate of the Wilson's Phalarope population at 1,000,000 individuals (Lesterhuis *et al*. in prep). Whether this estimate of 1,000,000 birds represents a decrease from the 1.5 million estimate in 1986 (Jehl 1988) is unclear, given the differences in methodology and areas covered of the two surveys (see Trends section).

Additional, coordinated surveys are needed at regular intervals, both at the North American staging sites and on the non-breeding grounds, to continue to develop a more accurate understanding of population and trends estimates.

Trends

Similar to the population status, trends of the population of Wilson's Phalaropes are somewhat uncertain because of limited data. In the early 20th century, population of the Wilson's Phalarope is believed to have undergone a significant decline due to loss of prairie and wetland habitats in North America (Lesterhis & Clay 2010, Colwell & Jehl 2020). However, the magnitude of this historic population decline is unknown. More recently, there are indications of a major population decrease occurring between the 1980s and 1990s. A recent analysis of International Shorebird Survey (ISS) data found an approximately 75% decline in abundance of the species in the U.S. and Canada from 1980–2019 (Smith *et al*. 2023). Most of that decline occurred in the period from 1980-1999, with a more stable population trend from 2009–2019 (Smith *et al*. 2023). Smith *et al*. (2023) considered their analysis robust, though most data included in the analysis were from the eastern U.S., where Wilson's Phalaropes occur in relatively smaller numbers (Smith *et al*. 2023). Nevertheless, the population decrease occurring in the 1980s and early 1990s observed by Smith *et al*. (2023) was consistent with a major decline in numbers at important staging sites in the 1990s (specifically Great Salt Lake, Lake Abert, and Mono Lake; Jehl 1999). 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). Likewise, Breeding Bird Survey results from 1980-1990 indicated a declining trend during that the 1980s and 1990s (Fig. 5).

The Breeding Bird Survey (BBS) provides another means of looking at Wilson's Phalarope trends, based on a long-term series of standardized surveys occurring across North America during the breeding season (Sauer *et al*. 2019). 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, Smith & Edwards 2021). Analysis of BBS data from the entire time-series (1966–2019) indicated a moderately declining trend survey-wide trend for Wilson's phalarope (-0.58% mean change per year; Table 2). Survey-wide trends for 3-generation periods (approximately 10 years for Wilson's Phalarope; Smith *et al*. 2023), were stable in the 1970s, negative in the 1980s and 1990s, positive from 2000-2009, and negative from 2010-2019 (Fig. 5).

Within specific BBS Bird Conservation Regions, there were strong (mean % annual change >1.0) declining trends 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

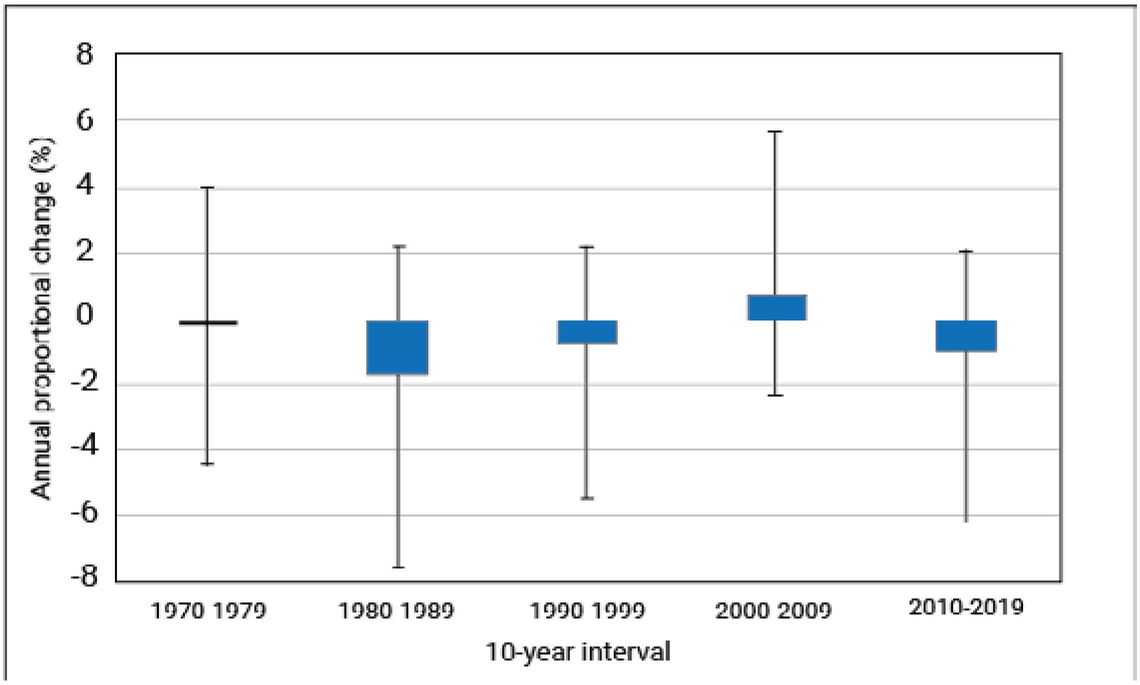

Figure 5: BBS trends for Wilson's Phalaropes over 10-year (3-generation) intervals. Trend is defined as yearly geometric mean of proportional changes in population size, expressed as a percentage. Error bars are upper and lower 95% CI. Trends shown are from results analysis reported at https://www.mbr-pwrc.usgs.gov/

Table 2. 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 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	2.1	Yellow	31
Central Mixed-grass Prairie	0.15	-3.17	3.86	Yellow	13

regions (Table 2). There were insufficient data to calculate reliable results for the Coastal California, Prairie Hardwood Transition, and Sierra Nevada 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). 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). In the past, BBS trends and trends from staging sites have not aligned well (Jehl 1988); further evaluation of the credibility of BBS trends, examination of the implication of varying regional trends within the BBS survey, and comparison of BBS and staging area survey trends would be useful.

Since 2019, surveys have been coordinated at six migratory staging sites: Great Salt Lake, Mono Lake, Lake Abert, Chaplin Lake, San Francisco Bay, and Owens Lake, with surveys every two weeks from July to September (Carle *et al*. 2022). High counts from those surveys (calculated as the highest total across all sites from a single week-long survey window) provide a comparison with Jehl's (1988, 1999) surveys of staging sites in the 1980s and 1990s. Combined annual high counts at the six staging sites surveyed in 2019-2022 averaged 274,690 birds (SD 76,399), 48% fewer than totals from 1986-1987 counts at the same sites (530,145 birds; Jehl 1988, Carle *et al*. 2022). Methodological and coverage details are important for such comparisons, and contemporary surveys differed methodologically at some sites from historical surveys. Nonetheless, contemporary surveys have shown consistently lower numbers than those reported by Jehl (1988, 1999). At specific staging sites, 2019-2022 peak numbers were generally lower than in the 1980s and similar to the lowest-count years in the 1990s (Jehl 1999, Carle *et al*. 2021). Given the great deal of inter-annual variation in numbers of Wilson´s Phalarope at staging sites (Jehl 1999), further effort is needed to understand trends. Likewise, work on building a standardized time-series of numbers from staging sites by examining the details of methodology and coverage of each survey is needed.

Most older trend estimates were considered uncertain for the species due to limited spatial coverage of phalarope habitat in surveys. Both the Canadian and U.S. Shorebird Conservation Plans (Donaldson *et al*. 2000, Brown *et al*. 2001, USFWS 2004) considered Wilson's Phalarope to show an "apparent population decline." Annual trends from 1974-1998 from the Maritimes Shorebird Survey and International Shorebird Survey (ISS) for Wilson's Phalarope indicated stable populations for the North Atlantic and Midwest regions (Bart *et al*. 2007). However, the range of Wilson's Phalarope only marginally overlapped with the geographic regions of that study (Bart *et al*. 2007). A more recent assessment reported "trend unknown" for Wilson's Phalarope, based on lack of data except from the Breeding Bird Survey (BBS; Andres *et al*. 2012).

Few data are available for analysis of trends at non-breeding areas outside of the North American staging sites. However, there is information that can be considered to assess the overall trends for the species at particular sites in the two main non-breeding regions. For the altiplano region of Bolivia, there was a major decline in numbers in the 2000s compared with the 1980s. None of the sites that had historical records counts 100,000 individuals in the region in the late 1970s and early 1980s have had such high counts recorded again. Counts carried out in nine years of the last two decades (between 1999-2021) for the species at these sites that had previous counts of ≥ 100,000 individuals yielded the following abundance ranges (min-max): Laguna Pastos Grandes 0-205; Laguna Chulluncani 0-407; Laguna Hedionda Norte 0-20,000; Laguna Calina 16-30,000; Laguna Colorada 7-7153 and Laguna Kollpa 0-149 birds (Omar Rocha, unpublished data).

At Laguna de los Pozuelos, Argentina, systematic surveys were conducted annually in February and July, between 2006 and 2021. During 8 of these 16 seasons the abundance of phalaropes was below 1% of the global population estimation (i.e., 10,000). In other seven seasons it was below 4% of the global population estimation (i.e., 40,000; Moschione & Sureda, *in litt*). These counts are far from the half million birds estimated in January 1988 by Moschione & San Cristóbal (*in litt*). However, in February 2021 an estimate of 449,108 Wilson´s Phalaropes were made for the site (Moschione & Sureda, *in litt*), reaching almost 45% of the 2020 global population estimation. These numbers are much closer to the historic counts for Laguna de los Pozuelos and support the importance of this site for the species during their non-breeding season.

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. The available data show that in the dynamic environment of the High Andes, the sites behave as a complementary network for phalaropes, with the use of individual sites depending on seasonal water levels, habitat conditions, and prey abundance.

Although there is less information from the Argentinian lowlands, recent observations at several saline lagoons of the Pampas region suggest similarly variable numbers and dynamically changing habitat. This landscape scale of habitat use must be considered in conservation strategies and actions planned and implemented for the species. Periodic but not systematic 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. From 2010 on, there are abundance data for each austral summer except 2013 and 2016. Estimated abundances during this period ranged from a minimum of 6,000 (March 2018) to a maximum of 613,640 individuals (February 2021; P. Michelutti, *in litt*). It is important to take into account that the month when the different surveys took place can have a great influence on the numbers recorded. Since 2020 there have been major efforts at Laguna Mar Chiquita to monitor the species on a monthly basis, to better understand fluctuations in abundances during the nonbreeding season. Maximum counts for summer seasons 2020-2021 and 2021-2022 surpassed the 50% of the global population estimation from Lesterhuis et.al (in prep). A potential bias in estimation must be considered given the challenge of accurately estimating numbers of birds within flocks this large.

Overall, Wilson's Phalarope trends remain unclear, with BBS data from the North American breeding range suggesting a moderately declining trend, phalarope-specific surveys from North American staging sites suggesting the possibility of a decline since the 1980s, and limited data from South America indicating highly variable numbers at key sites year to year. There is a clear need for better understanding the population trends of the Wilson's Phalaropes, which will require more focused, standardized monitoring of staging and non-breeding areas.

Important Sites

This section of the plan identifies the key sites of conservation importance for Wilson's Phalarope. We considered two different criteria:

Key sites of international importance: sites that are identified as holding 1% or more of the biogeographic population of the species. This is the standard criterion used by Ramsar and Important Bird Areas (IBAs), while WHSRN criteria considers sites holding 1% of the global population of one species as a site of regional importance. There are no significantly large discrete breeding populations of Wilson's Phalarope and there are also no subspecies, therefore 1% of the total population estimate is taken also to be 1% of the global population, currently estimated at 1,000,000 birds (Lesterhuis *et al.* in prep). Thus, any site holding 10,000 or more Wilson's Phalaropes qualifies as a site of international conservation importance for the species (Table 3).

Key sites of local importance: sites known to hold less than 1% of the global population but more than 0.33% (3,300 birds) are considered to be of local importance for the species (Table 4). Although this does not correspond to a Ramsar/IBA/WHSRN criterion, it was included to follow the same criteria considered in the previous version of the conservation plan (Lesterhuis & Clay, 2010).


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Table 3. Key sites of international importance for the species, meeting a 1% population threshold for Wilson's Phalaropes (≥10,000 individuals, based on a global population estimate of 1 million birds, Lesterhuis et al. in prep). 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.

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***)	122,850d (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)	230,000f (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
	Benton Lake NWR	Montana	20,000 (Jun 1986)	No data	No data	Jehl 1988

	Crescent Lake	Texas	18000	No data	No data	ISS
	Horsehead Lake	North Dakota	13,500 (Jul 1987)	No data	No data	Jehl 1988
	Tulare Lake Basin	California	12,000	No data	No data	ISS
	Between Cactus Lake and Etter, Moore County	Texas	10,000	No data	No data	Skagen *et al.* 1999
	Summer Lake	Oregon	10,000	No data	No data	Littlefield, C.D. 1990
	San Diego Bay	California	10,000 (Jul 1986)	No data	No data	Jehl 1988
	Lubbock	Texas	10,000	No data	No data	ISS
	Midland	Texas	10,000	No data	No data	ISS
	Pathfinder Reservoir	Wyoming	10,000	No data	No data	ISS
	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.*
	Piscinas artificiales de Ecuasal - Mar Bravo	Santa Elena	16,000 (Aug 2002)	44,309 (Aug 2014)	32227 (Aug 2021)	Agreda *et al.* 2009, A. Agreda *pers. comm.*, B. Haase *pers. comm.*
	Ciénaga de La Segua	Manabí	15,000	No data	No data	BirdLife 2005
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).
	RN Salinas y Aguada Blanca	Arequipa, Moquegua	20,000	537 (Feb 2021, **2**)	537 (Feb 2021, **2**)	FIR Ramsar 2003, WHSRN 2021, WHSRN 2021.

Peru / Bolivia	Lago Titicaca	Puna	12,906	No data	No data	Velarde Falconi 1998.
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***)	3302 (Feb 2020, ***2***)	Hurlbert *et al.* 1984, Omar Rocha *pers. comm.* WHSRN 2020.
	Laguna Chulluncani	Potosí	±25,000 (Feb 1979)	35 (Apr 2012, ***10***)	0 (Feb 2020, Feb 2021, ***2***)	Hurlbert *et al.* 1984, Omar Rocha *pers. comm.* WHSRN 2020, 2021.
	Laguna Saquewa	Oruro	11,034 (Feb 2006)	15,256 (Feb 2020, ***13***)	15,256 (Feb 2020, ***13***)	Omar Rocha *pers. comm.*, WHSRN 2020, WHSRN 2020
	Laguna Alalay	Cochabamba	10,000	129 (Feb 2020, ***1***)	129 (Feb 2020, ***1***)	Blanco & Canevari 1998, WHSRN 2020, WHSRN 2020
	Lago Poopó	Oruro	9,030 (Jan 2002)	18,614 (Sep 2012, ***13***)	1458 (Feb 2020, ***1***)	Omar Rocha *in litt*, Omar Rocha *in litt*, WHSRN 2020.
	Lago Uru Uru	Oruro	6,866 (Feb 2005)	21,064 (Sep 2012, ***13***)	2073 (Feb 2020, ***2***)	Omar Rocha *in litt*, Omar Rocha *in litt*. WHSRN 2020.
Paraguay	Lagunas Saladas – Riacho Yacaré	Presidente Hayes	25,000 (Nov 2000)	No data	No data	Lesterhuis & Clay 2001.
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
	Salinas Grandes	Córdoba	7,300 (Jan 2010)	15,000	15,000 (Feb 2020, ***2***)	D. Blanco 2010 (eBird list), M. Castellino unpublished data, M.Castellino unpublished data.
	Salinas Chicas	Médanos	SD	15,000 (Jun 2017, ***2***)	0 (Feb 2020, ***1***)	Castellino & Bucher 2017, 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.

Table 4. Key sites of local importance for the species, meeting a 0.33% population threshold for Wilson's Phalarope (≥3,300 individuals, based on a global population estimate of 1 million birds, Lesterhuis et al. in prep). 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.

Country	Site	State/Province	High count prior to 2010	High count 2010-2019	High count 2020-2022	Sources (historic, since 2010, and since 2020, respectively)
Canada	At 14 Lake Sites	Saskatchewan	8,230	No data	No data	ISS
USA	J. Clark Salyer NWR	North Dakota	6,000	No data	No data	WHSRN
	Long Lake NWR	North Dakota	6,555	No data	No data	Gregg Knutsen and Keri Lang (USFWS)
	Miller Lake	North Dakota	7,560 (Jul 1987)	No data	No data	Jehl 1988
	Kidder County Lakes	North Dakota	3,760 (Jul 1987)	No data	No data	Jehl 1988
	Kingsbury	South Dakota	6,200	No data	No data	Skagen *et al.* 1999
	Scott County	Kansas	4,000	No data	No data	Skagen *et al.* 1999
	Lake Thompson	South Dakota	3,592	No data	No data	Skagen *et al.* 1999
	Muleshoe NWR	Texas	4,500	No data	No data	Skagen *et al.* 1999
	Sheridan County	Nebraska	3,844	No data	No data	Skagen *et al.* 1999
	Minot sewage lagoons	North Dakota	3,500	No data	No data	Skagen *et al.* 1999
	Klamath	California	5,500	No data	No data	ISS
	Lake South of Westby	California	8,000 (Jul 1987)	No data	No data	Jehl 1988
	Crescent Lake NWR	Nebraska	8,055	No data	No data	Skagen *et al.* 1999
	Ruby Lake NWR	Nevada	5,000	No data	No data	ISS
	Owens Lake	California	SD	3,866 (Aug 2011, 10)	1,987 (Jul 2021, 3)	LADWP unpublished, Eastern Sierra Audubon unpublished

	Grass Lake NWR	Montana	4,200 (Jul 1987)	No data	No data	Jehl 1988
Bolivia	Laguna Verde	Potosí	5,000 (Dec 1975)	No data	0 (Feb 2020, 1)	Hurlbert *et al.* 1984, WHSRN 2020.
	Laguna Alalay	Cochabamba	10,000	No data	129 (Feb 2020, 1)	Blanco and Canevari 1998, WHSRN 2020.
	Laguna Capina	Potosí	7,992] (Feb 2000)	361 (Nov 2011, 10)	4 (Feb 2020, 2)	Omar Rocha *in litt*, Omar Rocha *in litt*, WHSRN 2020.
	Laguna Chojillas	Potosí	"hundreds" (Feb 1979)	8,145 (March 2013, 10)	67 (Feb 2021, 2)	Hurlbert *et al.* 1984, Omar Rocha *in litt*, WHSRN 2021.
	Laguna Colorada	Potosí	5,000 (Jan 1979)	7,153 (March 2013, 10)	1291 (Feb 2020, 2)	Hurlbert *et al.* 1984, Omar Rocha *in litt*, WHSRN 2020.
	Laguna Kollpa Khota	Potosí	No data	149 (March 2013, 10)	0 (Feb 2020, Feb 2021, 2)	Omar Rocha *in litt*, WHSRN 2020, 2021.
Chile	Laguna Los Palos	12th Region	No data	5,525 (Nov 2018)	No data	Ricardo Matus *in litt*
Argentina	Reserva Natural Guatrache	La Pampa	5,000	No data	300 (Feb 2020, 1)	F. Bruno/ D. Acevedo/ R. Olivera unpublished, WHSRN 2020.
	Bajo Giuliani	La Pampa	7,000	No data	9 (Feb 2020, 1)	F. Bruno unpublished, WHSRN 2020.
	Reserva Natural Chadilauquen	La Pampa	7,000	No data	No data	F. Bruno/ D. Acevedo/ R. Olivera unpublished
	Estancia El Fogón	Cordoba	6,750	No data	No data	NWC

Conservation Status

Conservation status at International and National levels

Wilson's Phalarope is listed as Least Concern by IUCN (BirdLife International 2016) and the population trend for the species is considered as Increasing (BirdLife International 2023). However, this may not reflect the current state of the population and needs review based on updated information from recent years. Based on the Smith *et al.* (2023) and Jehl (1999) analyses, the apparent population decline of Wilson's Phalarope in the 1980s and 1990s potentially qualifies the species for IUCN Red List "Endangered" status under the criteria of a ≥50% decline over any 10-year or 3-generation period (IUCN 2012). It is also necessary to consider the species near obligate relationship to hypersaline lakes and the high degree of vulnerability associated with the fact that a large number of birds congregates in a few specific places at the same time.

Canada: Wilson's Phalarope is included in the Canadian Shorebird Conservation Plan (Donaldson *et al.* 2000), where it is listed as a Species of High Concern. It was also listed as High Concern in a recent update on the conservation status of shorebirds in Canada (Hope *et al.* 2019).

United States: Wilson's Phalarope was included in the U.S. list of Birds of Conservation Concern 2002 (USFWS 2002) due to widely reported declines. However, the species has not been included in the more recent Birds of Conservation Concern 2008 or 2021 (USFWS 2009, USFWS 2021). It is listed as a species of Least Concern in the U.S. Shorebird Conservation Plan (U.S. Shorebird Conservation Plan Partnership 2016). 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.* 2000). 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 *et al.* 2013). At the state level in the United States, Wilson's Phalarope is listed as Threatened in Minnesota (Minnesota DNR 2021) and a Level


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I Species of Conservation Priority in North Dakota, South Dakota, and Wyoming (Butler *et al.* 2014).

Wilson’s Phalarope has been included as focal species in the Midcontinent Shorebird Conservation Initiative (MSCI), which is currently in development, led by the US Fish and Wildlife Service, Environment and Climate Change Canada, Conservation of Arctic Flora and Fauna through the Arctic Migratory Bird Initiative, and Manomet-WHSRN.

In South America, Wilson’s Phalarope is mentioned in the National Shorebird Conservation Plan for Argentina among the shorebird species that require special attention when planning and executing actions of habitat conservation and management at the site scale, but due to the criteria adopted the species it was not included in this version of the document as focal species. At the country level, in Argentina the species is considered Not Threatened (MADS and Aves Argentinas, 2017). Wilson’s Phalarope is being incorporated in the Plan Nacional de las Aves Playeras currently under development in Chile. No specific information about the species at national level is available for Bolivia, Peru, or Paraguay.

Threats

The Wilson’s Phalarope global population is under pressures and threats that put its conservation at risk. The survival of the species is threatened due to the loss or alteration of key habitats or food resources. Over half of the global population relies simultaneously on few key sites at very specific times of the year. The loss of habitat and food supplies at these few sites can have a profound impact on the species.

Of particular concern is the current status of many saline lakes in the Great Basin region of the western United States. Great Salt Lake’s surface elevation had dropped to 4188’ in late 2022, its lowest level in recorded history (1847-present; USGS 2023). This decrease was driven primarily by upstream water diversions for agriculture and urban use (Null & Wurtsbaugh 2020). In late 2022, the lake had lost >50% of its volume compared to the volume at an average lake elevation over the historical time-series (Null & Wurtsbaugh 2020, Hall *et al.* 2023). Great Salt Lake also had reached salinity levels in 2022 of >180 g/L (USGS 2023). These elevated salinity levels were above the maximum salinity tolerance threshold for healthy reproduction and survival of the lake’s brine shrimp (160 g/L; Marden *et al.* 2020). Brine flies have a higher salinity tolerance than brine shrimp but experience reduced growth at salinities above >120 g/L (Null & Wurtsbaugh 2020). Also, microbialite mats in Great Salt Lake, composed of cyanobacteria and algae, form an important rock-like substrate where brine fly larvae feed and pupate (Lindsay *et al.* 2019). The cyanobacteria that form these mats also are threatened by the high salinities (Lindsay *et al.* 2019). As Great Salt Lake’s water levels fell to record low levels in 2022, 40% of the lake’s microbialites were exposed to air and desiccated (Frantz *et al.* 2023). These changes are very recent, and to date there are few data showing the impact on flies from the loss of submerged microbialites and elevated salinities at Great Salt Lake. However, preliminary data indicated a major decrease in fly populations during 2022 (Larsen 2022, https://www.sltrib.com/news/environment/2022/11/08/great-salt-lakes-ecological/ ,Walter 2023, https://www.thenation.com/article/environment/dust-great-salt-lake/). Fundamentally, the low water and rising salinity threaten to destroy the base of the food web that phalaropes rely on (Baxter & Butler 2020, Lindsay *et al.* 2020, Frantz *et al.* 2023).

Similar situations are occurring in other important migratory staging sites in the Great Basin. Lake Abert, Oregon, became completely dry in 2014-2015 and again in 2021-2022 due to upstream water diversions and drought (Larson *et al.* 2016, Moore *et* *al.* 2016, Hall *et al.* 2023). The only water present in 2022 was localized spring water from adjacent slopes surfacing on the east side of the lakebed, which supported small numbers of phalaropes (Hall *et al.* 2023). These periods of desiccation resulted in exceptionally high salinities exceeding 180 g/L (Larson *et al.* 2016). These salinities were well above the salinity tolerance thresholds for Lake Abert's brine shrimp and alkali flies (Marden *et al.* 2020; Herbst 2023) and resulted in brine shrimp die-offs and decreased alkali fly abundances (Larson *et al.* 2016).

At Mono Lake, beginning in 1941, two of Mono Lake's major tributary streams were entirely diverted for use by the City of Los Angeles, causing the lake to decline by an average of one to two feet a year up to 1970 (Winkler 1977). 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). Driven by water diversions, the salinity of the lake nearly doubled between 1941 and 1982 (from 48 to 93 g/l), and the invertebrate foundation of the ecosystem was predicted to collapse if diversions continued (Herbst & Bradley 1993, Dana & Lenz 1986). Fortunately, Mono Lake was the subject of an intense legal battle over its water, which resulted in the California Supreme Court deciding in favor of restoring the lake (Blumm & Schwartz 1995). In 1994, the California State Water Board issued restoration orders to bring the lake to a management level of 6,392' above sea level and mandated the City of Los Angeles to reduce water diversions (Loomis 1995). Thanks to this restored water allocation, Mono Lake reached a high-stand of 6,385.1 in 1999. Since then, however, the lake level has fluctuated in the range of 6,378-6,384 for >20 years, and in 2022 the lake remained 13 feet below the mandated management level (the lake was at 6,378.4 in December 2022; Mono Basin Clearinghouse, 2023). Model-based predictions that the lake would reach the mandated 6,392' lake level in 20 years after the 1994 decision did not match reality, apparently due to mismatches between the 30-year period the models used (a relatively wet period) and the observed precipitation and evaporation in the last 30 years (McQuilken 2023). Although the lake still has not reached the mandated water level, water diversions to Los Angeles have not been adjusted. As of 2023, the California State Water Board was considering conducting new hearings about Mono Lake's water management. Thus, though Mono Lake is the best protected of major saline lake migratory sites, and has a dedicated water right, with continuing water diversions Mono Lake is predicted to remain below the 6,392' management level (McQuilken 2023). Because of this, it will continue to be vulnerable to decreased in water level and increases in salinity during the extreme, prolonged drought periods predicted to occur more frequently in the region with climate change (Seager *et al.* 2007, Ficklin *et al.* 2013).

Fortunately, Great Salt Lake, Lake Abert and Mono Lake's lake levels all rose by multiple feet in 2023 due to an exceptionally wet winter in the western U.S. in 2022-2023 (USGS 2023; Mono Basin Clearinghouse 2023, Ron Larson, *pers. comm.*). These rises decreased salinities (USGS 2023) and alleviated crisis-level ecosystem impacts (McQuilken 2023, Ron Larson, *pers. comm.*). However, these gains are likely to be short-term unless sustainable long-term solutions are found to address issues arising from diversion of freshwater inputs to the lakes, which are the primary drivers of their recent declines (Moore *et al.* 2016, Wurtsbaugh *et al.* 2017, McQuilken 2023).

These North American saline lake sites are linchpins for the survival of the Wilson's Phalarope. However, Wilson's Phalaropes are also dependent on threatened South American saline lakes during most of their non-breeding period. Many of these sites are critically threatened by water diversion as well, but their situations are less well-documented. A primary -and rapidly growing-threat to non-breeding sites is water extraction from aquifers for lithium mining (see Energy Production and Mining section). The simultaneous loss of saline lake habitat in North and South America could have profound negative consequences for Wilson's Phalaropes in the near future.

We approached the evaluation of threats to Wilson's Phalaropes in two ways. First, we conducted a survey of partners from 22 key sites for the species and asked them to identify the main threats at each of their sites for the species (Table 5). Only 18% of the sites consulted reported no existing threats for Wilson's Phalarope. The threats that were reported for the largest number of sites (out of a total of 22 sites) were: 1) Drought processes related to climate change (65% of the sites) and 2) Disturbances caused by human activities (55%; Table 5). Water and agriculture (including the effects of agrochemicals) were reported for 18% of the sites. Fourteen percent of the sites reported that regulated water extraction is a threat to phalarope conservation, while 9% reported that unregulated water extraction and the effects of climate change (not related to drought processes) affect their conservation. Finally, 5% of the sites reported that the introduction and expansion of exotic species and nest predation are threats to the conservation of the species.

It is important to clarify that these percentages are based on the responses from our survey and are not necessarily related to the magnitude of the negative effect they may have on the conservation of the species, since a threat identified in a few sites can have very important negative effects on a high percentage of the population (e.g., water extraction) and vice versa. In addition, not all sites that responded to the survey are used by the species during the same time of year, so certain threats that were reported at a low percentage of total sites may be underrepresented (e.g., nest predation only occurs at breeding sites, which were a low number of sites within the 22 that completed the survey).

All the information gathered in the surveys and from the bibliography available is compiled and incorporated into Table 5, summarizing the main threats affecting the conservation for each site:

Table 5. Summary of the current threats for key sites identified for Wilson's Phalaropes. Disturbances (tourism and human activities) category includes a wide variety of human-related disturbances, from productive activities to tourism, and with different impacts on the species and the site. *Sites for which local partners responded to the threat survey designed specifically for this Conservation Plan.

Site name	Country	Main threats to sites	Category of protection / Designations
Chaplin and Reed Lakes*	Canada	Climate change (related to drought); climate change (related to ecosystem changes other than drought); nest predation.	WHSRN; IBA
Cheyenne Bottoms*	U.S.A	Climate change (related to drought); introduction or expansion of exotic species.	WHSRN; Ramsar; IBA.
Great Salt Lake*	U.S.A	Regulated water extraction (upstream freshwater diversion and lake water for mineral extraction); climate change (related to drought); introduction or expansion of exotic species.	WHSRN; IBA.

Mono Lake*	U.S.A	Regulated water extraction (upstream freshwater diversion); climate change (related to drought); climate change (related to ecosystem changes other than drought).	WHSRN; IBA; Living Lakes.
Lake Abert*	U.S.A	Regulated water extraction (upstream freshwater diversion); unregulated water extraction; climate change (related to drought).	-
Bahía Santa María*	Mexico	Climate change (related to drought); agriculture (agrochemicals).	WHSRN.
Bahía de Ceuta*	Mexico	Agriculture (agrochemicals); anthropic disturbances (tourism and human activities); erosion and natural events.	WHSRN; Ramsar.
Piscinas de Ecuasal (Mar Bravo)*	Ecuador	Anthropic disturbances (tourism and human activities).	WHSRN; IBA.
Piscinas de Ecuasal (Pacoa)*	Ecuador	No threats identified.	WHSRN; IBA.
Laguna Pastos Grandes*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities), lithium mining.	Ramsar; IBA.
Laguna Loromayu*	Bolivia	No threats identified.	Ramsar; IBA.
Laguna Hedionda Norte*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities); water pollution.	None.
Laguna Calina*	Bolivia	No threats identified.	Ramsar; IBA.
Laguna Colorada*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities).	Ramsar; IBA.
Laguna Kollpa*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities).	Ramsar; IBA.
Laguna Chulluncani*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities).	Ramsar; IBA.
Lago Poopó*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities); water pollution.	Ramsar; IBA.

Lago Uru Uru*	Bolivia	Climate change (related to drought); anthropic disturbances (tourism and human activities); water pollution.	Ramsar; IBA.
Laguna Saquewa*	Bolivia	No threats identified.	None.
Salar de Surire	Chile	Mining; water extraction.	Ramsar; IBA.
Laguna Palar	Argentina	Mining; erosion and natural events.	Ramsar; IBA.
Laguna Guayatayoc	Argentina	Overexploitation of native species for firewood; overgrazing; hunting; natural events.	IBA.
Laguna de los Pozuelos	Argentina	Mining; overgrazing; overexploitation of native species for firewood; erosion and natural events.	IBA; WHSRN; Reserva de la Biósfera.
Laguna Mar Chiquita*	Argentina	Unregulated and regulated up stream water extraction; agriculture (agrochemicals); anthropic disturbances (tourism and human activities); water pollution.	WHSRN; Ramsar; IBA.
Lago Epecuén*	Argentina	Agriculture (agrochemicals); anthropic disturbances (tourism and human activities).	IBA.
Salinas Chicas	Argentina	Mining; anthropic disturbances (tourism and human activities); climate change (related to drought).	IBA.

Second, and based on the information gathered from the surveys and from literature, we applied the Conservation Measures Partnership Direct Threats Classification (CMP 2016) to assess conservation threats for Wilson's Phalarope, based on the best information available for the species. All conservation threats identified in the CMP classification system were considered, but only those that currently apply to Wilson's Phalarope are presented here. The threats are presented in the order used by this classification system, not in a priority order:

1. Residential & Commercial Development

The drainage of wetlands and the conversion of associated grasslands to residential and commercial developments have resulted in the loss and degradation of habitat (e.g., through changes in hydrological and chemical regimes of wetlands) for Wilson's Phalarope throughout its breeding and migration range. However, such geographically focused developments are of less concern when compared to the massive habitat loss resulting from other activities, such as agricultural expansion.

Although the impact they may have on the population of Wilson's Phalarope is unknown, in recent years there has been an increase in infrastructure related to tourism (hotels and inns, roads, and other development) in different important sites for the species in the High Andes of Bolivia, such as Laguna Colorada, Laguna Chulluncani and Laguna Hedionda Norte (Omar Rocha, *in litt*). This could have a direct negative impact due to the increase in disturbances in these sites from increased human presence, and an indirect impact through habitat degradation.

In a few cases, commercial developments can favor the species, through the development of saline lagoons or basins for mineral extraction (e.g., the lagoons at Salinas, Ecuador; Chaplin Lake, Saskatchewan) or for mining (Salar de Surire, Chile; see section 3. Energy Production & Mining).

2. Agriculture & Aquaculture

The massive loss of prairie wetlands in North America is believed to have had a significant impact on the past and current population of the species (Jehl & Colwell 2020). Once a vast expanse of grasslands with numerous wetlands, the prairies are now an agrarian system dominated by croplands (Dahl & Johnson 1991).

The remaining prairie wetlands are impacted by a number of agricultural practices that result in elevated sedimentation rates (Martin & Hartman 1987, Gleason & Euliss 1996), unnatural variance in water-level fluctuation (Euliss & Mushet 1996) and altered vegetative communities (Kantrud & Newton 1996).

One of the most severely affected parts of the prairies is the Prairie Pothole Region (covering the U.S. states of Iowa, western Minnesota, North Dakota, South Dakota, and north- eastern Montana, and the Canadian Provinces of Alberta, Saskatchewan, and Manitoba). More than half the historic Prairie Pothole wetlands have been lost (Dahl & Johnson 1991), and in the eastern parts of the region (e.g., Minnesota) fewer than 10% of the original wetlands and less than 1% of the native prairie grasslands still exist. Nearly 70% of the original Prairie Pothole grasslands now support crop production. Conversion of grassland to cropland peaked in the 1920s (Dahl & Johnson 1991) and grassland conversion is concentrated in close proximity to wetlands, resulting in loss of suitable breeding habitat for Wilson's Phalarope (Dahl & Johnson 1991, Wright & Wimberly 2013). Although by the1960s it was generally believed that all areas appropriate for agriculture had been converted, technological advancements and economic pressures continue to drive the conversion of more marginal areas (Dahl 2005). Conversion of pastures to corn and soybean crops within the Prairie Pothole Region has increased in recent years in response to increasing demand for biofuel feedstocks (Wright & Wimberly 2013, Alemu *et al.* 2020). Pastures and grasslands in the U.S. portion of the Prairie Pothole region experienced a significant net decreasing trend of almost a third between 2006-2018 (Alemu *et al.* 2020). However, recent habitat restoration and wetland re-establishment efforts have resulted in a net positive trend in wetland extent in the U.S., with a nearly 281,500 hectares increase in freshwater ponds 1998–2004 (Dahl 2005) and a 1% increase in emergent freshwater marsh from 2004-2009 (Dahl 2011). However, loss of prairie pothole wetlands (a habitat used by breeding Wilson's Phalaropes) continued to be greater than habitat gains in some mid-western states between 2004–2009 (Dahl 2011, Wright & Wimberly 2013, Alemu *et al.* 2020).


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Habitat loss from agriculture appears to be less of a concern on the species' main non-breeding grounds, particularly because the species depends on a different type of habitat during this period. However, many wetland areas of the species' migration and non-breeding range have also been extensively modified by agriculture. Latin America is one of the world regions with the highest rates of change in land use and land cover in recent times. Argentina, Brazil, and Uruguay are among the countries with the greatest changes in the area planted with agricultural crops (Baeza & Paruelo 2020). The Pampas grasslands of Argentina, for example, have been extensively altered by agriculture (Bucher & Nores 1988, Soriano 1992), with a more than 60% decrease in the extent of grasslands in the Argentine Pampas from 1880–2000 (Viglizzo & Frank 2006). There has been a particularly rapid loss in recent decades through agricultural intensification and a shift from cattle ranching to crops in the most fertile grasslands (Viglizzo *et al.* 2005). In this region, channelization to reduce or control floods often favors agriculture expansion at the expense of wetlands (Brandolin *et al.* 2013). Agriculture not only causes habitat loss, but also the pollution derived from pesticides and herbicides was identified as a main threat to habitat and biodiversity integrity in almost 20% of the sites surveyed (see section 6. Pollution).

The development of shrimp farms has also been a major factor in habitat loss and fragmentation in migration sites. Shrimp farming is expanding greatly in the Pacific tropical coastal wetlands that support large concentrations of migratory shorebird populations (Páez-Osuna *et al.* 2003, Navedo *et al.* 2017, Navedo & Fernandez 2018). For other shorebird species, farms can offer complementary feeding habitat since after being harvested, the pools can simulate intertidal feeding areas for a short period of time (Navedo *et al.* 2015, Navedo & Fernández 2018). Due to their aquatic habits and foraging ecology, phalaropes are not among the main species that use shrimp farms as foraging habitat. Thus, the exponential growth of shrimp farms in Central America and the consequences shrimp farms have on coastal wetlands (e.g., loss of natural habitats like saltmarshes and mangroves, alteration of physical and biological processes, changing nutrient cycles and increasing environmental pollution through effluents, Navedo *et al.* 2017) could have negative effects on Wilson's Phalaropes during its migration by altering their migration sites.

Harvesting of Artemia spp. (brine shrimp) occurs at some U.S. staging sites, including currently at Mono Lake and Great Salt Lake and historically at Lake Abert. There is no evidence of a negative impact of Artemia harvesting on phalaropes, but the subject has also not been studied. Moreover, such harvests can provide an economic rationale for maintaining water and salinity levels appropriate for invertebrates and, hence, for phalaropes and other birds.

3. Energy Production & Mining

Among the several human activities that physically disturb salt lakes, mining is the most important. Mining especially threatens ephemeral lakes. Rarely, if ever, is such damage repaired after mining has ceased (Williams 2002). In South America, the core wintering habitat of the species falls within the "lithium triangle" in the high Andes of Argentina, Bolivia, and Chile (Hurlbert 1984, WHSRN 2020, Gajardo & Redón 2019, Heredia *et al.*) This area could host up to 68% of world's lithium salt brine reserves (Sticco 2021). Lithium mining development has been increasing rapidly in the region as demand for lithium-ion batteries for electric cars increases (Martin *et al.* 2017, Garcés & Alvarez 2020). World lithium production has increased more than 8-fold from 1995-2017 and is expected to increase at a rate of 10% per annum or more in the next two decades (Maxwell & Mora 2020). 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).

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). Approximately 0.4-0.5 million liters of brine is 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, Marconi *et al.* 2022). Depending on the technology used and lithium concentrations, 80,000-140,000 liters of freshwater are used per ton of lithium production (Marconi *et al.* 2022). 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). The altiplano has a naturally negative water balance, where annual evaporation exceeds annual precipitation and groundwater discharge often exceeds groundwater recharge (Marconi *et al.* 2022). There is evidence that interbasin groundwater flow is important in the modern hydrologic balance and that fossil groundwater (i.e., from 100–10,000 or more years age) is the dominant water discharge in wetlands in the region (Marconi *et al.* 2022). Thus, though hydrologic connections in the altiplano are not well understood, it is known that water balances 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 200 l/s for 25 years (Acosta & Custodio 2008). However, current brine water use by mining in the nearby Salar de Atacama is estimated at over ten times that, at 2,142 l/s, daily (Gajardo & Redón 2019).

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, Gajardo & Redon 2019). Observed negative impacts to date include a direct negative correlation between lithium mining and abundance of two species of flamingos and decline in winter surface water in the Salar de Atacama (Gutierrez *et al.* 2022). 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 absence of better governance, regulation, and mitigation of lithium mining (Gutierrez *et al.* 2022, Izquierdo *et al.* 2015, Marconi *et al.* 2022). There has been no research directly on the impacts of lithium mining on Wilson's Phalaropes, other than surveys showing the altiplano region is of great importance to a large proportion of the world population (Castellino & Lesterhuis 2020, Hurlbert *et al.* 1984, Jehl 1988). Approximately 25% of Wilson's Phalaropes recorded in a survey across wintering habitat in South America were in the High Andes region known as the "lithium triangle" (Castellino & Lesterhuis 2020).

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Apart from the physical disturbance and water consumption caused by mining (Sticco 2021), the activity may impact salt lakes in other ways, particularly by adding pollutants, which can have various effects depending upon the pollutants involved (Williams 2002). Although in many of these sites the impact of the activity on the presence and abundance of the Wilson's Phalaropes is not known, some evidence indicates that not all mining activity is detrimental to the species. For example, in semi-arid regions, the construction of solar salt ponds (from which salt is obtained by the evaporation of seawater or saline groundwater) provides a unique example of unnatural saline water-bodies that have been constructed to 'mine' salt from the sea or underground and may provide additional habitat (e.g., Chaplin Lake, Canada; Salinas, Ecuador). For example, Glauber's salt solution mining operation at Chaplin Lake, Canada, is thought to maintain water levels at optimal depth for shorebirds (Beyersbergen & Duncan 2007). Another example is Salar de Surire, Chile, where the preferred area used by Wilson's Phalarope is an artificial lake generated by the mining company in which the salinity is higher than the surrounding waters (V. López, Oikonos, Ecosystem Knowledge, *pers. comm.*). An artificial salt pond complex associated with mining in Silver Peak, Nevada in the U.S. also supported migratory Wilson's Phalaropes (Jehl & Miller 2020).

4. Human Intrusions & Disturbance

Disturbance of phalaropes has not been identified as a major threat at most North American sites according to our survey, nor has it often been monitored.

At Mono Lake, California, large flocks of phalaropes sometimes congregate in places with groves of submerged tufa rocks, which are also popular non-motorized boating destinations. Hundreds to thousands of birds are frequently disturbed by people in canoes or kayaks or by pedestrians with unknown impacts (Oikonos, unpublished data). However, in some years phalaropes do not congregate in areas overlapping with human recreational use, so disturbance and its impacts appear highly variable at Mono Lake (Oikonos, unpublished data).

Human disturbances for recreational activities are present in many sites in South America, like Laguna Mar Chiquita, an important tourist hotspot in Córdoba (Argentina). At this site activities like kitesurfing, motorized and non-motorized boating, recreational flights in motorized paragliders and small airplanes, and ATV driving on the shores are increasing, and remain poorly regulated. A study during austral summer 2021-2022 assessed disturbance on shorebirds in 10 sites along the south shore of Laguna Mar Chiquita and showed that pedestrians were the most frequent disturbance agent present on the shores, but they did not generate major disturbances on the birds present. The largest disturbances were caused by aerial vehicles such as paragliders and small airplanes, which caused 90% of the waterbirds and 100% of the shorebirds present to leave the site (Manomet & Aves Argentinas, 2022). It's expected that with the recent designation of the Ansenuza National Park on the site and with the consequent development of management plans for the protected areas (National Park and Provincial Reserve) more regulation of recreational and economic activities will help reduce disturbances. At some sites of the altiplano region there is a growing development of tourist activities, for example in Laguna Hedionda Norte and Laguna Colorada, Bolivia (Omar Rocha, BIOTA, *in litt*). However, the impact of these recreational activities on Wilson's Phalarope is unknown.

5. Natural System Modifications

The most important threat related to natural system modifications is water management and use (see the introduction to the Threats section, above). Wetland drainage to enhance agricultural production has been the primary factor resulting in the loss of wetlands in North America (Tiner 1984, Millar 1989, Dahl 1990, Dahl & Johnson 1991). Extraction of water for agricultural and/or urban use is a particular concern at many sites that is greatly exacerbating natural fluctuations in water levels and the effects of climate change (Wurtsbaugh *et al.* 2017, Kintisch 2022). A decision in 2023 by the United States Supreme Court to end Clean Water Act protection of a seasonal wetlands could be detrimental to phalarope breeding habitat in the U.S. (https://www.nytimes.com/2023/08/29/climate/epa-wetlands-protection-rollback.html).

In the case of saline lakes, the major challenge is to ensure sufficient water inflow, which has been and continues to be changed in many cases through water diversion (Williams 2002, Moore *et al.* 2016, Wurtsbaugh *et al.* 2017). Decreased inflow leads to reduced lake surface and increased salinity levels, which result in a decreased mass of invertebrates and changes in the species composition, negatively affecting the ability of Wilson's Phalaropes to replenish their fat reserves (Andrei *et al.* 2009). Decreasing water levels and increasing salinity levels also can affect within-site habitat quality and phalarope habitat use (Senner *et al.* 2018). For example, Wilson's Phalaropes prefer to use shallower parts and brackish waters of Great Salt Lake, areas that are especially impacted by the decreased freshwater inflow into the lake (Frank & Conover 2019).


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In South America, the altiplano lakes favored as non-breeding sites are also threatened by water extraction, primarily for mining (see 3. Energy Production & Mining section). At Laguna Mar Chiquita, there is a growing threat of increasing water diversion and development of dams and irrigation channels on the Dulce River, mainly for expansion of irrigated agriculture, but also for industry and human consumption (Bucher & Curto 2012). In some cases, the water diversion is regulated, but illegal channels for water diversion also frequently occur along the Dulce River. The increasing development of dams and irrigation infrastructure on the Dulce River affects both the amount of water reaching Laguna Mar Chiquita and alters the natural flood cycles that are essential for the ecological functioning of the system (Bucher 2006). Currently, there is not enough information or data available to understand the magnitude and impact of water diversion at Laguna Mar Chiquita.

6. Invasive and problematic species.

Nest predation is a potentially important limiting factor for Wilson's Phalarope. During a 6-year study in Saskatchewan, Canada, nest success varied from 17% to 56% of clutches hatching ≥1 egg (Colwell & Oring 1988). Most clutch loss (on average 59% of failed clutches) was due to predation, with the next highest losses (average 17%) resulting from abandonment, which often follows partial clutch loss to predators (Colwell & Oring 1988a). During this period, unknown predators killed 1% (n = 275) of incubating males. Colwell (1992) showed that predation rates are highest during years of greatest human activity around nests. In studies in North Dakota and Alberta, 96% (n=23) and 85% (n=7) of clutches, respectively, failed to hatch at least one chick (Höhn 1967, Kagarise 1979). Observed predators of eggs and young include garter snake (*Thamnophis spp*.), various gulls (*Larus spp*.), American Crow (*Corvus brachyrhynchos*), Red-winged Blackbird (*Agelaius phoeniceus*), Yellow-headed Blackbird (*Xanthocephalus xanthocephalus*), Striped Skunk (*Mephitis mephitis*), Franklin's Ground Squirrel (*Spermophilus franklini*), Richardson's ground squirrel (*Urocitellus richardsoni*), and Northern Raccoon (*Procyon lotor*) (Kagarise 1979, Colwell & Oring 1988).

Another potential threat that needs further investigation is the introduction of exotic species. The remaining prairie grasslands in North America are already threatened by the expansion of non-native weedy species such as European cheatgrass (*Bromus tectorum*; Mack 1981); it is unknown whether this species provides suitable breeding habitat for Wilson's Phalarope.

The Great Salt Lake wetlands are threatened by a number of invasive plants, and the most problematic is the non-native phragmites (*Phragmites australis*). This species is widespread and abundant around the lake, despite the efforts made to limit its coverage. Naturally unvegetated areas, such as mudflats and drawdown areas that are critical shorebird habitat, are very susceptible to phragmites invasion. The fundamental alteration to wetland plant communities (composition and structure) and the macroinvertebrates they support have been observed with concern by Great Salt Lake wetland stakeholders, who in 2018 ranked phragmites as the second most important threat to these wetlands (Kettenring *et al.* 2020).


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Breeding and nonbreeding lagoons may be abandoned by Wilson's Phalarope when introduced exotic fish species (i.e., Rainbow Trout *Oncorhynchus mykiss*) cause dramatic trophic changes in lagoons (Fjeldså *in litt*. 2009).

7. Pollution

As described previously, Wilson's Phalaropes depend on saline and hypersaline wetlands during migration and non-breeding seasons. In most cases these wetlands are endorheic basin terminal lakes. This implies that the pollutants and effluents that can be transported by inflowing water will inevitably end up in the lake, potentially generating changes in the ecosystem (such as acceleration of the eutrophication process due to massive discharge of nutrients) or bioaccumulating in living organisms and finally in the sediments.

In North America, the remaining prairie wetlands are impacted by a number of agricultural practices that result in drift of agricultural chemicals into wetlands (Grue *et al.* 1989) and large inputs of nutrients (Neely & Baker 1989). It would seem likely that there are both direct (through exposure to agrochemicals) and indirect (through changes in wetland ecology) impacts to Wilson's Phalarope. Exposure to agrochemicals may cause death or reduce longer-term survival and/or reproductive rates (Blus & Henry 1997). Birds are susceptible to agrochemicals during most of their annual cycle. For example, significant levels of agriculture-derived contaminants were found at Laguna Mar Chiquita (Ballesteros *et al.* 2014). The Puna and Paramo ecosystems are increasingly being cultivated for potatoes and other crops (Rolando *et al.* 2017), leading to greater run-off of sediments and agrochemicals into the high Andean lakes. Household sewage and urban wastewater is another threat at many saline lakes. At the hydrological basins of Lake Titicaca, the Desaguadero River, Lake Poopó and the Salt Lake of Coipasa, growing urban centers have added pollution due to inadequate sewage systems and intensive groundwater use (Jellison *et al.* 2004). At Lake Uru Uru the pollution due to sewage water discharges has been increasing in recent years (Omar Rocha, *in litt*).

One of the three tributary rivers of the Laguna Mar Chiquita, the Suquía River flows for about 40 km through Córdoba, a city with over 1 million inhabitants. This has a negative impact on the river's water quality, which is particularly serious after the city's sewage discharge (Pesce & Wunderlin 2000, Monferrán *et al.* 2010). The operating capacity of the only sewage effluent treatment plant for Córdoba city had been exceeded since the 1990s. However, at the beginning of 2022, an expansion of the effluent treatment plant was finished, allowing the treatment of all the effluents produced by the city, thus reducing the enormous environmental impact caused by the release of untreated sewage effluents into the Suquía River for more than 20 years (https://prensa.cba.gov.ar/gobierno/bajo-grande-ya-funciona-la-nueva-planta-de-tratamiento-de-liquidos-cloacales/).

8. Climate Change

Globally, the Intergovernmental Panel on Climate Change predicts that global temperatures will rise between 1.4 and 5.8°C by 2100, with global-scale impacts to ecosystem function. Anthropogenic-driven climate change is one of the main threats that ecosystems in general are facing, and important sites for Wilson's Phalaropes, especially wetlands, are already experiencing impacts of climate-driven aridification and other changes (Langham *et al.* 2015, Donnelly *et al.* 2020, Hall *et al.* 2023). At present, in most cases globally, climate change is a secondary, exacerbating factor in the decline of water levels of saline lakes used by phalaropes, with water diversion usually being the larger and more immediate threat (Wurtsbaugh *et al.* 2017).


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Water diversions for agriculture, urban use, and mining are amongst the main causes of water diversion and alteration of water levels at important sites for the species (see section 5. Natural System Modification). This negative effect of water diversion on saline lakes are magnified by climate change effects, including higher temperatures, decreased snowpack, and a shortened hydro-period in the Great Basin region (Haig *et al.* 2019, Donnelly *et al.* 2020, Hall *et al.* 2023). Terminal lakes that are key migratory stopping sites and key non-breeding sites are highly susceptible to the impact of climate change because their water levels depend on the balance between water entering the system through precipitation, runoff, and groundwater, and water leaving the system via evaporation (Hall *et al.* 2023). When this balance is negative, lake levels decrease and salinity levels increasing, which can have an enormous negative impact on these ecosystems (Larson *et al.* 2016, Wurtsbaugh *et al.* 2017, Baxter & Butler 2020). Generally, when saline lakes experience increased aridity, ephemeral saline lakes stay dry for longer and permanent saline lakes become smaller and more saline (Williams 2002, Wurtsbaugh *et al.* 2017). These impacts are already being observed in the Great Basin region of the U.S., between 1984-2018 surface water declined at lakes and wetlands by 27% and 47%, respectively, in snowmelt watersheds (Donnelly *et al.* 2020).

Staging sites such as the Great Salt Lake (Utah), Lake Abert (Oregon), and Mono Lake (California) are vulnerable to changing climate because the southwestern United States it is expected to experience increased temperatures, increased evaporation, decreased precipitation, and more frequent and longer droughts in the 21st century (Seager *et al.* 2007, Hall *et al.* 2023, Snyder *et al.* 2019). These changes are expected to reduce the water supplies for cities, agriculture, and ecosystems alike, exacerbating the effects of water diversion from saline lake watersheds, and resulting in negative impact in water levels at saline lakes (Ficklin *et al.* 2013, Baxter & Butler 2020, Hall *et al.* 2023). As lake levels drop, increased salinity results in negative impacts on reproduction and survival of invertebrates eaten by phalaropes (especially brine shrimp, alkali flies, and brine flies; Dana & Lenz 1986, Herbst 2023), which in turn may negatively affect the ability of the phalaropes to molt and replenish fat reserves at these staging sites (Andrei *et al.* 2009).

In South America, climate-driven changes also threaten saline lagoon habitat used by phalaropes. The Andes has a complex precipitation regime influenced by local topography, with the main sources of precipitation coming from the east from the Amazon Basin (Neukom *et al.* 2015). The altiplano area used by Wilson's Phalaropes, located in the rain shadow of the Andes, is naturally an exceptionally dry region (Lupo *et al.* 2018). The region has experienced an increasing trend in temperature during the 20th and 21st centuries that is projected to continue (Vuille et la. 2015, Urrutia & Vuille, 2009). Precipitation is predicted to decrease by up to 33% in the central Andes by 2071-2100 driven by increased westerly winds (Neukom *et al.* 2015). However, predicting precipitation in the region is complex (Urrutia & Vuille, 2009), with some regions and model scenarios predicting potential increases in precipitation (Blin *et al.* 2022). 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, Torres-Batlló *et al.* 2020). The same areas experienced increasing evapotranspiration over the same period (Torres-Batlló *et al.* 2020). In other parts of the Andes, aridification is already being observed: water levels in altiplano wetlands are at a 600-year low, based on a tree ring study (Morales *et al.* 2015), and southwestern South America experienced a long-term drying trend from the early 20th century to the early 21st century (Morales *et al.* 2012, Rivera *et al.* 2020). In Central Chile (latitude 30ºS, including the southern edge of where Wil-


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son's Phalaropes occur in Chile), 2010-2019 was the longest sequence of drought years in the historical record (since 1918), and this period was considered a "mega-drought" (Garreaud *et al.* 2020). This drying is expected to continue in the future because of anthropogenic climate forcing (Garreaud *et al.* 2020). Overall, the altiplano is expected to continue to experience warmer temperatures (Vuille *et al.* 2015, Minvielle & Garreaud 2011), increased evaporation rates (Torres-Batlló *et al.* 2020), and regional decreases in precipitation (Neukom *et al.* 2015). This, combined with the impacts of water extraction for lithium mining, is likely to result in continued shrinking of wetlands in the region that phalaropes rely on.

On the species' breeding grounds, the observed and predicted impacts of climate change are similar to that described above for other parts of the range - a hotter, drier climate is expected in the 21st century, resulting in loss of wetland habitat (Shepherd & McGinn 2003, Ballard *et al.* 2014). In a study modeling the "climate envelope" for the species, 46% of breeding range habitat was expected to be lost under a 3 degree warming scenario (Audubon 2023). In the same model, temperature increases of 1.5 and 2 degrees C resulted in increases in breeding habitat loss of 22% and 30%, respectively (Audubon 2023). That study predicted that virtually all breeding habitat in the Great Basin and Great Plains regions of the United States would be lost as suitable breeding habitat, but also stable habitat in the Canadian plains and habitat gains in northern Canada, presumably as boreal forest is converted to grasslands (Audubon 2023).

In addition to drought and aridification, climate change may impact Wilson's Phalaropes via sea level rise, changes in wind patterns, or indirect impacts such as the expansion of non-native species. Estimates of increased sea levels (resulting from thermal expansion of ocean water and melting of landfast ice) indicate they may rise 1–2 meters or more by 2100 (Rahmstorf 2007, Sriver *et al.* 2012). Such sea-level rises could eliminate many coastal areas used by the species, of particular concern being the coastal lagoons in western South America (e.g., Ecuasal in Ecuador) where the species arrives after its transoceanic flight.

Migrating Wilson's Phalarope are presumably dependent on favorable winds and weather patterns to complete their long transoceanic flights. Warming ocean temperatures could change wind and weather patterns, thus disrupting migration (Gill *et al.* 2005). An increase in the number and severity of storms, both during migration and while at staging sites, could also have negative consequences for the species (Piersma & Lindstöm 2004).
Climate change could also intensify the expansion of non-native vegetation (see 6. Invasive and problematic species), because opportunistic exotic species are well-suited to take advantage of the ecosystem disturbances caused by warming temperatures.

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Conservation needs

The conservation of species that depend on diverse sites throughout an extensive geographic region, as is the case of Wilson's Phalarope, is challenging not only from an ecological point of view, but also from a social, political, economic, and cultural.

According to what was discussed in the previous section, we can identify numerous threats that the species faces throughout its distribution in the American continent, all of which must be considered when planning and implementing conservation and research actions. However, some of these threats stand out as likely to have dramatic and immediate effect on the species globally, with major direct negative consequences on the survival of Wilson's Phalarope. The aim of this section is to identify the most urgent threats identified for the species and describe the management, conservation, and research priorities necessary to reverse or mitigate them.

Priority actions and strategies:

1. Secure the inflow of water to important sites.

Currently, the main threat for Wilson's Phalarope is the loss and degradation of hypersaline wetlands on which it depends during its non-breeding season. This includes both migratory stopover sites in the western United States, some of which have recently reached the level of ecosystem crisis as water levels decrease, and hypersaline lakes in South America, which are especially threatened by water use for mining. See background about these issues in the introduction to the Threats section (p. 27) and the Energy production and mining section (p. 33).

The priority actions identified and recommended to address the problems related to water inflow in saline lakes are:

- Engagement on water policy for key sites, to ensure adequate inflow of fresh water, especially at Great Salt Lake, Lake Abert, Mono Lake, and Laguna Mar Chiquita. The most lasting solution is to obtaining permanent legal water rights for these sites. Water rights for wetlands have been obtained in the past through both litigation (e.g., Mono Lake) and direct purchase of water rights (e.g., Walker Lake, Nevada). Changing legal impediments to water delivery is sometimes needed as well, such as a law passed in 2022 that allows water flowing to Great Salt Lake to be a "beneficial use" of water in the state of Utah (Miller 2022).

- Identify potential interventions at sites where water could be more effectively managed to maintain habitat for phalaropes, e.g. through within-site movement of water to maintain it at appropriate water levels or salinities. Sites where water is already highly managed, with potential for management to better benefit phalaropes, include Chaplin Lake, San Francisco Bay, Owens Lake, Great Salt Lake, and Ecuasal. For example, at Great Salt Lake management actions to control salt and water exchange between the North and South Arms of the lake has shown promise for regulating salinity levels in the South Arm to maintain a healthy ecosystem (Utah DNR 2023). Short-term management and engineering solutions should be carefully balanced with goals of more holistic ecosystem restoration, such as securing permanent water rights for lakes.

- Evaluation of the status of water inflow and water budgets for all globally important sites. Better evaluation of the threat of water diversion is especially needed for South American sites. Establish minimum inflows of water needed to sustain key sites.

- Monitoring of lake elevation and salinity data at key sites, critical to understanding climate impacts and water budgets.
- Engagement with the mining sector, government agencies, non-governmental organization, and local communities to better evaluate the status of aquifers and water inflow at sites, especially in High Andean lakes.

- In areas with managed water extraction, establishment of best practices to protect sites for shorebird values.

- Consider interventions at specific sites through engineering and management projects aimed at maintaining habitat with appropriate water levels and salinity on smaller scales (i.e., dikes and water control structures).

2. Identify and protect critical habitat and landscapes throughout the species range:

The population of Wilson's Phalaropes can only be sustained if all key sites for the species are conserved, and their habitats maintained. Working at the site level is key; actions at the local level have an impact at the global level. Although all the threats listed in the previous section affect the species, it is clear that each site is subject to different pressures, regulations, sociopolitical and economic environments, among other factors.

Identifying those priority sites/landscapes for Wilson's Phalarope will allow us to focus appropriate research, management, and conservation efforts on each of them. It will also increase the possibilities of collaboration between partners, global visibility, and inclusion in projects for funding actions on the ground, which will have global benefits for the species.
The priority actions identified and recommended to identify and protect critical habitat and landscapes are:

- Designate sites of global importance for Wilson's Phalarope (those holding 1% or more of the global population) as Important Bird and Biodiversity Areas (IBAs), Key Biodiversity Areas (KBAs) and/or WHSRN sites.

- Promote the permanent protection of key conservation areas (or portions thereof) through government or private procurement of lands or long-term landowner commitment.

- Identify places and opportunities for active management to maintain required habitats and food resources. Clearly establish highest-priority sites for conservation action through a participatory process combining the importance for Wilson's Phalarope (and other species) with urgency (level of threat). Identify site-based priority actions therein.

- Complete site conservation plans for the highest-priority sites for conservation action for Wilson's Phalarope.

3. Understand global population status and trends, and habitat use through monitoring and research.

Knowing the status of the global population of Wilson's Phalarope and its variations at the global, regional and local scale is the basis for the prioritization of conservation actions and for the identification of key sites in which to develop all the actions mentioned in this section. Understanding population status is also critical to funding initiatives for Wilson's Phalarope conservation, and informed prioritization of the species relative to the many other threatened shorebird species globally.
The priority actions identified and recommended to address this are:

- Systematic coordinated surveys at major molt staging sites in North America;

these should include at least Great Salt Lake, Mono Lake, Lake Abert, Chaplin Lake (Saskatchewan), San Francisco Bay (California), and Owens Lake (California). Inclusion of more sites would be ideal.

- Systematic annual coordinated surveys at a subset of main key sites for the species within their non-breeding season in the altiplano of northern Argentina and Chile, Bolivia and southern Peru, and the Southern Cone lowlands (i.e., Chaco, Pampas, and the Patagonian Steppe) in Argentina, Bolivia, Chile, and Paraguay. Identification and prioritization of which South American sites should be monitored annually is also needed.

- Simultaneous and periodic surveys at important sites for the species within their non-breeding range in South America. There is the opportunity to collaborate with the GCFA (Conservation Group of Andean Flamingos by their acronym in Spanish) during their 5-year simultaneous survey for flamingos in High Andean Wetlands, which surveys the same sites used by phalaropes.

- A systematic survey of coastal areas in western Ecuador and Peru to better understand key sites and habitats used by arriving migrants.


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- Surveys of potential key sites in northern South America, Central America, and Mexico during the northbound migration.

- Tracking studies (i.e., radio or satellite telemetry, light-level geolocation) for better understanding of migratory connectivity, residence times at key sites and movement among sites at landscape scale, both in North America breeding and staging and South America non-breeding sites.

4. Articulate efforts on conservation, research, and monitoring.

The different conservation, monitoring and research actions mentioned above will be enriched if there is coordination among partners within and across the different important sites for the species. Keeping abreast of what is happening in different places throughout the hemisphere is challenging, but also opens to new opportunities for collaboration on data collection, optimization of resources, funding, community engagement, education, sharing of lessons learned, and more. That is why working on the growth and development of spaces for interaction, cooperation and consultation will result in direct benefits for the conservation of the species and their sites.

The priority actions identified and recommended to articulate efforts on conservation, research, and monitoring are:

- Continuation of the work on the International Phalarope Working Group (established 2019) including continued annual or bi-annual meetings and capacity building of members.

- Work with relevant parties to ensure that Wilson's Phalarope habitat needs are being addressed within the following conservation initiatives and frameworks: Habitat Joint Ventures in the U.S. and Canada, National Shorebird Conservation Plans in Argentina, Ecuador and Peru, and in the creation of a National Shorebird Conservation Plan in Chile. Also, the Chilean National Strategy for Birds, the Ramsar Convention's High Andean Wetland Conservation Strategy, and the Midcontinental and Pacific Shorebird Conservation Initiatives.

- An interchange program in which researchers, conservationists, and/or managers from different sites can travel to other sites to connect, collaborate, and learn from each other. Emphasis for a program such as this should be on supporting expenses of researchers from Latin America to participate in interchanges.

Other actions considered important for the species

We consider the priorities described above as the highest priority for the conservation of the species. Below we provide a list of more detailed actions, many of which are connected to the broader priorities described above.

Conservation:

- Update the Wilson's Phalarope's IUCN status and national-level conservation statuses with current information.
- Assess threats at key sites and prioritization of site protection, including quantifying threats to important non-breeding sites (e.g., the impact of mining activities) and desiccation from water extraction and climate change throughout their range.
- Within protected areas of global or regional importance for Wilson's Phalarope, identify all conservation actions required to maintain or increase the species' populations.
- Evaluate the impact of recreational activities in important sites for the species.
- Assess and document the protection status (regional, national, international, voluntary) for all sites of global importance for Wilson's Phalarope.

Monitoring and research:

These actions are organized by where each would occur in the annual cycle of Wilson's Phalarope, whether the entire range, the breeding grounds, the North American staging sties, or the South American non-breeding range.

Entire range:

- Update the world population estimate with contemporary data from both the South American non-breeding range and North American migratory staging sites.
- Refine models to explore the likely effects of climate change on breeding, staging, and non-breeding habitats.
- Quantify the exposure to and impacts of agrochemicals, heavy metals, and other contaminants.
- Determine if Wilson´s Phalarope shows site fidelity to North American staging sites, or non-breeding sites and stopover areas in South America. Genetic techniques may be useful for this question.
- Model the population trajectory (e.g., Population Viability Analysis), with a focus on the expected impact on population from loss or degradation of wetlands from climate change.
- Explore genetic techniques for determining effective population size, population trends, and genetic diversity/structure.

Breeding grounds:

- Evaluate the extent to which agricultural land uses (i.e., habitat conversion and water management) alter the ecology of wetland breeding and migration habitats, and the impacts on the species.

- Evaluate the impact of nest predation and methods for reducing nest predation and increasing reproductive output.

- Investigate the impact of the introduction of exotic species, i.e., how Wilson's Phalarope responds to the expansion of European cheatgrass (Bromus tectorum) on breeding grounds.

North American staging sites:

- Compile historic data from major staging sites (e.g., Great Salt Lake, Mono Lake, and Lake Abert) to develop time-series of data collected with consistent methods and coverage to evaluate population trends.

- Further test of the relationship detected by Jehl (1999) between availability of breeding habitat and number of birds at staging sites in the subsequent year. This should include a refinement to take into consideration the actual extent of habitat availability (and not just an indication based on the number of ponds).

- Annual coordinated surveys at the largest staging sites during June-August.

South American non-breeding range:

- Compile historic data from major non-breeding sites (e.g., Laguna Mar Chiquita, Argentina) to develop time-series of data collected with consistent methods and coverage to evaluate population trends.

- Quantification of the importance of non-breeding areas and habitats other than the altiplano saline lakes (i.e., the Southern Cone lowlands, including Patagonia).

- At non-breeding sites of known importance, regular surveys of phalarope numbers and monitoring of habitat metrics, especially water levels, are needed. Ideally, at priority sites, surveys should be conducted monthly during November-March. These types of monitoring are needed to better assess variability in site use, assess threats, and feed into future management plans and conservation designations of sites. Monthly surveys from January-March of the most important non-breeding sites. If monthly surveys are unsustainable, these sites should ideally be surveyed at least once annually during expected peak-timing of Wilson's Phalaropes.


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Morphology

With a length of 22–24 centimeters (9.25 inches), Wilson's Phalarope is the largest of the three phalarope's species (Colwell and Jehl 2020). It is separable from the two other phalaropes [Red Phalarope (P. fulicarius) and Red-necked Phalarope (P. lobatus)] by its longer and thinner bill, proportionately longer neck and legs, lack of a white wing-stripe in flight (Figure A1) and a completely white rump (Blake 1977, Paulson 2005). Phalaropes are highly aquatic shorebirds with lobed toes that enable them to swim.

Figure A1: Female in flight showing the lack of white wing-stripe. ©Steve Mlodinow

Wilson's Phalarope is, however, the most terrestrial of the three species and has less developed lobes (Hayman *et al.* 1986). Unlike most shorebird species (but as with other phalarope species), *P. tricolor* shows highly noticeable reversed sexual dimorphism, with females being larger and more brightly colored (in the breeding season) than males. Average measurements (in grams and millimeters) of birds from Mono Lake, California (Jehl 1988) are shown in the following table:

Metric	Culmen			Tarsus			Wing			Mass		
	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD	n
Adult Male	30.1	1.2	319	31.6	1.3	317	121.2	2.8	318	56.9	10.0	345
Adult Female	33.1	1.3	100	33.2	1.2	101	131.7	3.3	101	75.6	14.6	138
Juvenile Male	29.6	1.2	29	31.7	1.0	28	119.5	3.0	28	49.6	5.2	33
Juvenile Female	32.8	1.3	21	33.6	1.5	21	129.4	2.8	21	60.2	7.6	23

Plumage

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 (Figure A2). This band is chestnut red on the sides of the mantle, becoming an orange-pink wash on its breast and strongest 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; 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 (Figure A2)

Figure A2: Female (left) and male (right) in breeding plumage. ©Steve Mlodinow

The nonbreeding plumage (Figure A3) 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 (a feature not shared by the other species of phalarope).

Figure A3: non-breeding plumage of Wilson's Phalarope. ©Nino Grangetto

Figure A4: juveniles with remaining downy feathers. ©Steve Mlodinov

Juvenal plumage is dark brownish on the upperparts with broad buff fringes on the feathers, giving a scaly appearance (Figure A4). The breast sides are washed with buff, while the rest of the underparts are white, and the legs yellowish. First-year birds retain the buff fringes to the inner median coverts until the definitive pre-alternate molt (Feb through May) (Prater *et al.* 1997).

Molt

Wilson's Phalarope is one of the few shorebird species that is known to have a molt migration, flying from breeding grounds to staging areas where they undergo an almost complete molt in as little as 32–40 days (Jehl 1987, 1988). Although males arrive later than females at the staging areas, they start molting the head and body feathers on the breeding grounds while caring for the chicks (Jehl 1988, Colwell and Jehl 1994). However, the majority of the male's plumage is also replaced on the staging grounds (Jehl 1988). In both sexes, flight-feather molt begins approximately one week after the body molt and is suspended around mid-July or August for migration to the non-breeding grounds, where molt is completed on arrival (Burger and Howe 1975, Jehl 1987, 1988). Molt in juveniles is far less intensive and is mostly completed on the non-breeding grounds (Burger and Howe 1975, Jehl 1988).

Taxonomy

The three phalarope's species—Red Phalarope (*Phalaropus fulicarius*), Red-necked Phalarope (*P. lobatus*), and Wilson's Phalarope (*P. tricolor*)—were once placed in their own family, Phalaropodidae, but genetic studies have since shown them to be nestled within Scolopacidae (e.g. Ericson *et al.* 2003). Wilson's Phalarope is a monotypic species, either recognized within the genus Phalaropus or placed in its own genus, Steganopus. In this document we follow the American Ornithologists' Union (1998) in using Phalaropus, although other authors (e.g., van Gils and Wiersma 1996) prefer Steganopus in recognition of the genetic distance between Wilson's Phalarope and the other two phalarope species. In the historical literature, Wilson's Phalarope has been referred to as *Phalaropus wilsoni*(i), *P. stenodactylus*, *P. frenatus* (Hellmayr and Conover, 1948) or even confusingly as *P. lobatus* (Smith 1889). The degree of relatedness between the three phalarope's species has yet to be fully resolved. Nonetheless, mtDNA data analyzed by Dittmann and Zink (1991) suggest that the Red Phalarope (*P. fulicarius*) and Red-necked Phalarope (*P.lobatus*) are sister species and that Wilson's Phalarope is more distantly related. In fact, the species has been considered to be the more primitive of the three (Jehl 1968). Dittmann and Zink (1991) also considered it more likely that the three phalaropes are monophyletic, as opposed to Wilson's Phalarope being a case of convergence.

Although "Wilson's Phalarope" is the species' only common name in English, it is known by a variety of common names in other languages throughout its range in Central and South America, including: Faláropo de Wilson (Mexico and Central America), Falaropo Piquilargo (Mexico), Falaropo Tricolor (Ecuador, Paraguay, Colombia), Pollito de Mar Tricolor (Chile), Falaropo Común, Chorlito Palmado Grande, Chorlo nadador (Argentina), Zarapico de Wilson (Cuba), Chorlillo Piquilargo (Mexico, Belize, Guatemala and El Salvador), Falaropo Pico Largo (Honduras), and Pisa N´Agua (Brazil).

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We extend our apologies to anyone accidentally omitted from the above list – please let us know!