Natural and Social Sciences of Patagonia

Gabriela Mataloni
Rubén D. Quintana
*Editors*

# Freshwaters and Wetlands of Patagonia

## Ecosystems and Socioecological Aspects

Springer

# Natural and Social Sciences of Patagonia

Despite being an underpopulated region, Patagonia has attracted the attention of scientists since the very beginning of its settlement. From classical explorers such as Darwin or D´Orbigny, to modern science including nuclear and satellite developments, several disciplines have focused their efforts on unraveling Patagonia's natural and social history. Today, scientific and technological research is shifting from being shaped by northern agendas, towards more locally oriented objectives, such as the management of natural resources, the modernization of energy production and distribution, and the coexistence of rural and cosmopolitan social lifestyles. At the intersection of all these topics, new conflicts concerning the economy, human development, population, and the proper and long-standing planification and management of the landscape and its natural resources have emerged. These conflicts, of course, have also caught the attention of many interdisciplinary research groups.

This series is aimed at describing and discussing various aspects of this complex reality, but also at bridging the gaps between the scientific community and governments, policymakers, and society in general. The respective volumes will analyze and synthesize our knowledge of Patagonian biodiversity at different scales, from alleles, genes and species, to ecosystems and the biosphere, including its multilevel interactions. As humans cannot be viewed as being separate from biodiversity, the series' volumes will also share anthropological, archaeological, sociological and historical views of humanity, and highlight the wide range of benefits that ecosystems provide to humanity including provisioning, regulating and cultural services.

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# Foreword

It is a pleasure for me to introduce the book *Freshwaters and Wetlands of Patagonia: Ecosystems and Socioecological Aspects* edited by Drs Gabriela Mataloni and Rubén Quintana. The contents of this carefully edited book will be of great interest to many actors working on the conservation and management of these important ecosystems in such an iconic region of Argentina as Patagonia. Nowadays, freshwaters and wetlands are critical ecosystems for humanity all around the world, not only due to their contributions to people but also because of the impacts on them derived from climate change and the change and intensification of land uses. Patagonia, in particular, is a vast region wherein freshwater resources are not abundant. In fact, the greatest part of it is constituted by a cold desert wherein water will be a limiting resource in the coming decades, affecting both human settlements and productive activities. For these reasons, this book is a fundamental work to understand not only the diversity of freshwater and wetlands ecosystems present in Patagonia but also the threats they face. Management of these systems requires in-depth understanding of their intricate structural and functional aspects, with inputs from sociological, legal, institutional and economic aspects in addition to disciplines such as biology, ecology and hydrology. The book contributes to integrating information and visions from different disciplines. The chapters were written by renowned specialists who contributed their expertise in a way accessible to a wide audience. The planning of this book is highly original in that the opening chapter presents an introduction to the Patagonian region and its water-dependent ecosystems, whilst the last one discusses how to reach the UNDP Sustainable Development Goals in this unique region. Yet, in the 18 chapters, a vast number of concepts, experiences and case studies are intertwined by cross-references in an exemplary editorial labour. This book is in line with the efforts and initiatives of the Ministry of Environment and Sustainable Development of Argentina towards conserving freshwater and wetlands, particularly in the context of the Wetlands National Program. I therefore congratulate the editors and the publisher for bringing out this

book of topical significance, which I am certain will serve as a vital reference for researchers as well as freshwater and wetland site managers.
Directora Nacional de Gestión Ambiental del Agua y los Ecosistemas Acuáticos
Ministerio de Ambiente y Desarrollo Sostenible de la Nación,

Buenos Aires, Argentina
April 2022

Gabriela González Trilla

# Preface

Commonplace as it may sound, there is a point in explaining to the reader *why this book* in particular. Originally, we were kindly invited by the editors of the Natural and Social Sciences of Patagonia series to edit a volume that would synthesise current limnological knowledge on this region. In so attempting, we found it impossible to present a comprehensive view of the present state of Patagonian freshwaters without considering the multiple aspects encompassed in the study of water, as it is the core of many economical activities and object of social conflicts, which are also relevant to this series. We also felt that the two main types of water-dependent ecosystems – aquatic ones and wetlands – have a number of common ecological features and face threats alike, while also standing on their own in many other aspects, and thus call for particular solutions. This is the reason for the *Freshwaters and Wetlands of Patagonia* title.

In facing the challenge of accounting for the present state and future prospects of these ecosystems, we embraced a socioecological approach not only to understand their actual condition but also to summarise their many contributions to people, and discuss the conflicting interests around them that can drastically change their evolution. This is the reason for the *Ecosystems and Socioecological Aspects* subtitle.

Contributing authors are well-recognised experts from across the extensive Patagonian geography, both from Argentina and Chile, and devoted to disciplines from hydrology to microbiology to political ecology, among many others, who, through the different chapters, review recent, scientifically rigorous literature as well as pieces of their own research, but also interact to produce environmentally sound information. Moreover, the book as a whole encourages inter- and transdisciplinary dialogue through multiple cross-references, with an aim at bridging conceptual gaps between natural and social sciences. As such, our ultimate challenge is to inform a broad audience of students, scientists, engineers, environmental managers and policymakers, that is, any politically and professionally relevant actor around water, from an overarching perspective in order to contribute to the integration of efforts towards the conservation and sustainable management of these ecosystems.

The book is organized in two parts. Part I, Ecosystem Patterns and Processes, includes eight chapters and offers an up-to-date overview of the status of Patagonian glaciers, discusses the nature of freshwater and wetland ecosystems of this region, and depicts both local and large-scale biodiversity patterns and their potential use as sentinels of environmental changes at different scales. It also characterises the impact on the community structure of extreme events, such as volcanic eruptions, glacier lake outburst floods and mercury natural pollution, and of extreme environments like a hyperacidic river. Part II, Socioecological Aspects of Freshwater Ecosystems in Patagonia, comprises ten chapters. These deal with hydrologic systems and wetlands, and the pressure on these water resources resulting from changes in land use within a climate change context, with a focus on emerging conflicts around freshwater availability. The effects of fisheries and fish species introduction are also discussed, as well as different examples of conservation policies, from dealing with an invasive diatom to salvaging populations of endangered endemic amphibians and waterbirds depending on such ecosystems and presently endangered by human activities. Some sociological and governance aspects show different cultural valuations of water, from ancestral Mapuche rural communities to modern legislation balancing social, environmental and economic interests around it. Finally, the obstacles for meeting the water-dependent Sustainable Development Goals of the UN 2030 Agenda for Sustainable Development are discussed, while actions towards the adoption of integrative water management measures are envisaged.

Again, we are most grateful to the editors of this series for their obliging invitation, and to all authors whose excellent contributions enriched this book as much as our own view of Patagonia. Many thanks are due to Joao Pilderwasser at Springer Nature, who was always there for us, and to all chapter reviewers, whose insightful comments greatly contributed to the quality of our work. They are Adonis Georgi, Alejandra Volpedo, Alejandro Giraudo, Ana Ladio, Anahí Magdaleno, Bart Van de Vijver, Donaldo Bran, Eleonora Carol, Elizabeth Wencke, Enrique Lara, Fernando Unrein, Jean-Pierre Descy, Juan Luis García, Luz Allende, Magdalena Licursi, María Azpelicueta, María José Viñals, Mariana Schmidt, Marilyn Cebolla, Mariusz Lamentowicz, Marta Litter, Maximiliano Attademo, Melina Devercelli, Melisa Olivelli, Nora Gómez, Patricio Macchi and Víctor Cussac. And finally, a special thanks to Gabriela González Trilla for her kind foreword. We hope this endeavour will contribute not only to the scientific knowledge of Patagonian aquatic and wetlands ecosystems but also to foster the integration of such knowledge in governance practices aimed at an environmentally just Patagonia.

Buenos Aires, Argentina
April 2022

Gabriela Mataloni
Rubén D. Quintana

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# Part I
# Ecosystem Patterns and Processes

# Chapter 1
# Navigating the Freshwaters of Patagonia… and This Book

**Gabriela Mataloni and Rubén D. Quintana**

## 1 The Many Ways to Look at Patagonia

Patagonia is the southernmost geographical region outside Antarctica, which comprises the southern tip of continental South America and the surrounding archipelagos. Its main geographic feature, the Andes Mountain Range, extending in the NS direction, roughly sets the limit between the republics of Argentina and Chile, which encompass around 88% and 12% of the Patagonian land, respectively (Peri et al. 2021).

From a geological viewpoint, Patagonia constituted an independent terrane whose convergence with western Gondwana originated the Huincul High, a deformation belt running WE which represents its northern natural limit (Ramos et al. 2004; Mosquera et al. 2011) (Fig. 1.1). Its geological structure is composed of two distinct subregions: the mountainous Andean Patagonia, mainly resulting from the Tertiary folding, and the Extra-Andean plateau, resulting from the Mesozoic and Cenozoic filling of the ancient basement. Within these two large units, a number of geological provinces can be recognised (Coronato et al. 2017).

The location of Patagonia stands as the only piece of continental land south of the 40º S, and its climate is therefore signed by the Andes ridge acting as a mighty barrier against the powerful westerlies from the South Pacific anticyclone. This creates a strong rain shadow effect, with distance from the Andes accounting almost on its own for the eastward decreasing trend of the annual precipitation in mid-Patagonia (Jobággy et al. 1995). In the Tierra del Fuego Island, the Andes change their orientation towards WE, thus diminishing the rain shadow over the eastern

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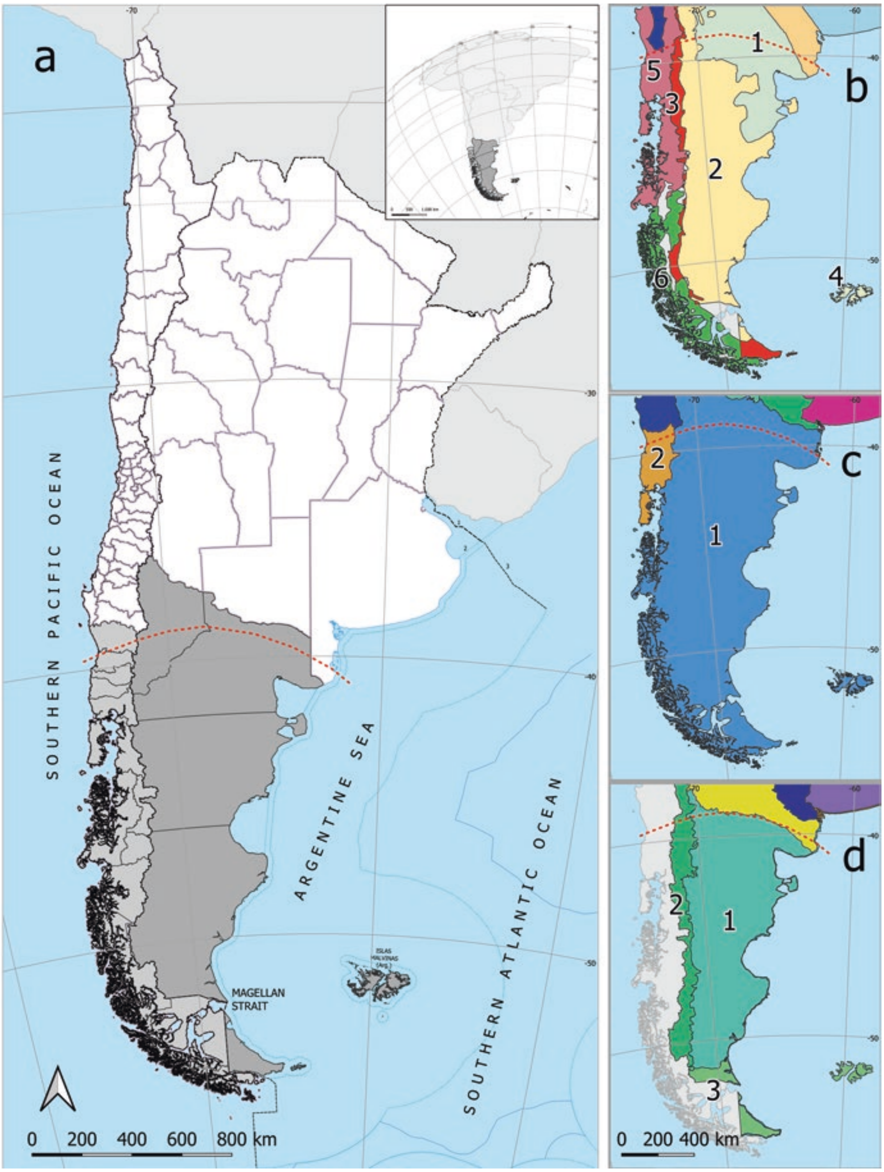

**Fig. 1.1** Consistent regionalisation criteria of Patagonia. (**a**) Geographical division of Argentina and Chile, showing the provinces belonging to Patagonia (dark and light shadow, respectively). The dashed red line shows the position of the Huincul High according to Mosquera et al. (2011). (**b**) Terrestrial ecoregions of Patagonia, according to Morello et al. (2018) in Argentina and World Wildlife Fund (2022) in Chile. 1: Monte; 2: Patagonian steppe; 3: Patagonian forests; 4: Malvinas Islands; 5: Valdivian temperate rainforest; 6: Magellan subpolar forest. (**c**) Freshwater ecoregions according to Abell et al. (2008) 1: Patagonia; 2: Valdivian lakes. (**d**) Wetlands regions of Argentina according to Benzaquén et al. (2017) 1: lakes, watercourses and meadows of Patagonian Andes; 2: shallow lakes and meadows of the Extra-Andean Patagonia; 3: meadows and peat bogs of southern Patagonia and South Atlantic Islands

section of the island. The ample latitudinal range of Patagonia (ca. 20 °C) determines a strong decrease in mean annual temperatures from 12 °C in the NW to 3 °C in the S (Peri et al. 2021). Interestingly, the decreasing width of the land mass with latitude also determines a decreasing thermal range from 16 °C in the Patagonian plateau to 4 °C in the southernmost magellanic islands according to Coronato et al. (2017). These authors also give a detailed classification of the 13 identified Patagonian climatic types (Coronato et al. 2017).

Beyond geoclimatic features, Patagonia can be comprehensively regarded from an ecoregional viewpoint. To this end, Morello et al. (2018) devise an ecoregion as a geographically defined territory with a common geological and biogeographical history and a distinct regional climate based on annual mean precipitation and the presence and extent of a dry season and a cold season, characterised by homogeneous ecological responses to climate and tectonics, expressed by the vegetation, the fauna, the relief and anthropic activities such as agriculture and industry. These authors thoroughly described the whole territory of Argentina in terms of ecoregions, sub-regions and ecosystem complexes. According to them, three main ecoregions are present in Patagonia, following the formerly described geophysical and climatic patterns: from NE to SW, the *Monte* (Monte desert), *Estepa patagónica* (Patagonian steppe) and *Bosques Patagónicos* (Patagonian forests) each of them comprising different subregions. The latter include two separated N and S High Andean subregions that hold a continuity with the southern Andean Steppe Ecoregion in Chile, as defined by the World Wildlife Fund (World Wildlife Fund 2022). Beyond the Andes, the Valdivian temperate rainforest to the north and the Magellanic subpolar forest to the south spread towards the Pacific Ocean (Morrone 2001). In turn, Abell et al. (2008) also used an ecoregion-based approach for the first biogeographic regionalisation of freshwaters in order to identify global and regional conservation priorities of fish species. Amongst the 426 ecoregions so defined, they identify a Patagonia ecoregion, which roughly coincides with the one geologically delimited here, but excluding the Valdivian Lakes ecoregion, which includes the lake district of southern Chile and the island of Chiloé.

In Argentina, since the beginning of the century, a National Wetlands Inventory has progressed, with the aim of informing adequate conservation and sustainable use policies. The first spatial level of this inventory defines wetland regions and subregions strongly based on a hydrogeomorphic approach which takes into account environmental features driving the presence, spatial expression and ecological features of wetlands (Benzaquén et al. 2017). In this context, Patagonia represents a whole region, divided into three wetlands subregions: The lakes, watercourses and *mallines* (a meadow type) of Patagonian Andes; the shallow lakes and *vegas* (another meadow type) of the Extra-Andean Patagonia; and the *mallines* and *turberas* (peat bogs) of southern Patagonia and South Atlantic Islands. Unlike Argentina, Chile has not identified wetland regions and subregions. Anyway, it is recognised that most wetland areas are located in the Chilean Patagonia, particularly in the Aysén (XI) and Magallanes y la Antártica Chilena (XII) Regions (Promis 2010).

In sum, the preceding paragraphs illuminate a number of possible ways to regard the complex aspects of Patagonia through the use of complementary ecoregional approaches to terrestrial, overall freshwaters or particularly wetland ecosystems. Notwithstanding, geopolitical boundaries are most commonly used for administrative purposes. Political organisation of Chile recognises 16 regions, with those of La Araucanía, Los Ríos, Los Lagos, Aysén y Magallanes y la Antártica Chilena being included in Chilean Patagonia; while in Argentina it comprises the provinces of Neuquén, Río Negro, Chubut, Santa Cruz and Tierra del Fuego, Antártida e Islas del Atlántico Sur (AIAS). Nevertheless, as geopolitical limits of both countries encompass sovereignty claims on Antarctica suspended in accordance with the Antarctic Treaty (Secretaría del Tratado Antártico 2021), such areas are not included in the scope of this book.

## 2 The Many Ways to Read This Book

Due to its complex history and geographical setting, Patagonian freshwaters abound in many different forms, from small, high altitude proglacial lakes to large deep lakes of glacial origin, to shallow lakes, mountain streams and different types of wetlands in the Andean zone, to large rivers, complexes of shallow lakes and ponds and many different wetland types, particularly meadows all over the steppe and peat bogs in the southernmost Tierra del Fuego Island, and are key landscape elements on which human activities depend. Therefore, the order in which the chapters of this book are presented is not unique, and a number of different Ariadne's threads can guide the reader along thematically related chapters.

### *2.1 The Water, the Ecosystems and the Communities*

Undoubtedly, in the last decades, for public opinion, wetlands have gone from being discarded lands to very valuable ecosystems that humanity should protect and use in a sustainable way. In any case, even today, there is much discussion about what wetlands really are. This debate has not only taken place in the media but also within the scientific community. Different viewpoints, particularly between limnologists and wetland ecologists, have given rise to hard discussions. One of the main debates regarding what a wetland is relates to its genesis since these ecosystems are neither aquatic nor terrestrial. In Chap. 2, we debate the ecological character of wetlands, comparing different definitions with constructive criticism. In addition, we describe the Brinson's hydrogeomorphic approach for wetland classification and how it was applied for the Argentinean wetland regionalisation. A general description of Patagonian wetlands is also included as well as some considerations on the wetland–society relationships and the contributions of wetlands, particularly Patagonian ones, to people. A final reflection about the Argentinean Wetland Law Project is included as an example of the difficulties in the way to get a better protection and sustainable use of these ecosystems.

The existence of different environmental (altitudinal, latitudinal, trophic) gradients in Patagonian freshwater bodies gives rise to distinct biogeographical patterns at different scales, as Schiaffino and Izaguirre extensively review in Chap. 5. These authors describe patterns of planktonic prokaryotes and eukaryotes along a gradient of Patagonian water bodies, which also includes some Antarctic lakes and highlight the role of spatial and environmental factors in controlling the bacterial community structure. They found a decrease towards higher latitudes of heterotrophic bacteria, Archaea and photosynthetic picoplankton abundances, while also remarking the importance of light conditions and trophic status of the lakes in shaping picoplankton structure. Phytoplankton diversity was also influenced at large spatial scale by geographical and environmental factors. Here, a decreasing biodiversity pattern with latitude was observed for particular taxonomic microalgal groups, as well as for the whole phytoplankton community, although local effects were stronger. In this review, the authors propose the co-existence of a 'core biosphere' containing a reduced number of dominant microeukaryote operational taxonomic units (OTUs) on which classical ecological rules apply, together with a much larger seedbank of rare OTUs driven by stochastic and reduced dispersal processes. They also present these findings as important tools for implementing land planning focused on conservation and sustainable use objectives at multiple scales.

Globally, peat bogs are much appreciated for acting as key sinks for atmospheric $CO_2$. In addition, these particular wetlands represent biodiversity hotspots, as they host a very distinctive acidophilic biota. In Chap. 6, Quiroga et al. take a closer look at the community structure of microalgae, ciliates and microinvertebrates in two peat bogs from Tierra del Fuego Island, which are encompassed in the largest and southernmost peatland complex in the Southern Hemisphere. The authors show that both elements of the landscape (lentic waterbodies and the surrounding *Sphagnum* moss matrix) account for the high, particular biodiversity of these systems, as they host contrasting communities building up different trophic webs. Also, taxonomic composition and community attributes are ultimately driven by interactions between environmental features sensitive to anthropogenic impacts, both direct – such as increases in nutrient concentrations, total hardness (TH) and conductivity potentially driven by housing development – and mediated by climate change, such as decreasing water table depth and increasing temperature. By means of general additive models, authors predict that changes in these parameters could have contrasting effects on the different planktonic communities. They also model their influence on the diversity and composition of microinvertebrates, covering both landscape elements at a regional scale, and pinpoint species highly indicative of environmental changes in all three communities. For instance, as easily identifiable *Bosmina chilensis* is at the same time an indicator of high microinvertebrate diversity and low TH, its decline could allow for the early detection of wastewater-mediated impacts. The synthesis of these approaches thus serves as a tool to monitor the impact of environmental changes on the fragile communities of these unique ecosystems.

Fish and fisheries distribution patterns in the Patagonian steppe are described in Chap. 13. This contribution comes to fill in a void of comprehensive information, because until now, most available knowledge on these topics pertains to the Andean sector. In this chapter, Baigún et al. describe a wide variety of the ichthyofauna characteristics, their assemblages and fisheries along the main basins of the Patagonian steppe. North Patagonia assemblages include species from the Patagonian, Andean Cuyean and Pampean ichthyological provinces with an overlapping of species from Austral and Brazilic subregions. Salmonid and anadromous species richness, on the contrary, increases from northern to southern latitudes. In addition, some endorheic basins are inhabited by species with restricted distribution ranges, including some with an extreme degree of endemism, which renders them paramount species from a conservation viewpoint. At present, several Patagonian fish species are threatened by human impacts, including climate change. In addition, historical and current management policies have permitted uncontrolled stocking of alien species due to their high fishing value, with the consequent reduction in the distribution and abundance of native fish species. The information included in this chapter thus constitutes a relevant source of information for fish and fisheries managers and decision makers.

## *2.2 The Many Influences of Volcanic Activity*

As pointed out earlier, the Andes are a key structuring agent of the climate and the geomorphology of Patagonia, but also encompass some areas of active and even intense volcanic activity, which can have particular effects on the waterbodies subjected to it. Such is the case of the Copahue Volcano, which hosts the source of the River Agrio-Lake Caviahue system. This system was extensively studied by Temporetti et al. (Chap. 7) for two decades. Volcanic activity confers the Upper Agrio River and Lake Caviahue very distinct chemical features, such as an extremely low (<2–3) pH and very high ion concentrations, although strong eruptive episodes over the last decade have altered the ionic balance of Lake Caviahue, resulting in the deposition of ferric compounds. Interestingly, the metals are deposited in the sediments in non-toxic forms. Yet the system can be regarded as an extreme one, wherein very few algal species can survive. In turn, the Lower Agrio River is progressively neutralised by the confluence of more typical oligotrophic mountain streams, resulting in a circumneutral pH and a well-developed epilithic community. Some uses of the upper system features have been envisaged, as two acidophilic microalgae resulted in good candidates to test soil pollution by polycyclic aromatic hydrocarbons (PAHs). Also, the authors found that Fenton decontamination processes are feasible under the particular Lake Caviahue conditions and can lay the foundation for the development of a treatment plant for the wastewaters from Caviahue town in the natural conditions of the lake.

Further south along the Andean Volcanic Belt, the Puyehue–Cordón Caulle Volcanic Complex exerts a strong influence on the deep lakes within the Nahuel Huapi National Park through a strong input to the mercury biogeochemical cycle. Indeed, according to Diéguez et al. (Chap. 8), total Hg concentrations measured in the waters of the Nahuel Huapi Lake after the last eruption of this volcanic complex are amongst the highest recorded in natural systems, wherein major sources from the lithosphere are volcanic and geological activity and volatilisation, notably from extensive forest fires. Anthropogenic Hg emissions from mining, coal combustion and industry, in turn, globally exceed natural emissions and are also reflected in lake sediment archives. As pointed out by the authors, the Hg biogeochemical cycle is complex, and Hg is mobilised under different forms, some of which are toxic. Such is the case of $CH_3Hg$, which can be produced by microbial methylation in freshwaters. In these systems, the concentration and molecular composition of the DOM pool determine Hg fractionation, availability and biotic uptake, as well as the net $Hg^{2+}$ methylation. Moreover, $CH_3Hg$ can concentrate along food chains. As authors demonstrate, in large, deep lakes with extended pelagic zones, both THg and $CH_3Hg$ concentrations in fish species vary by foraging habitat, increasing together with the proportion of benthic diet over pelagic diet. Yet, many features of the Hg cycling, such as compartment storages, fluxes, and chemical transformations, such as methylation are all climate-sensitive, and therefore the potentially profound consequences of climate change on the biogeochemical cycling of Hg are discussed.

Volcanic eruptions influence the physical and chemical properties and plankton communities of deep Andean lakes in yet other ways, as studied by Balseiro et al. (Chap. 3). Floating pumice and sestonic ashes from the last Puyehue–Cordón Caulle mega eruption have profoundly affected the light climate of ultraoligotrophic transparent lakes by decreasing the depth of the Deep Chlorophyll Maximum (DCM) wherein plankton communities find the best trade-off between avoiding excessive light radiation and P limitation. They also enhanced the food quality of phytoplankton by decreasing its C:P ratio. These effects are also generated by the presence of clay in the water column of lakes downstream from glacier or moraine-dammed lakes and can be dramatically changed by GLOFs (glacial lake outburst floods), which consist of the sudden release of such dams in the latter waterbodies due to glacial recession. Interestingly, these apparently alike events can have opposite consequences on zooplankton structure on account of their distinct nutritional requirements and feeding habits. In all, changes in DCM and the type of disturbance, including potential anthropogenic ones due to changes in land use, emerge as valuable clues to track and understand the behaviour of trophic webs in future scenarios.

## 2.3 Patagonian History and the Memory of Waters

First traces of human population in some reduced areas of Patagonia date back from 13,000 to 10,500 years BP. Before the European arrival, Patagonia was already occupied by different ethnic groups, mostly nomad hunter-collectors, amongst which the Mapuche were the most advanced and displaced other groups. Many Spanish (but also Portuguese, English and Dutch) expeditions and colonisation attempts failed over the sixteenth and seventeenth centuries (Matteuci 2012). During the nineteenth century and after their independence from Spain, Argentina and Chile competed over Patagonian land occupation. This was mostly aimed at sheep farming for wool trade with England and controlling the maritime transport of goods between Europe and Eastern Asia by connecting the Atlantic and Pacific oceans. In this context, both countries carried out expeditions with the objective of advancing the frontiers of Western civilisation by reducing or exterminating indigenous groups (Peri et al. 2021) under the deceitful names of *Campaña del Desierto* (desert campaign) in Argentina and *Pacificación de la Araucanía* (Araucanía pacification) in Chile. As a result of these events, most indigenous populations were annihilated, and small groups of survivors displaced to unproductive lands or absorbed as cheap labour force by large estate owners. In Argentina, five national territories were created, which later acquired their present provincial status and constituted modern Patagonia. By the end of the nineteenth century, large fractions of the so-acquired government-owned land were distributed by sale, as concessions or as payment for due services. Notably many of them were allocated to foreign companies, mostly British, and thus furthered the alienation of indigenous groups, in a process which still shapes economic activity and social relations in Patagonia: The main activities are sheep farming, oil and mineral extraction, and agriculture in the valleys along large rivers, which continue to displace local inhabitants and export most of the economic benefits (Matteucci 2012).

Presently, the National Institute of Indigenous affairs records 48 different originary people all over Argentina, amongst which the Mapuche are by far the most widespread, with a large number of communities recorded in northern Patagonia, mainly on the western side of Neuquén, Río Negro and Chubut provinces. Also relevant in Chubut and Santa Cruz are the Mapuche Tehuelche and Tehuelche communities, while only one Selk'nam and one Yagan communities remain in the Tierra del Fuego Island (Instituto Nacional de Asuntos Indígenas 2022). In a context of displacement of indigenous people to arid lands, traditional limnological knowledge is of key importance for community survival. In Chap. 16, Molares et al. apply a transdisciplinary cultural limnology approach to understand the reciprocity relationship between Mapuche people and the waterscapes they inhabit, wherein traditional water management practices allow them to perform family horticulture in a context of precarious basic services and low or no state economic or infrastructure assistance. The notion of biocultural memory about waters is key to sustain Mapuche views in the discussion of environmental policies regarding co-managed protected areas and in the claim for their right to land and water frequently lost to foreign occupation. Such a notion is one of community character, as it passes on from one generation to the next; of spatial character, as it depends on the intersubjective relationship with every specific body or source of water; and also dynamic, as it incorporates new information about perceived changes in the hydrological cycle. Yet, viewpoints rooted in traditional knowledge sometimes differ from those based on science. Here, the transdisciplinary field of cultural limnology aims at constructing a common language that is paramount for discussing inclusive, innovative ways of dealing with contemporary socio-ecological issues involving originary people.

## 2.4 Freshwaters and Wetlands in Present-Day Patagonia

According to the projection based on the last national census, population in Argentinean Patagonia reached 2,569,791 in 2020 (Argentina.gob.ar 2022) with a descending trend from Neuquén southwards and a minimum population density of 1.5 inhabitants $km^{-2}$ in the vast Santa Cruz Province. Population of Chilean Patagonia was 2,440,460 inhabitants as of 2017, with an average density of 7.18 inhabitants $km^{-2}$, steeply decreasing from North to South in the continental section. Notably, in both countries, the population rises again around the Magellan Strait and Tierra del Fuego Island. As pointed out before, the main economic activities in both countries as of late nineteenth century have been mining, sheep farming, agriculture (mainly crops and fruit production along northern rivers valleys), and oil and gas extraction. Over the second half of the twentieth century, several large hydroelectric dams were built along many rivers, while aeolic farms are gaining importance in taking advantage of the strong Patagonian winds (Peri et al. 2021). Although the luxuriant temperate forests and vast glacial lakes that fringe the Andes make magnificent landscapes much appreciated by international tourism, areas outside protection regimes gave way to pine afforestation. All these activities interact in complex ways that affect the natural geographical distribution and cycling of water in Patagonia, as extensively reported by Urciuolo and Iturraspe in Chap. 9. All main Patagonian hydrological basins are minutely described, including information on their sizes and location. Also, flows, yearly hydrological variations and characteristics of the main rivers are given, as well as the origin and features of stagnant water systems from ancient, large glacial lakes to smaller ones formed by glacial retreat, and shallow lakes and wetlands in endorheic basins. Also, economic activities based on the physical geography of each basin are discussed. With regard to water management, both countries have developed contrasting regulatory frameworks. On the one hand, Chile is a unitary country wherein one General Water Directorate manages water resources in accordance to the 1981 Water Code all over the country. This code grants transferable ‘water rights’ to particular users, thus enabling the existence of a water market. Argentina, in turn, is a Federal Republic, wherein each province regulates and manages the use of water. Nevertheless, as many basins are shared between provinces, there is a Federal Council which oversees the many interjurisdictional basin authorities. Yet, institutional regulation does not prevent the socio-environmental conflicts around water triggered by disputes in land possession and use. Furthermore, as projected consequences of climate change foresee critical flow decreases in many key river basins over the twenty-first century, water management will be a great governance challenge in years to come.

Amongst an important number of contributions to people, wetlands are invaluable sources of freshwater, particularly in Patagonian areas where this resource is scarce. In addition, these ecosystems contribute significantly to regional biodiversity. In Chap. 10, Epele et al. describe the different Patagonian wetland types as well as their main ecological characteristics, the uses they are subjected to, and the different threats they face on account of changes in both climate and land use. In particular, the authors introduce a detailed description about the following wetland types and their contributions to people: vertientes (*springs*), lagunas (*shallow lakes* and *ponds*), vegas, mallines and turberas (*peat bogs*). They also point out the actions that should be taken in the face of present Patagonian wetlands loss and degradation. In this context, they call for the implementation of concerted actions based on adequate knowledge about these ecosystems dynamics and functioning in order to revert this negative trend. They also observe that, despite being essential for the regional economy, Patagonian wetlands are still far from meeting global conservation targets, since less than 3% of them in arid and semiarid plateau are located within protected areas. These conclusions highlight the need for advancing conservation and sustainable management policies for wetlands over the entire Patagonian region.

Introduction of alien species is presently considered one of the main environmental threats worldwide on account of its impact on both biodiversity and natural ecosystems structure and function. In Patagonia, exotic fish introduced for sport fishing are now affecting native fish populations. In Chap. 14, Porcel et al. analyse the consequences of rainbow trout (*Oncorhynchus mykiss*) introduction in many Patagonian lakes. In this context, they conducted a comparison of trout effects on community composition of fishless and stocked lakes in the southern Patagonia Plateau. By exploring changes in primary productivity and food webs in wet and dry years, the authors found that rainbow trout shaped the community of pelagic crustaceans. In addition, they observed differences in phytoplankton structure and an increase in microplanktonic cyanobacteria abundance in stocked lakes particularly during dry periods, indicating the concomitant effect of trout introduction and climate fluctuations. Changes in both food webs and habitat coupling were also observed with trout presence. These impacts give rise to major potential consequences for waterbirds, particularly for the critically endangered Hooded Grebe (*Podiceps gallardoi*), which reproduces in those aquatic systems.

Changes and intensification in land uses are deeply affecting Patagonian freshwater and wetlands in recent decades. In Chap. 11, Miserendino et al. examine the influence of the main land use practices on the ecological integrity of Patagonian aquatic and wetland environments, as well as on the organisms inhabiting them at population and community levels. Although the region exhibits a diverse array of productive activities, two of them stand out: livestock rising and agriculture. While historical mismanagement of such pursuits has deeply impacted aquatic and wetland ecosystems, more recent human activities have also had important effects: pasture conversion, forestry, pine plantations, mining, damming, oil extraction and, to a lesser extent, urbanisation and industrial development significantly impacted periphyton, macrophytes and macroinvertebrate communities. According to these authors, integrity of riparian corridors plays a crucial role in the ecological integrity of these systems. They conclude that the implementation of management and conservation actions on these ecosystems is urgently needed, and propose mitigation measures to minimise the impacts produced by the different land uses types which should be conducted through collaborative work, involving governmental agencies, scientists, landowners and local communities. Importantly, the success of these measures should be assessed by long-term programmes to monitor the evolution of the impacted ecosystems. Because of the complex nature of these problems, undertaking these measures must also consider a socio-ecological perspective for the entire region.

Chapter 4 addresses the current state and recent changes of glaciers in the Patagonian Andes between ~37 °S and 55 °S. In this context, Ruiz et al. describe the distribution and characteristics of the Patagonian glaciers together with their recent changes and hydrological implications. According to the authors, the Patagonian Andes contain the largest glaciated area in both the southern Andes and the Southern Hemisphere outside Antarctica. This region includes 24,000 ice masses which represent ca 26,100 km$^2$. There are also several thousand smaller ice masses which are crucial water resources to people. As in other parts of the world, Patagonian glaciers are affected by climate change. The authors point out that in the last decades most of them have experienced considerable thinning and recession, yet recent findings highlight that they have not responded in a similar way to this phenomenon. While ice dynamic processes drive mass change of larger Patagonian glaciers, increases in ice melt and snowfall depletion would be the main causes of shrinkage of the smaller ice masses. Glacier retreat is expected to continue with the consequent impact on runoff and glacier-related hazards. In particular, increase in the number and size of proglacial lakes could result in larger, more frequent glacial lake outburst floods (GLOFs) in the Patagonian Andes in the near future, with the consequent impact on people and productive activities. As this will put an additional strain on water resources in the region, this synthesis on the state and trends of the Patagonian glaciers constitutes a valuable contribution for decision-making regarding their conservation and management.

### 2.5 Conservation and Protection: Are We Winning?

The invasive, bloom forming diatom *Didymosphenia geminata* (Didymo) exemplifies how environmental issues challenge institutional monitoring and conservation capabilities, as exposed by Beamud et al. (Chap. 12). In 2010, the first blooms of this species in South America were detected in Chile, and only 3 months later in Argentina. As of 2012, Neuquén Province (Argentina) led a series of regional meetings and launched a provincial programme aimed at identifying and preventing the presence of this nuisance diatom in waterbodies. This was of great importance, as Didymo grows in bulky, mucous masses on the riverbed of oligotrophic Andean streams and rivers much appreciated for productive and touristic activities. Along with public informative campaigns and sanitary stations, the authors conducted a province-wide monitoring programme between 2012 and 2019. Although the measures taken did not prevent the expansion of *D. geminata*, the information collected along the monitoring programme allowed to relate the temporal invasion pattern with recreational activities and to learn that the presence and abundance of the alga is primarily driven by factors other than those described in the literature, most importantly the river order. The authors call the attention to the fact that the environmental conditions of a number of invaded sites overlap those of many un-invaded ones, thus rendering the latter susceptible to invasion by *D. geminata*. Also, the wide range of features of invaded sites suggests an ongoing expansion of its ecological spectrum. On the basis of this experience, they also discuss the role of jurisdictional issues in delaying the effective response of control measures.

The state of freshwater and wetland ecosystems worldwide has a deep impact on freshwater biodiversity. Particularly, amphibians are currently amongst the most endangered taxonomic groups on a global scale. Therefore, they are considered a target wildlife group in many international conservation programs. Waterbirds, on the other hand, have a strong dependence on freshwater habitats, and many of them exhibit a congregatory behaviour, so they are particularly sensitive to many threats. Patagonia has a remarkable diversity of both amphibians and waterbirds highly adapted to live under extreme environmental conditions. Many species are endangered by both direct and indirect anthropogenic threats including climate change. In Chap. 15, Kacoliris et al. picture the diversity, ecology and history of the Patagonian amphibian and waterbird species, describe the most relevant ones and highlight their conservation status. The authors discuss the importance of these species in supporting ecological processes that link freshwater, wetland and terrestrial ecosystems and how conservation planning should integrate them to conserve threatened species. Interestingly, they also show how some amphibians and waterbirds have served in turn as flagship species for the protection of habitats and their biodiversity. Finally, the needs for the long-term conservation of both groups in Patagonian freshwaters and wetlands are discussed, including a description of the main conservation projects in the Argentinean Patagonia. The case of the Hooded Grebe (*Podiceps gallardoi)* conservation project, which is one of the main species-framed conservation projects in Patagonia, is particularly analysed.

## 2.6 Water and Environmental Justice in Patagonia

As already pointed out in Sects. 2.3 and 2.4, the present configuration of human settlements, productive activities and subsequent water uses in Patagonia results from the interplay between its geography and the history of its occupation in the light of global politics, especially since the industrial revolution. As a consequence, struggles around access to water – either as a vital substance or to waterbodies – persist to this very moment. In Argentina, vast foreign-owned land properties acquired through legal loopholes can encompass whole public lakes, such as Escondido Lake, in Río Negro province, included in Hidden Lake ranch, property of the British billionaire Joseph Lewis (Iñigo Carrera 2020). Although justice granted access to the lake shore, partakers of the Sixth March for the Sovereignty of Escondido Lake were recently intercepted, harassed and illegally retained by armed Lewis employees (Página 12 2022). Such episodes are not uncommon and call for urgent, effective action at all governmental levels to democratise access to water. In

Chap. 17, Giomi et al. adopt the perspective of the Latin American Political Ecology to analyse the role of different water valuation languages underlying conflicts around water uses. The many visions of water as an economic good, a human right or a key ecosystem element conduce socio-political actors to regarding it as a public, private or common good, and hence take stands on the many conflicts derived from different uses and the environmental consequences thereof. Within this framework, authors expose the obstacles submerged in the so-called *subsoil of politics* that impeded the legislative deliberation of as much as 13 successive water bills over more than 20 years in the Tierra del Fuego, AIAS Province. Only by revealing the cultural debate amongst disparate water valuations was the construction of social legitimacy possible in order to bring the most powerful actors to negotiate the terms of a Water Law at the surface of politics level. The many conflicts due to the previous lack of regulation are described, as well as the key role of some institutions and state agencies, and the parallel processes in both levels of politics which led to the passing and regimentation of the present-day Water Law. The uncovering and analysis of such processes is particularly valuable in the face of Argentina's persistent failure to table a national law of minimum standards for the protection of wetlands.

A number of environmental justice matters have been tackled at a global level by the UN 2030 Agenda for Sustainable Development, which includes 17 Goals (SDGs) for world countries to fulfil human needs and expectations in an equitable and environmentally sustainable manner. Amongst these, SDG 6 on Clean Water and Sanitation strives to secure universal and equitable access to drinking water and sanitation, while also strongly related with SDG 3 through the importance of clean water supply to prevent and reduce illnesses; SDG 11 about risks posed by water-related disasters; SDG 12 in remarking how waste affect freshwater; and SDG 15 by highlighting the importance of protection, restoration and sustainable use of inland freshwater ecosystems and their services. Due to the abundance of freshwater and scarce population in Patagonia, the challenge of meeting SDG 6 does not relate to water availability but rather to its wise use guided by solid scientific knowledge and strong governance. As the integrated water resource management (IWRM) is recognised by the UN as a keystone policy to meet this objective, in Chap. 18 Pascual et al. critically examine the causes for the low degree of implementation of IWRM by both Argentina and Chile, which relates to the poor application of science and technology to water research and management, and water management being a top-down, bureaucratic process. Globally, the notion that human well-being depends directly on the multiple benefits that societies obtain from nature – termed 'ecosystem services' or 'Nature's contributions to people' – has fostered the application of the ecosystems services (ES) paradigm in both the academic and policy design fields. This paradigm allows to analyse the land–water–atmosphere interface in a dynamic way and to evaluate the consequences of different human interventions and remediation options for aquatic, terrestrial and wetland environments. Therefore, in the case of Patagonian freshwater and wetland ecosystems, the perspective of the ES can bridge the gaps amongst the views of water of different disciplines, thus promoting an integrated environmental perspective of water security problems and implementation of nature-based solutions as well as infrastructure-based solutions.

Furthermore, while IWRM is more paradigmatical, ES allows for normative products and applications. The authors thus propose that its assimilation can strengthen IWRM, reinforcing the environmental view on water management issues, providing technical tools to map and quantify the flow of ecosystem services, and fostering the objective evaluation of nature-based solutions as an alternative to solve such issues. This would represent a most needed change of paradigm towards reaching a more equitable water access for all.

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# Chapter 2
# Are Wetlands Freshwaters?

**Rubén D. Quintana and Gabriela Mataloni**

## 1 Introduction

Limnology is defined as the scientific study of inland waters, including not only well-defined aquatic ecosystems (e.g. rivers, lakes and reservoirs) but also a number of other particular types (e.g. ponds, swamps and marshlands). As was pointed out by Margalef (1983), limnology is the ecology of non-oceanic waters. Wetland ecology, on the other hand, focuses on the scientific study of an array of ecosystems globally called 'wetlands' wherein water is determinant in their structure and functioning (Keddy 2016). It considers both continental (or 'interior') and marine-coastal ecosystems like mangroves, estuaries and salt marshes. From this point of view, only some wetlands are freshwater ecosystems because many of them depend on salt water.

Limnology and wetland ecology are intimately related because water is an essential factor that cannot be ignored in continental aquatic and wetland ecosystems studies. Anyway, there are certain aspects that clearly differentiate both fields of knowledge. In fact, limnology focuses on permanent aquatic ecosystems or, at least, on those that undergo temporary drought. Wetland ecology, on the other hand, while excluding large aquatic systems wherein a pelagial zone can be differentiated, includes not only this types of ecosystems but also environments wherein the rhizospheric area of the soils is permanently saturated with water, which Brinson (2004a) named as 'dry wetlands'.

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## 2 But, What Are Wetlands?

Although the term ‘wetland’ dates back to the mid-twentieth century (Shaw and Fredine 1956), even today there is no consensus on the concept of wetland within the scientific community. For example, Cowardin et al. (1979), in their definition of wetlands, state that they are transitional lands between terrestrial and aquatic systems. Although it is true that some wetlands can be considered as ecotones between terrestrial and aquatic ecosystems (e.g. wetlands located in alluvial plains), it is also possible to find wetlands inserted in a terrestrial matrix, without any connection with an aquatic ecosystem. In fact, from a scientific perspective, wetlands are water-dependent ecosystems that are neither terrestrial nor aquatic ones. On the one hand, unlike aquatic ecosystems, they have soils but, on the other hand, they have standing water, so they are not truly terrestrial. For this reason, some limnologists have historically considered that wetlands were an object of study for terrestrial ecologists and vice versa (Keddy 2016).

Despite wetlands arising fundamentally because there is water, there are also other key environmental factors (e.g. hydrological cycle, disturbances, nutrients) on which the different wetland types depend upon. In particular, the hydrological cycle is essential to define their structural and functional features. Consequently, the study of wetland ecology also focused on them (Keddy 2016). According to Mitsch and Gosselink (2015), wetlands display particular properties which are not properly covered by terrestrial or aquatic ecology paradigms as well as by different fields such as limnology and estuarine ecology. These authors pointed out that wetland science should be thought of as a unique discipline, which encompasses these fields and others like chemistry, hydrology and engineering.

The International Convention on Wetlands (Ramsar, Irán, 1971) itself contributed to the confusion generated around the wetland concept because it adopted an enumerative definition of wetlands[1] that includes purely aquatic ecosystems. This definition could be considered adequate in a management context because it is not possible for the sustainable use and conservation of many wetlands without the conservation of aquatic ecosystems they depend upon.

The Keddy’s (2016) wetland definition stated that is an ecosystem that arises when the presence of water (flooding or waterlogging) produces soils dominated by anaerobic processes and forces the biota, particularly rooted plants, to present adaptations for tolerating the flood. This definition summarises the true character of wetlands: a diverse array of ecosystems with standing water or waterlogged soils, anoxic conditions and plant and animal adaptations to these environmental conditions.

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[1] For the purpose of this Convention, wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed 6 metres. May incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than 6 metres at low tide lying within the wetlands.

In Argentina, an expert panel proposed in 2016 the following wetland definition, which was adopted in the framework of the National Wetland Inventory: A wetland is an environment in which the presence of temporary or permanent water, both surface or subsurface, determines its own biogeochemistry flows, different from terrestrial and aquatic ecosystems. The presence of adapted biota to these conditions, such as hydrophytic plant species and hydric soils or hydromorphic substrates, is a distinctive feature of these ecosystems (Kandus and Minotti 2018).

## 3 The Brinson's Approach

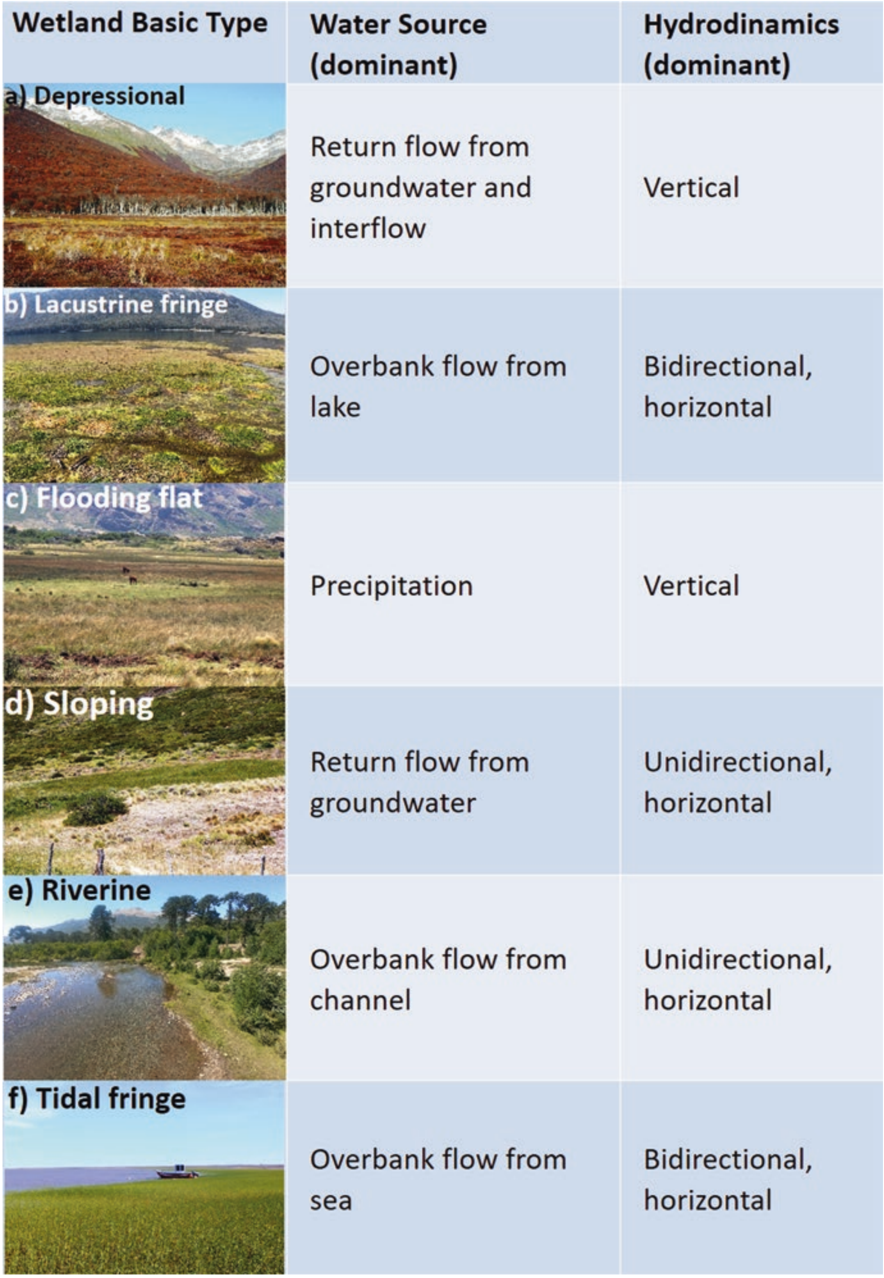
According to Brinson (2004a), within the same climatic-geographical region, it would be expected a greater variability of wetland habitats than that of terrestrial ones. This situation is a consequence of the control that hydrological and geomorphological conditions exert on these ecosystems. Factors in environmental control of wetlands include geomorphic setting, soil properties, fire frequency, hydroperiod and water sources (Brinson 1993). In this context, a hydrogeomorphic classification of wetlands was proposed by Smith et al. (1995), which is considered as a functional wetland classification (Brinson 2004b). This classification system considers six basic functional classes: depressional, lacustrine fringe, tidal fringe, sloping, flooding flats and riverine wetlands (Smith et al. 1995; Fig. 2.1).

The geomorphic setting influences both the water sources and the hydrodynamic or water movement when it gets into the wetland. Water sources are precipitation, horizontal surface or subsurface flows and groundwater discharges. Hydrodynamic, in turn, refers to water direction (horizontal or vertical) and energy. As for water energy, two situations can be found: low-energy systems (e.g. depressional wetlands with vertical fluctuations of water level) or high-energy systems where water shows an important erosion power (e.g. flooding flats and tidal fringes) (Brinson 2004b).

In this classification scheme, both climatic regime and topographic position determine the general water supply but geomorphic setting influences water sources and hydrodynamics. These features then define the hydrological regime which, in turn, determines the main structural and functional characteristics of wetlands and, therefore, their biotic communities (Kandus et al. 2011).

## 4 Patagonian Inland Wetlands

The wetlands regionalisation in Argentina (Benzaquén et al. 2017) involved the integration of Brinson's hydrogeomorphic approach (Smith et al. 1995) with other ecological aspects considered by Janauer (2000) and Vaughan et al. (2007). Variables related to the regional water balance and the geomorphic setting were considered as the main factor conditioning the presence of wetlands at the regional scale and their characteristics at greater detail scales.



| Wetland Basic Type | Water Source (dominant) | Hydrodinamics (dominant) |
|---|---|---|
| a) Depressional | Return flow from groundwater and interflow | Vertical |
| b) Lacustrine fringe | Overbank flow from lake | Bidirectional, horizontal |
| c) Flooding flat | Precipitation | Vertical |
| d) Sloping | Return flow from groundwater | Unidirectional, horizontal |
| e) Riverine | Overbank flow from channel | Unidirectional, horizontal |
| f) Tidal fringe | Overbank flow from sea | Bidirectional, horizontal |

**Fig. 2.1** Wetland categories according to their geomorphic settings showing dominant water sources and hydrodynamics (Based on Smith et al. 1995 and Brinson 2004b). Pictures show examples of the wetland basic types in the Argentinean Patagonia: (**a**) Depressional wetland, *turbera* in Valle Tierra Mayor Natural Reserve, Tierra del Fuego. (**b**) Lacustrine fringe bordering the Moquehue Lake, Neuquén. (**c**) Sloping wetlands around the Agrio River, Neuquén. (**d**) Flooding flat valleys next to Route 27, Caviahue, Neuquén. (**e**) Riverine wetlands, Quillahue Stream, Moquehue, Neuquén. (**f**) Tidal fringe, salt marshes in Pocitos, Buenos Aires. (Photos by Rubén Quintana)

As for Patagonia, except for the coastline, along which only coastal wetlands are found, most of this region is composed by three different interior wetland subregions which differentiate on topographic and climatic features: (1) *Lakes, watercourses and mallines of the Patagonian Andes*. (2) *Lagunas and vegas of the extra-Andean Patagonia.* (3) *Mallines and turberas of Austral Patagonia and South Atlantic Islands* (Benzaquén et al. 2017).

The subregion 'Lakes, watercourses and mallines of the Patagonian Andes' includes diverse wetland types such as lakes, *mallines* (freshwater marshes), *turberas* (peatlands) and periglacial environments (see Chaps. 3 and 10). Lakes were originated by tectonic or glacial processes and cover areas greater than 5 km$^2$ and deeper than 100 m (reaching 400 m in the Nahuel Huapi lake). The northernmost lake is the Aluminé, in Neuquén province, while the southernmost continental one is the Argentino lake in Santa Cruz province. In addition, lakes less than 5 km$^2$ and 12 m deep are also located in this subregion, which are covered by hydrophytic vegetation when located at an altitude less than 1700 masl. In both types of lakes, wetlands are restricted to a littoral zone, corresponding to the Brinson's lacustrine fringe (Buria 2017) (see Chap. 3).

In this subregion, there exists a wide variety of *mallines* of different sizes, from small isolated patches to complex wetland systems associated to flooding plains and extense depressions (Buria 2017) (see Chap. 10). They are characterised by the presence of hydrophytic prairies of Cyperaceae, Juncaceae and Poaceae species (Roig 1998; Raffaele 1999). Depending on their location, these wetlands correspond to the Brinson's types depressional (those located in depressions) or riverine (those associated to rivers).

*Turberas* are located in areas with precipitation greater than 1000 mm; they are found in riparian areas, river embankments and borders of springs (Blanco and de la Balze 2004). They accumulate dead organic material because of permanent waterlogging conditions. Depending on the type of *turbera*, the dominant vegetation is a prairie of Cyperaceae species or a continuous cover of *Sphagnum* mosses. Certain turberas can include small water bodies (Mataloni 2017) (see Chaps. 6 and 10). In this subregion, *turberas* are restricted to particular sectors within rainy forests (Roig 1998; Rovere et al. 2002). According to their location, *turberas* can be assigned to the following Brinson's categories: sloping and depressional wetlands.

The subregion '*Lagunas and vegas of the extra-Andean Patagonia'* is characterised by the presence of isolated and dispersed wetlands. Wetlands are usually small in size with areas ranging from tens to a few hundred hectares. Two main wetland types are found: *lagunas* (shallow lakes less than 5 m deep) and *mallines* (also locally known as *vegas*) (see Chap. 10). *Lagunas* are usually located in endorheic depressions locally called '*bajos sin salida'*, in sedimentary environments and over basaltic flows; most of them show an exclusive dependence on precipitations so that their use to have a temporary hydrologic regime (Mazzoni 2017). *Lagunas* are depressional wetlands according to the Brinson's categories.

Similar to the previous subregion, *mallines* are freshwater marshes located in different geomorphic settings which include four of the six Brinson's wetland types: depressional, sloping, riverine and flooding flat wetlands. They cover an area between 1 and 4% of the subregion (Bran 2004).

The last subregion, '*Mallines and turberas of Austral Patagonia and South Atlantic Islands*', is characterised by the presence of *mallines* and *turberas* (Mataloni 2017) (see Chaps. 6 and 10). According to Martínez Carretero (2004), three main wetland units are described: Andean *turberas*, within forests; *mallines* and wet prairies. While *turberas* are found mainly in the province of Tierra del Fuego (Blanco and De La Balze 2004), mallines are located in the steppe zone, covering 30% of its surface south of the Coyle River (Collantes and Faggi 1999). In the steppe of Santa Cruz province, there are also many *lagunas* (shallow lakes) both temporary and permanent as well as small ponds produced by eolic erosion, which can be temporarily flooded (Mataloni 2017). In this subregion, four Brinson's wetland types are found: depressional, riverine, sloping and flooding flat wetlands according to their geomorphic location.

In Chile, 95% of wetland area is located in the Patagonian region, particularly in Regions XI (*Aysén*) and XII (*Magallanes y la Antártica Chilena*) (Promis 2010). Chilean Patagonia presents both coastal and continental wetlands which cover approximately 43263.42 km$^2$ (35.37%) (Schlatter and Schlatter 2004). The main continental wetland types are similar to those found in Argentina (e.g. *turberas, mallines*, lakes fringes and shallow lakes) but in that country, wetland names based on edaphic, geomorphic hydric and botanical aspects coexist (Schlatter and Schlatter 2004).

Peatlands also reach their maximum development in these regions (Schlatter and Schlatter 2004). As for *mallines* and *vegas*, while in Argentina both names are considered as synonyms, in Chile they represent different wetland types. There, *vegas* are wetlands with fluvial sediments in alluvial plains which are constantly under the flooding effects (Schlatter and Schlatter 2004). From this point of view, they are equivalent to *mallines* associated with rivers from the Argentinean sector. In Chile, *Mallines* or *vegas* are particularly abundant in the steparian area of the Magellanic Region (Filipová et al. 2013). Ñadis and cátricos constitute other particular wetland types present in the Chilean sector of Patagonia. *Ñadis* are swampy wetlands (Promis 2010) with volcanic ashes-soils over sand and/or gravels and located in flat areas. They are waterlogged between 4 and 8 months per year, usually in the May–September period. These environmental conditions favour organic matter accumulation. They show a heterogeneous and discontinuous vegetation cover, adapted to winter wet soils, including evergreen forests although in best drained *ñadis*, deciduous species can be also found. Some sites show an important abundance of *Sphagnum* moss, with the subsequent peatland aspect (Ramírez et al. 1993; Hauenstein et al. 2005; Ramírez and San Martín 2008).

*Catricos* are lowlands which collect water from the surrounding areas. Their name derives from *catrú*, mapundungum word from de Mapuche Nation, which means 'divided or isolated by pits and *quebradas*'. Usually, they maintain swampy evergreen forests. They are originated by hydric erosion processes and characterised by permanently waterlogged or flooded soils during winter, which are relatively rich in organic matter (Ramírez et al. 1983; Hauenstein et al. 2005; Ramírez and San Martín 2008).

## 5 Wetlands and Society

Humans have had a contradictory attitude towards wetlands throughout history. On the one hand, these ecosystems have provided water, food and construction materials, but on the other hand, they have been considered wastelands due, among other reasons, to the fact that they are places under recurrent flooding cycles wherein many disease vectors thrive. In this sense, as pointed out by Keddy (2016), it is worth asking how it is that one system can be both life-giving and death-dealing. In addition, wetlands have been usually considered as sinister and threatening because of their appearance (Mitsch and Gosselink 2015). In fact, prior to the widespread use of the word ‘wetland’ to name these ecosystems, terms with negative connotations were used (e.g. swamp, bog, barren land).

According to Mitsch and Gosselink (2015), from the beginning of civilisation, many cultures lived in harmony with wetlands whereas others quickly transformed them by different actions like drainages and embankments to avoid their negative aspects.

Our own origin in Africa is linked to a wetland; Chan et al. (2019) pointed out that the first human lineage emerged within the residual Makgadikgadi–Okavango palaeo wetland of southern Africa, approximately 200,000 years BP. Wetlands around the world were also early occupied by humans since the late Pleistocene. For example, humans have inhabited with the consequent impacts the European tidal freshwater wetlands in both Neolithic period and Bronze Age (starting 800 BC) (Zonneveld and Barendregt 2009). Even in environments as extreme as the hyper-arid South American Atacama Desert, humans colonised wetlands far away from the Pacific Ocean. During the latest Pleistocene, this region harboured wetlands due to the increase of rainfall in the central Andes. These paleowetlands oases would have sustained human populations (Latorre et al. 2013). Archaeological records in palaeowetlands from the Tibetan Plateau show Epipaleolithic human occupation after 6000 years BP, supporting evidence for widespread colonisation of that area in the early and mid-Holocene during warm, wet post-glacial conditions (Hudson et al. 2016).

Summarising, wetlands have been and continue to be part of many human cultures in the world. Ancient Babylonians, Egyptians, Mayas and Aztecs, among others, had an intimate relationship with the wetlands in which they developed (Mitsch and Gosselink 2015). So much so that Coles and Coles (1989) coined the term ‘wetlanders’ to name the people who live in proximity to wetlands and whose culture is linked to them.

As for wetlands contributions to people, these ecosystems are among the most important in the world since they provide approximately 40% of all ecosystems contributions (Costanza et al. 1997). For example, wetlands not only provide good-quality water (in the case of freshwater wetlands) but also many other valuable goods to humans. In Patagonia, lakes and shallow lakes are sources of freshwater and fish (Chaps. 3 and 13) while *mallines* are important areas for livestock grazing (Chaps. 10 and 11). In addition, they are recognised as important carbon sinks and climate stabilisers on a global scale such as the Patagonian *turberas* (Chaps. 6 and 10). Patagonian wetlands also provide a high habitat availability for particularly threatened wildlife species, as observed in Chap. 15.

These contributions and many others are provided by wetlands only if their structure and functioning is kept and they are lost when wetlands are degraded, mainly due to changes in their hydrological regime. Land use change has the greatest impact on wetland ecosystems, particularly agricultural expansion (IPBES 2019). For example, Zedler (2003) pointed out that flood control, water quality improvement and habitat suitability for biodiversity declined when about 60% of the wetland area was drained, mostly for agriculture, in the Upper Midwestern region of the United States. In the lower Delta of the Paraná River (Argentina), cattle rising intensification and afforestations with poplars (*Populus* spp.) and willows (*Salix* spp.) were the main drivers in wetlands conversion between the mid-1990s and the beginning of the second decade of the current century, with a decrease of 41.85% (Sica et al. 2016), and the consequent impact on biodiversity (Sica et al. 2018) and lack of flooding regulation (Aquino et al. 2021). As a consequence of these impacts, during the twentieth century, the loss and degradation of these ecosystems ranged between 64 and 71% and continues today. Thus, between 1970 and 2015, there was a global decline of approximately 35%, which represents more than three times the average annual rate of natural forests loss (Convention on Wetlands 2018). Particularly, the most widespread deterioration reported was for Africa and Latin America and the Caribbean wetlands (McInnes et al. 2020). This fact impacts directly on biodiversity; since 1970, inland wetland-dependent species have declined far more than species dependent on other ecosystems and an increasing number are facing extinction (Convention on Wetlands 2021).

In Patagonia, continental wetlands as well as aquatic ecosystems are facing many threats associated with global change. Particularly, changes in land uses (Chap. 11), introduction of alien species (Chaps. 12, 13, 14, and 15), pollution (Chap. 11), climate change (Chaps. 4, 10, and 11) and losses of biodiversity (Chaps. 13 and 15) are among the main threatening factors. Wetlands are critical ecosystems for people in this region, particularly as freshwater sources. Large-scale real estate or productive projects in synergy with climate change will put much more pressure on these ecosystems in the short term. Therefore, there is an urgent need for public policies focused on their conservation and sustainable use, considering not only their value as habitats for biodiversity but also their contributions to people.

But, what is to be expected from the application of such policies? A desirable ‘null net losses’ – scenario at regional and national levels implies achieving a null net loss of wetland area and/or its ecological character. Likewise, a policy of public investment in large-scale restoration of wetlands would allow a reduction in disaster risk and an increase in the resilience of local communities, (e.g. by securing water supply for domestic use and pastures) (Convention on Wetlands 2018). On March 2, 2022, the Project on Minimum Budgets for Environmental Protection for the Rational and Sustainable Use of Wetlands was presented in the Lower House again. This will be the fourth attempt to sanction a Wetland Law in the country since 2013, as this had lost parliamentary status at the end of 2021 due to constant pressures of the economic and productive lobbies for avoiding its approval, in a similar way to that revealed by the analysis of the process conducting to the sanction of the Water Law in Tierra del Fuego, AIAS (Chap. 17). It is desired that this time the increasing urgency to take bold action towards protecting the health of both ecosystems and people will grant the political willingness to remove such impediments and the sanction of a Wetlands Law can finally be achieved.

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# Chapter 3
# North Patagonian Andean Deep Lakes: Impact of Glacial Recession and Volcanic Eruption

**Esteban Balseiro, Beatriz Modenutti, Marcela Bastidas Navarro, Nicolas Martyniuk, Luca Schenone, and Cecilia Laspoumaderes**

> *'Qué tranquilo y bello el cuadro en las cercanías del Leman argentino ¡más grandioso que el Suizo!.' (How calm and beautiful scene in the surroundings of the Argentinian Leman, more magnificent than the Swiss one!)*
>
> *F.P. Moreno – January 22, 1876.*

## 1 Introduction

The largest lake basin in Argentina is in the Andean-Patagonian region; these lakes are the heart and essence of the landscape of the region. Considering their volume, these deep and large lakes (area >5 $km^2$, depth $\geq$100 m) are important water reserves that represent 77% of the water contained in lakes of Argentina (Quirós 1988).

The first data on these Patagonian lakes were obtained by geographers in the last decades of the nineteenth century, who referred to the similarity with lakes in the Alps (Biedma 1987). In the early twentieth century, salmonids were successfully introduced in most of the deep lakes (Quirós and Drago 1999). However, it was until the 1950s when biological samples were studied for the first time by the Swedish

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expedition of Kuno Thomasson (Thomasson 1959; Thomasson 1963). Thus, there are no scientific samples from before this introduction occurred.

The area has been affected by glacial processes that modeled the landscape including the lakes. In addition, the region has been affected by the activity of several volcanos producing volcanic ashes. These events (including glacial recession and catastrophic events such as moraine breaks and volcanic eruptions) cause changes in the input of inorganic suspended particles into lake ecosystems.

## 2 The Deep Andean-Patagonian Lakes of Argentina

Deep Andean-Patagonian lakes of Argentina (maximum depth >100 m) correspond to the Glacial lake district of the Southern Andes (Iriondo 1989) and the Wet Andes according to glacio-climatological regions (Lliboutry et al. 1998). This lake district in Argentina extends from 37° S (Lake Aluminé) to 55° S (Lake Fagnano), covering almost 2000 km (Fig. 3.1). During the late Pleistocene, the area was glaciated extensively and repeatedly. Thus, the landscape is dominated by glacial processes (moraines and glacial-fluvial plains) and also by volcanic events (Pereyra and Bouza 2019). Climate is cold temperate, with the prevalence of west-winds (westerlies) coming from the Pacific Ocean (Paruelo et al. 1998; Masiokas et al. 2008). Elevation of most mountain peaks and massifs does not exceed 4,000 m a.s.l. and the intense influence of the westerly circulation from the Pacific results in high precipitations (Kitzberger and Veblen 2003). Thus, the region is characterized by strong precipitation gradients, with clear contrasts between the western (~3000 mm $y^{-1}$) (Viale et al. 2019) and the eastern slopes (2000–1500 mm $y^{-1}$) (Paruelo et al. 1998; Viale et al. 2018). However, diverse macro-climatic phenomena such as SAM (South Annular Module), El Niño Southern Oscillation (ENSO), and the Pacific Decadal Oscillation (PDO) have affected interannual and month-to-month variations in the intensity of rains (Trauth et al. 2000; Rivera et al. 2018; Viale et al. 2019; Poveda et al. 2020). The region contains a wide variety of glaciers including permanent snowfields, mountain glaciers, valley glaciers, outlet glaciers, piedmont glaciers, icecaps, and extensive icefields (Masiokas et al. 2020; Chap. 4). The presence of a profuse hydrographic system including large deep lakes and rivers is characteristic of the Wet Andes, wherein 4800 $km^2$ of the former extent of large glaciers are now occupied by more than 4,000 lakes (Wilson et al. 2018). However, the high precipitation amounts cause that most river flow is dependent on rainfall and snowmelt patterns (Masiokas et al. 2019). The rivers fed from Andean waters cross the Andes, flowing toward the Pacific Ocean or run through the Patagonian plateau steppe and outflow to the Atlantic Ocean (Chap. 9).

The vegetation of the area is mainly composed of evergreen and deciduous trees dominated by species of the genus *Nothofagus*. In particular, the deciduous species of the southern beech, locally named "lenga," *N. pumilio* (Poepp. et al.) Krasser, occurs from 35° S down to 55° S (Hildebrand-Vogel et al. 1990). Within the broad

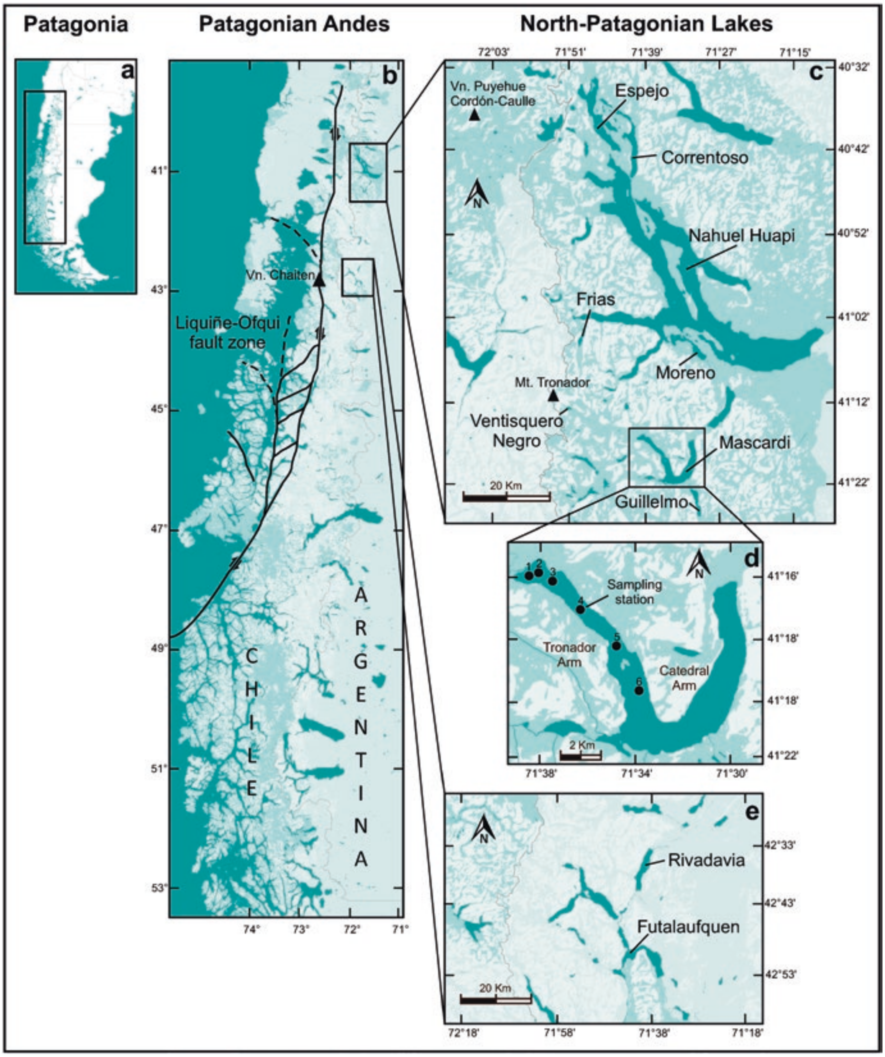

**Fig. 3.1** Maps of the Andean Patagonian lake area. (**a**) Location of the Andean lakes. (**b**) Andean lakes and the Liquiñe-Ofqui fault in the west side of the Andes. (**c**–**e**) Details of the most studied lakes. In (**d**), the sampling stations (numbered) are indicated along the Tronador arm of Lake Mascardi

distribution area of more than 2000 km in length, the altitudinal distribution varies, constituting high mountain forests up to the timberline (Krummholz: stunted wind-blown trees) to pure stands at the sea level in the Southern Tierra del Fuego (Mathiasen and Premoli 2010). *N. pumilio* constitutes an important source of organic matter for both lakes and rivers (Albariño and Balseiro 2001; Bastidas Navarro et al. 2014; Bastidas Navarro et al. 2019).

The lakes exhibit a warm, monomictic thermal regime, with stable thermal stratification during late spring and summer (Baigún and Marinone 1995). Thermocline depth ranges between 15 and 40 m (or more), and this is related to lake morphometry, fetch, and interannual wind variations (Baigún and Marinone 1995; Pérez et al. 2007; Modenutti et al. 2008). Transparency is extremely high with very low vertical extinction coefficients of the different wavelengths (Kd PAR = 0.10–0.16 $m^{-1}$) (Morris et al. 1995). However, some lakes are under the influence of clay discharged by glaciers such as Lake Argentino, Viedma, and Mascardi, and in these lakes, transparency decreases sharply (Modenutti et al. 2000; Richter et al. 2016). Earlier baseline studies indicated that nutrient concentrations are very low corresponding to the oligotrophic and undisturbed lake status (Calcagno et al. 1995; Markert et al. 1997; Quirós and Drago 1999).

Extensive studies of the whole lake district are scarce and most of them were carried out in a single summer campaign (Quirós 1988; Drago and Quiros 1995; Quirós 1997; Quirós and Drago 1999). However, the North Patagonian Andean lakes (around 39° to 42°S and 71°W, 400–750 m a.s.l.) were more intensively studied (Modenutti and Balseiro 2018). In this chapter, we will focus on this lake district located at mid-latitudes (Fig. 3.1). We aimed to summarize the events that occurred in the water column of deep oligotrophic lakes as a consequence of glacial recession due to climate change and natural catastrophic events (volcanic eruptions and sudden moraine breaks). These events produced noticeable changes in the light distribution of the water column because of the increase of suspended solids (volcanic ashes or glacial clay), affecting the plankton communities' interactions, and thus pelagic ecosystem functioning.

## 3 Climate Change

In North Patagonian Andes, contemporary changes in the atmosphere have caused precipitation and mean temperatures to change at values previously experienced in geologic time; however, the velocity of change appears to be faster than that of similar periods (Marcott et al. 2013; Neukom et al. 2019). In particular, the increase in temperature (up to 1 °C since 1950) has been higher than in the rest of Argentina (Villalba et al. 2003; Barros et al. 2014), while precipitation significantly decreased (around 5% per decade) (Castañeda and González 2008; Masiokas et al. 2008). Thus, climate models predict around 10–30% less precipitation over northern Patagonia by the end of the century (Marengo et al. 2011; Barros et al. 2014; Pessacg et al. 2020), while in terms of temperature, simulations project an increase from 1.5 to 3 °C for the far future scenario (2071–2100) (Pessacg et al. 2020).

Glaciers are sensitive to climate change (Roe et al. 2017) and in the North Patagonian Andes (40.5°–44.5° S) glaciers decreased ~25% of their area between 1985 and 2011 (Paul and Mölg 2014), and this negative balance was confirmed by remote sensing data (Dussaillant et al. 2019; Chap. 4). As a consequence, the number of proglacial lakes and lake volume increased in response to climate change and glacier retreat (Paul and Mölg 2014; Shugar et al. 2020). In particular, the upper slopes of Monte Tronador host one of the largest contiguous ice covers (~57 km$^2$ in 2012) in the northern Patagonian Andes (Ruiz et al. 2015). Glaciers in Monte Tronador show a clear retreating and thinning pattern with a long-term frontal recession (Bown and Rivera 2007; Masiokas et al. 2010; Paul and Mölg 2014; Ruiz et al. 2017). As glacial lakes drain, hydrologic and geomorphic changes can occur (Kershaw et al. 2005; Harrison et al. 2018; Shugar et al. 2020) including catastrophic and hazardous events as glacial lake outburst floods (GLOFs) due to the sudden release of a glacier or moraine-dammed lakes (Masiokas et al. 2020; see Chap. 4). These GLOF events produce strong impacts in downstream valleys and the runoff of the outlet rivers (Dussaillant et al. 2012). In Monte Tronador in 2009, a GLOF event occurred due to heavy precipitation and ice blockage of the outlet, causing a break of the end moraine that impounded the proglacial Lake Ventisquero Negro (Worni et al. 2012). The event modified the downstream valley, and then the volume of the proglacial Lake Ventisquero Negro increased (Ruiz et al. 2017; Modenutti et al. 2018a).

Bedrock abrasion by glaciers from Monte Tronador produce fine rock and mineral fragments which constitute the glacial flour (glacial clay) that is carried by meltwaters through streams (Chillrud et al. 1994) to proglacial lakes (Ariztegui et al. 2007). Glacial clay from the Argentinian side of the Monte Tronador is carried into three main proglacial lakes: Ventisquero Negro, Frias, and Mascardi affecting the light vertical distribution (Bonetto et al. 1971; Morris et al. 1995; Modenutti et al. 2000). Variations in the transport of glacial clay to the basin can be directly linked to changes in climate (Ariztegui et al. 2007). The marked retreating of Ventisquero Negro glacier (Ruiz et al. 2015) and the 2009 GLOF event (Worni et al. 2012) caused a concomitant change in sediment carried downstream by Upper Río Manso that produced substantial variation in the light distribution in Lake Mascardi Tronador Arm (Bastidas Navarro et al. 2018).

## 4 Volcanic Eruptions

In the southern Andes, from 40 to 46 °S, there is an extended fault zone called Liquiñe-Ofqui, which extends for about 1000 km NS with more than 40 active volcanos (Cembrano et al. 1996). This fault coincides with the Quaternary volcanic chain that contains recent active stratovolcanos as Chaiten and Puyehue-Cordón Caulle. In May 2008, the Chaiten erupted about 1 km$^3$ of ashes (Carn et al. 2009), and in June 2011, the Puyehue-Cordón Caulle had a mega eruption with a spread of more than 100 million tons of pyroclastic material mostly carried by wind to the east side of the Andes, affecting a wide area of Argentina (Elissondo et al. 2016) including deep lakes (Elser et al. 2015). The eruption carried pumice of several sizes (4 mm to >10 cm in diameter) to the lake surface. Due to the low density of the sponge-like rocks, this pumice remained floating on the water surface from weeks to many months, creating new conditions for aquatic life (Elser et al. 2015;

Modenutti et al. 2016). In addition to the pumice, ashes from the volcano affected lakes, as they drastically changed their optical climate, turning highly transparent waters to grey, cloudy ones (Modenutti et al. 2013b). At least six historical eruptions of the Puyehue-Cordón Caulle have been recorded (1759, 1893, 1921, 1960, 1990, 2011) (Elissondo et al. 2016). Thus, lakes in the area receive this volcanic input periodically.

## 5 Lake Water Column: Light, Temperature, and Nutrients

Light and nutrient supply dynamics interacting with thermal water column structure and internal food web interactions are decisive for pelagic ecosystem functioning (Sterner et al. 1997; Falkowski and Raven 2007). Geologically, the region of North Patagonian Andean deep lakes is dominated by a mixture of crystalline igneous, volcanic, and plutonic rocks, while sedimentary rocks are proportionally scarce (Flint and Fidalgo 1964). Thus, waters in lakes and rivers in the region are extremely dilute solutions in which major ion concentrations are below world averages (Pedrozo et al. 1993). Nutrient concentrations are also very low (total phosphorus $<6\ \mu g\ L^{-1}$ and total nitrogen $<100\ \mu g\ L^{-1}$) and no significant differences throughout the water column were noticed (Corno et al. 2009). Within the lake water column, light may have a complex pattern of spatial and temporal variability (Litchman 2003; Stomp et al. 2007a, b), and fluctuations in irradiance may affect photosynthesis and respiration (Quéguiner and Legendre 1986; Ferris and Christian 1991; Falkowski and Raven 2007), as well as primary producers' growth rates (Litchman 2000, 2003). Light is a distinctive feature in deep North Patagonian Andean lakes since early studies have reported their high transparency to different wavelengths including ultraviolet bands, due to the low dissolved organic carbon (DOC) (Morris et al. 1995). The concentration of DOC shows values below 0.6 mg $L^{-1}$ (Corno et al. 2009) and remains without changes along precipitation gradients (Zagarese et al. 2017; Queimaliños et al. 2019). Considering light and nutrients, these deep lakes can be described as high-light:low-nutrient environments (Balseiro et al. 2007), and this condition drives most processes that occur in the water column.

Comparing the different wavelengths, in most of the deep transparent lakes, the photosynthetically active radiation (PAR, 400–700 nm) can reach up to 40–50 m depth (euphotic zone) while hazardous UV-B (305 nm) and UV-A (320 nm) affect up to 12–20 m depth, respectively (Fig. 3.2a). However, this condition changes when lakes receive the input of glacial clay producing a reduction of the euphotic zone (e.g., in Fig. 3.2b see lakes Frías and Mascardi 3).

Lake thermal structure is also related to the light availability in the different layers of the water column. North Patagonian Andean lakes are warm monomictic, and summer stratification is characterized by a wind-mixed surface layer that is isolated from colder deep waters by a marked thermal gradient at the metalimnetic level (Pérez et al. 2002). Interestingly, extended euphotic zones include the epilimnion, the metalimnion, and, in certain cases, the upper part of the hypolimnion (Fig. 3.3).

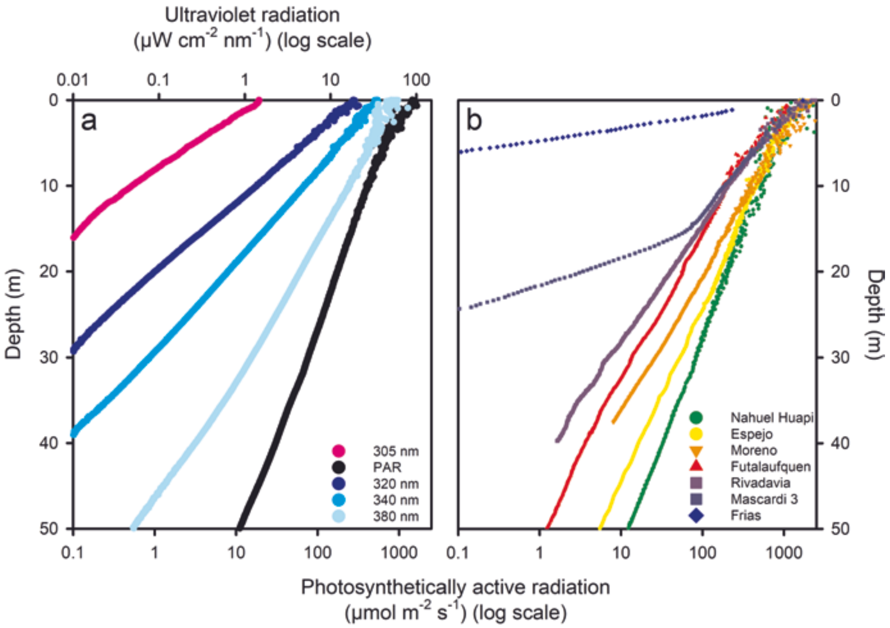

**Fig. 3.2** (**a**) Light vertical profiles in a North Patagonian Andean lake (Lake Nahuel Huapi), showing the transparency to several wavelengths, including ultraviolet radiation (UV-B, 305 nm and UV-A, 320, 340, and 380 nm) and photosynthetically active radiation (PAR, 400–700 nm). (**b**) Light vertical profiles of PAR of several lakes with different transparency. Less transparent lakes (Mascardi and Frías) are lakes with the input of inorganic particles (glacier clay). Mascardi 3 refers to Lake Mascardi sampling station number 3 (E3)

Vertical mixing can lead to a shortage of light if planktonic organisms are frequently dragged down to the deep dark layers, whereas stratification enhances light supply by decreasing mixing depth (Diehl 2002). This condition will imply that variation of the mixing depth affects the available mean light in the upper layer (Fig. 3.3). The available light in the mixing layer is defined as mean intensity in the mixolimnion ($I_m$) (Helbling et al. 1994; Kirk 1994) and changes in $I_m$ affected the dominance of different planktonic species (Modenutti et al. 2008).

As mentioned, glacial lakes frequently receive inputs of finely grounded rock particles of glacial origin; thus, upper glacial lakes often have a grey or whitish appearance. In contrast, the lower lakes in a series of glacial lakes can be blue because all particles have settled out of the water column. Early studies in Lake Mascardi (Bonetto et al. 1971) have shown that light was affected by glacial clay input, and Modenutti et al. (2000) showed a significant statistical relationship between the light extinction coefficient of PAR and total suspended solid concentrations. In particular, light extinction coefficients decreased steadily from the clay source (Upper Manso River) and along the Tronador Arm indicating an increase in transparency (Fig. 3.4). The GLOF event in May 2009 in Ventisquero Negro caused a significant decrease in turbidity over the years following the event (Fig. 3.4, black

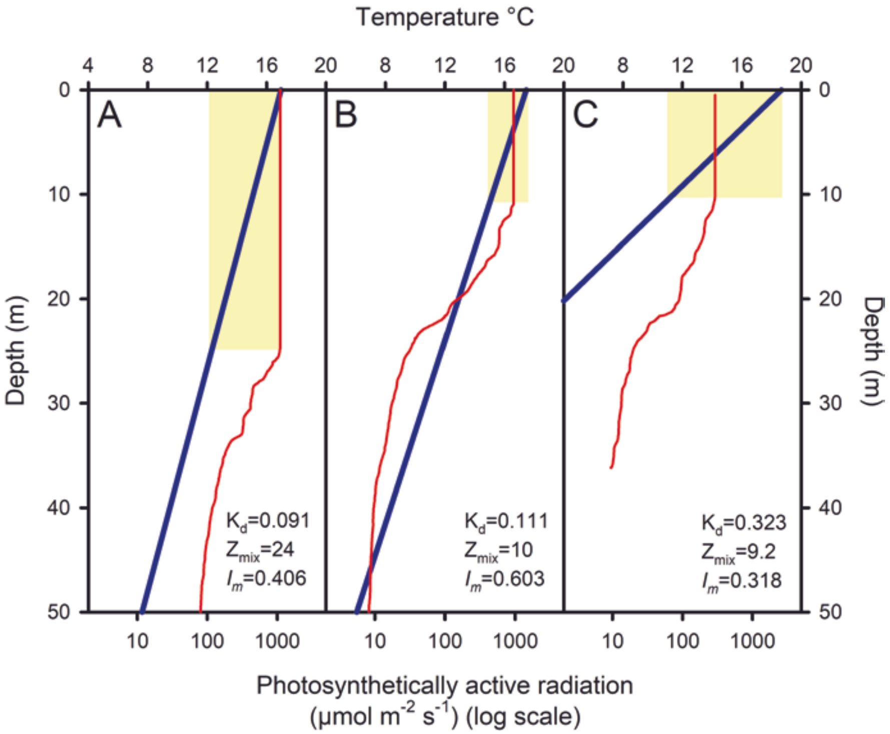

**Fig. 3.3** Light (blue) and temperature (red) vertical profiles of three different Andean Patagonian lakes. (**a**) and (**b**) have similar transparency (~Kd), but very different temperature profiles, (**b**) and (**c**) have similar temperature profiles, but (**c**) is less transparent. The yellow-shaded area represents the section averaged for the estimation of the irradiance of the mixed layer ($I_m$). Note how $I_m$ changes with the variation in the temperature profile

vs. red lines). Analysis of satellite images with SoPI (SRL 2018) revealed an acceleration in the increase of the lake area from 2.5 ha year$^{-1}$ (before GLOF, 1998–2009) to 3.5 ha year$^{-1}$ after the event (2010–2016). The increase in lake size increased the glacial clay sinking, and thus decreased the amount of suspended solids that were transported by the Upper Manso River downstream and into lake Mascardi Tronador Arm (Bastidas Navarro et al. 2018) (Fig. 3.5).

The eruption of the Puyehue-Cordón Caulle in 2011 discharged massive amounts of ash and pumice into the surrounding landscape in Argentina, producing an increase in total suspended solids (1.5 to 8-fold) in the surrounding lakes (Modenutti et al. 2013b). However, no differential specific absorption of the different light wavelengths occurred due to ash input, thus effects of volcanic particle inputs on the water column were comparable to those of glacial clay, decreasing light by increasing scattering, and thus light extinction coefficient. The result was a reduction of the euphotic zone during extended periods after the eruption occurred. Interestingly,

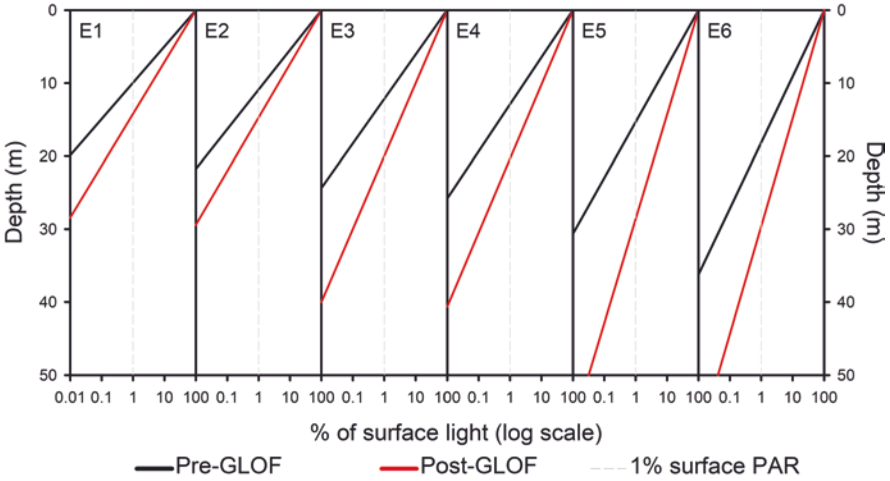

**Fig. 3.4** Light vertical profiles of photosynthetically active radiation (PAR) along the gradient of the Tronador arm of Lake Mascardi. Black lines correspond to data before the Glacier Lake Outburst Flood (GLOF) occurred in 2009 and red lines to data after the event. Note that the lake becomes more transparent from E1 to E6, before and after the GLOF

glacial clay or volcanic ashes did not affect DOC concentration, however, volcanic ashes increased phosphorus in the water column.

## 6 Deep Chlorophyll Maximum (DCM) as a Sensitive Variable

The high transparency (both to PAR and UVR, Fig. 3.2a) strongly reduces primary production and phytoplankton growth rates in the upper layers of the water column (Callieri et al. 2007). The net effect on the phytoplankton community is strong photoinhibition with DNA damage (Villafañe et al. 2004). These upper layers are inhabited by pigmented phytoplanktonic species that have high carotenoid content (Pérez et al. 2007). In addition, in many lakes, these upper layers are dominated by mixotrophic dark-pigmented ciliate species such as *Stentor araucanus* (Modenutti et al. 2005). The success of this species in highly illuminated layers has been associated to the pigment stentorin and the high concentration of mycosporine-like aminoacids (MAAs) that contribute to the high performance of these ciliates under UVR (Modenutti et al. 1998; Tartarotti et al. 2004). In particular, this species profits from high irradiances and is particularly abundant when the epilimnion mixing layer is reduced by calm wind weather. In this case, differences in $I_m$ with an increase in mean light in the upper levels favored the prevalence of the dark ciliate (Modenutti et al. 2008).

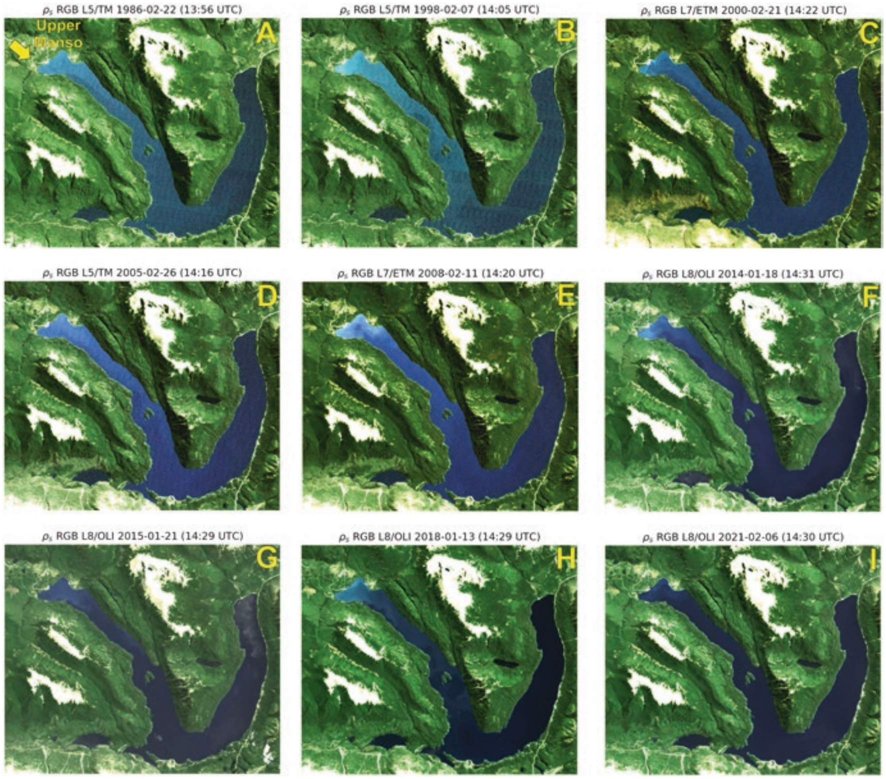

**Fig. 3.5** Satellite photographs of Lake Mascardi. RGB composites of the surface reflectance (ρs, Level-2) in all available wavelengths based on the atmospheric correction procedure using ACOLITE Python (https://github.com/acolite/acolite). Sensors used: Landsat 5 (L5/TM), Landsat 7 (L7/ETM), and Landsat 8 (L8/OLI). Date format on each map: year-month-day (hh:ss UTC). The sequence shows the change in reflectance (due to glacier clay) A-E: before (years 1986, 1998, 2000, 2005, and 2008), and F to I: after (2014, 2015, 2018, and 2021) the GLOF event

Because of the avoidance of high irradiances in the epilimnetic mixing layer, the distribution of phototrophic organisms throughout the water column results in the development of a meta-hypolimnetic deep chlorophyll maximum (DCM) located approximately at 1% of surface PAR (Pérez et al. 2002; Modenutti et al. 2004; Modenutti et al. 2013a). The colonization of these deep levels by phototrophic organisms represents a trade-off between higher survival (decrease of UVR effect) and lower cell-specific primary production (low irradiance) (Modenutti et al. 2004). Static primary production experiments, both *in situ* and in experimental system incubations, showed that bright light is a major factor reducing primary production (Callieri et al. 2007) and that is not compensated by the addition of nutrients (N and P) (Modenutti et al. 2013b). Although in lake experiments with light intensities below 10% of surface PAR (<200 μmol photon $m^{-2}$ $s^{-1}$), production increases significantly (Modenutti et al. 2004), this condition is achieved within the mixing layer where cells are dragged to higher damaging irradiances (Diehl 2002). Thus, in deep stratified lakes, primary production at these mixed levels remains low and increases only below the thermal gradient of the metalimnion. In fact, the development of DCM in North Patagonian Andean lakes, as in oceans and other deep lakes, has been associated with the existence of stable dim-illuminated metalimnetic layers (Kirk 1994; Sharples et al. 2001), and the variables that explained the DCM depth were the light extinction coefficients of PAR and 320 nm ($Kd_{320}$ and $Kd_{PAR}$) (Modenutti et al. 2013a).

The low DOC concentration observed in these lakes causes that the underwater wavelength-specific absorption depends mainly on the water itself, and thus the light at the DCM is dominated by green and blue wavelengths of the visible light spectrum (Pérez et al. 2002). For phototrophic organisms, this condition corresponds to the blue spectral niche where phycoerythrin and chlorophyll *b* are the accessory pigments better adapted (Stomp et al. 2007a, b; Holtrop et al. 2021). Indeed, at this level, the contribution of photosynthetic accessory pigments, such as fucoxanthin, phycoerythrin, and chlorophyll *b* increased (Pérez et al. 2007), and correspond to the observed increase in dinoflagellates, picocyanobacteria, and the *Chlorella*-bearing mixotrophic ciliate *Ophrydium naumanni* (Modenutti and Balseiro 2002; Modenutti et al. 2004; Callieri et al. 2007; Bastidas Navarro et al. 2018). Thus, at the DCM, different competition and predator-prey relationships co-occur. At these levels, autotrophs (both prokaryotes and eukaryotes) compete for light and nutrients, but at the same time, mixotrophs (i.e., nanoflagellates and ciliates) prey actively upon picocyanobacteria, other Eubacteria and Archaea (Modenutti et al. 2008). Furthermore, light is a decisive factor for clearance rate in ciliates and nanoflagellates. Thus, the vertical distribution of both predator and prey (picocyanobacteria) show a high coincidence (Modenutti and Balseiro 2002; Schenone et al. 2020).

The depth of the DCM appeared to be very sensitive to changes in light distribution. As mentioned for Lake Mascardi, the input of glacial clay produces light extinction coefficients to steadily decrease from the clay source (Upper Manso River) and along the Tronador Arm (Fig. 3.4). This light gradient causes, in turn, that DCM becomes deeper along the Tronador Arm (Hylander et al. 2011). As a consequence of the GLOF event in 2009 (Worni et al. 2012), the volume of the Lake Ventisquero Negro increased and the amount of suspended solids that were transported downstream into Lake Mascardi decreased. This condition resulted in a decrease in the light extinction coefficient (Fig. 3.4), and this higher transparency provoked the deepening of the DCM in the years after GLOF (Fig. 3.7) without changes in autotrophic species composition (Bastidas Navarro et al. 2018).

On the other hand, geological events such as volcanic eruptions can also increase the amount of suspended solids and decrease water transparency. During the Puyehue-Cordón Caulle eruption, the turbidity caused by ash fall triggered an upper position of DCM in the affected lakes. In Lake Espejo, the DCM moved upward from 40 m to 15 m depth (Fig. 3.6). However, ashes mainly composed of silica carried other elements as P, Fe, etc. (Caneiro et al. 2011) that changed the

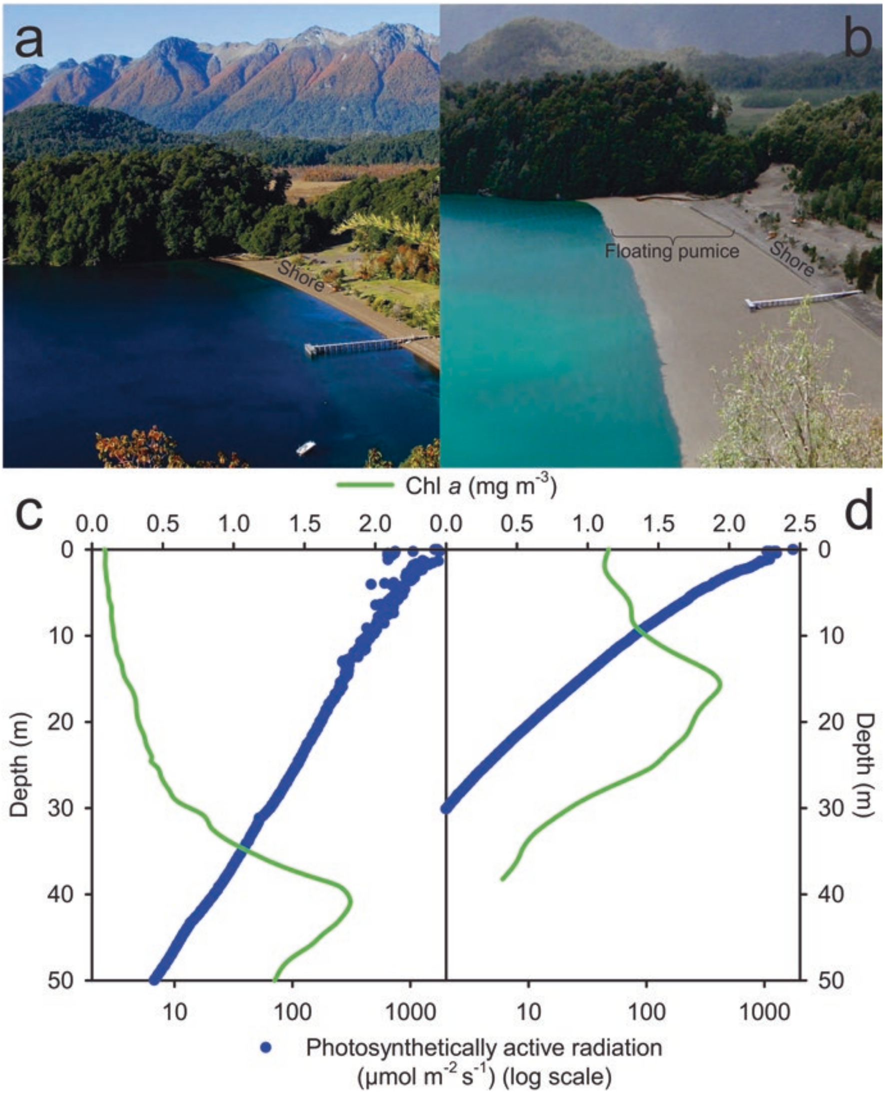

**Fig. 3.6** (**a**) Lake Espejo before and (**b**) 5 months after the eruption of Puyehue-Cordón Caulle volcanic complex. Note the change in the color of the lake water and the floating pumice near the lakeshore in (**b**). Lower panel: vertical light and chlorophyll *a* profiles: (**c**) before eruption and (**d**) 5 months after the eruption of Puyehue-Cordón Caulle volcanic complex

phytoplankton composition favoring diatoms such as *Tabellaria flocculosa* and *Aulacoseira granulata*.

Summarizing, both suspended particles, glacial clay, and volcanic ashes have shown a similar effect decreasing water transparency, and consequently, provoking an upper location of the DCM. Again, the analysis of environments with glacial clay

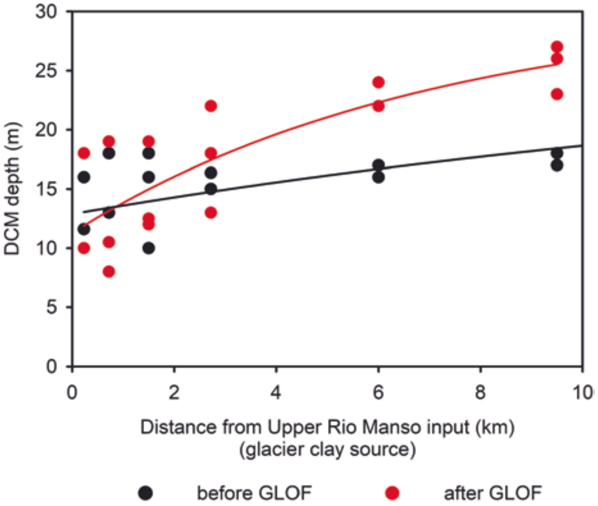

**Fig. 3.7** Changes in the depth of the deep chlorophyll maximum (DCM) along the transparency gradient of Lake Mascardi: black dots and line: before the Glacier Lake Outburst Flood (GLOF) event; red dots and lines: after GLOF event. Note the deepening of the DCM after the GLOF with the increase in transparency (see Fig. 3.4)

inputs (before and after GLOF event) and volcanic ash influence indicated that PAR and UV-A (320 nm) wavelengths have major importance in determining DCM depth (Modenutti et al. 2013a). These two catastrophic events provided unique opportunities to directly test the effect of changes in inorganic particle input and to relate different variables with previous lake conditions. These two events can be seen as natural experiments in which DCM was the most sensitive biological variable that quickly and accurately reacted to changes in light extinction.

## 7 Mixotrophic Nanoflagellates: Light and Feeding

Mixotrophy is a widespread strategy among phytoplankton in highly transparent environments, wherein mixotrophic nanoflagellates (MNF) account for up to 80–90% of total phytoplanktonic cells. MNF have a combination of different feeding strategies: while phagotrophy primarily provides nutrients and carbon for biosynthesis, photosynthesis provides carbon for both biosynthesis and respiration (Jones 2000; Berge et al. 2017; Hansson et al. 2019). In transparent lakes with high light and low nutrient concentration, small picoplanktonic cells (i.e., heterotrophic bacteria and picocyanobacteria) are favored since they are better competitors for P than large osmotrophic phytoplankton because of their higher surface/volume ratio (Gurung et al. 1999; Danger et al. 2007a, b). In this scenario, MNF have an advantage over strict osmotrophic algae by feeding on these P-rich picoplanktonic cells (i.e., bacterivory). Thus, MNF can obtain P by preying on competitors. MNF bacterivory is now recognized as a key pathway of energy and matter transfer through the planktonic food web (Mitra et al. 2014). Recent research has pointed out that mixotrophy is particularly successful in oligotrophic environments with high light-low nutrient conditions, whereas more productive, less illuminated systems would favor strict heterotrophic and phototrophic organisms (Fischer et al. 2017; Waibel et al. 2019). Phagotrophy by heterotrophic nanoflagellates (HNF) is negatively affected by suspended solids (Sommaruga and Kandolf 2014) while that of MNF has a more complex response because of their dependence on light (Schenone et al. 2020).

North Patagonian Andean deep lakes are very transparent, displaying different levels of turbidity due to glacial influence and volcanic activity, and both events produce an increase in suspended solids by the input of fine minerogenic solid particles affecting bacterivory. On the one hand, suspended solids reduce light penetration for phototrophic organisms for photosynthesis (Kirk 1994). Light has a non-monotonic effect on phytoplankton's phototrophy, where too low or too high light levels will reduce carbon uptake (Jassby and Platt 1976; Litchman 2003). On the other hand, bacterivory studies have suggested a simple and monotonic light dependence on phagotrophy of MNF (Jones 1997; Flynn and Mitra 2009), however, these studies evaluated a narrow range of light intensities. Interestingly, under a wider light range, Schenone et al. (2020) also found a non-monotonic response of phagotrophy to light in MNF. Based on this evidence and combining bacterivory experiments in a gradient of suspended clay with modelling, it was observed that MNF bacterivory is affected by turbidity and particle interference (Fig. 3.8) (Schenone et al. 2020).

In addition to the effect on light penetration, volcanic particles release P to the environment. This would imply less competition for P among osmotrophic cells and trigger the dominance of autotrophy over mixotrophy (Fischer et al. 2017). After the 2011 Puyehue-Cordón Caulle volcanic eruption, community-driven changes were observed in the dominance of MNF species in lakes with high suspended solids due to volcanic particles, from highly bacterivorous species (*Chrysochromulina parva*) (Gerea et al. 2019) to more phototrophic ones (*Plagioselmis lacustris*) (Modenutti et al. 2013b).

## 8 Changes in C:P and Crustacean Zooplankton Distribution

The crustacean zooplankton communities of the North Patagonian Andean lakes are dominated by calanoid copepods of the Family Centropagidae, in particular by *Boeckella gracilipes* and *B. michaelseni*, and among cladocerans, *Bosmina chilensis* and *B. longirostris* and daphnids, like *Ceriodaphnia dubia* and *Daphnia commutata* (Menu Marque and Marinone 1986; Modenutti et al. 2003; Balseiro et al. 2007; Balseiro et al. 2008). In several lakes at very deep layers, the presence of predaceous

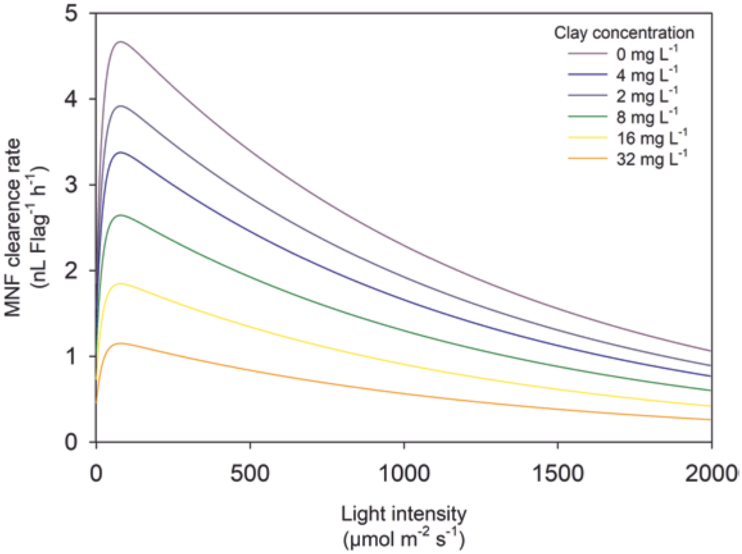

**Fig. 3.8** Results of the modeling of mixotrophic nanoflagellate bacterivory under a light and glacial clay gradient. The different curves represent the decay in bacterivory as glacier clay increases. For details in the model equations, see Schenone et al. (2020)

copepods, such as the calanoid *Parabroteas sarsi* and the cyclopoid *Mesocyclops araucanus,* was observed (Reissig et al. 2004; Modenutti et al. 2018b).

Crustacean zooplankton performance is commonly known to be affected by numerous environmental factors, such as fish predation pressure (Brooks and Dodson 1965), food particle size (Sommer 1989), and food quantity (Lampert 1977; Smith and Cooper 1982). Light penetration has a critical role in many features of oligotrophic aquatic ecosystems including zooplankton vertical distribution. In this sense, zooplankton evaded the higher level of the water column due to UVR and PAR+UVR (Alonso et al. 2004; Modenutti et al. 2018b). In addition, light can shape the composition of zooplanktonic communities through food quality. As stated by the "light:nutrient hypothesis" (LNH), the elemental carbon:phosphorus ratio (C:P) of primary producers is a result of the relation between light penetration in the water column and inorganic P availability (Sterner et al. 1997). Under high light intensities (PAR) and low levels of inorganic P, the nutrient limitation for phytoplankton becomes more severe, resulting in a disproportionate accumulation of C relative to P in algal tissue. Food quality has a strong influence on the fitness of herbivorous zooplankton (Sterner and Elser 2002). Hence, the variable C:P ratio in the phytoplankton, as a result of varying light intensities in the water column, represents a nutritional challenge for zooplankton. In turn, the elemental composition of zooplankton varies significantly among grazer species and taxonomic groups. For example, copepods have and require relatively low amounts of P, while cladocerans, in particular *Daphnia*, have higher P content and requirements (Andersen and Hessen 1991; Hall et al. 2004). As a consequence, the impact of different light penetration (Fig. 3.2b) in P limited systems varies for different zooplankton species according to their somatic P requirements, via the indirect effect of light as a modulator of food quality.

In glacier-influenced lakes inputs of finely ground rock particles cause a turbidity-induced attenuation of photosynthetically active radiation (PAR) and, as stated by the LNH, affect the nutrient composition of phytoplankton. Such shifts in the effects of glacial clay on the light environment can be observed among lakes (i.e., Lake Frías compared with other lakes, Fig. 3.2b), and within a single lake as glacial clay settles, moving away from the source input (i.e., Lake Mascardi). The northernmost end of the Lake Mascardi receives the Upper Manso River with a high load of glacial clay, generating a longitudinal gradient in light (Fig. 3.9). As predicted by the LNH, as light penetration increases with increasing distance from the clay input, also sestonic C:P increases, with a consequent decrease in food quality for grazers (Laspoumaderes et al. 2013; Laspoumaderes et al. 2017). Thus, there is a switch from the dominance of P-rich *Daphnia* in the turbid, low C:P area of the lake to the dominance of P-poor copepods as the distance from the clay input increases, turbidity decreases and sestonic C:P is higher (Fig. 3.9).

In addition to the better food quality in the turbid area of the lake, also ultraviolet radiation (UVR) is attenuated. Stoichiometric constraints (high food C:P ratio) affect *Daphnia*'s capacity to respond to the oxidative stress caused by UVR (Balseiro et al. 2008). This means that in the more transparent end of the gradient with higher P limitation, the fitness of *Daphnia* is reduced by its inability to cope with the effect of UVR. Although the copepods could suffer a similar effect of food quality on cellular defense (Souza et al. 2010), due to their lower P requirements, they experience a less severe stoichiometric imbalance than *Daphnia*.

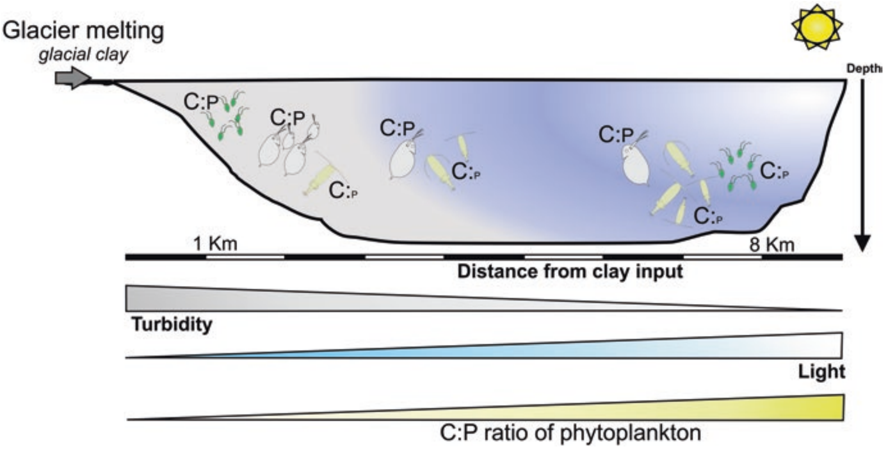

**Fig. 3.9** Schematic representation of the light gradient of the Tronador Arm of Lake Mascardi and zooplankton distribution. Note the change in the C:P ratio of phytoplankton and the concomitant change in the relative abundances of cladocerans and copepods along the gradient. Relative size of the C and P represent changes in C:P ratio

The ingestion of particles, such as suspended clay, in concentrations over 50 mg $L^{-1}$, is known to reduce fecundity and survival in daphnids (Kirk and Gilbert 1990), but the highest total suspended solids reported in Lake Mascardi were 5 mg $L^{-1}$. In this lake, *Daphnia* coexists with clay particles that have a net positive effect on their fitness (UVR protection, lower C:P food) over the negative effect that can pose a feeding interference. In an experimental clay gradient from 0 to 5 mg $L^{-1}$, Laspoumaderes et al. (2017) found that glacial clay alone is indeed a source of stress for *Daphnia* as it affects its enzymatic activities, which represents a feeding interference, and affects respiration rates. However, in the lake, this negative effect is overcome as clay decreases underwater light intensity (both PAR and UVR), which results in a better food quality and a decrease in UVR damage and visual fish predation.

Turbidity affects the foraging of visual planktivorous fishes (Vinyard and O'Brien 1976; Gregory and Northcote 1993; Utne 1997). In particular, members of the Galaxiidae family (*Galaxias maculatus* and *Aplochiton zebra*) are important zooplankton feeders in North Patagonian Andean deep lakes (Barriga et al. 2002; Lattuca et al. 2007). While *G. maculatus* is not affected by turbidity (Stuart-Smith et al. 2007), *A. zebra* is strongly affected (Jönsson et al. 2011). In addition, eye-size growth trajectories in *A. zebra* differ between turbid and clear water environments (McDowall and Pankhurst 2005; Lattuca et al. 2007). In Lake Mascardi, *A. zebra* foraging success decreases as an effect of impaired vision and increasing difficulty in finding prey when the water gets more turbid (Jönsson et al. 2011). Thus, changes in suspended solid may also affect other trophic links between zooplankton and fishes.

## 9 Volcanic Eruption and Cladoceran Disappearance

The eruption of the Puyehue-Cordón Caulle volcanic complex in 2011 discharged massive amounts of ash and pumice into the surrounding landscape in Argentina, producing an increase in total suspended solids in the surrounding lakes. Particle inputs from either glacial clay or volcanic ashes produce comparable effects on the water column, by generating a decrease in the underwater light penetration and in the depth of the DCM, and a consequent decrease in the C:P ratio of primary producers. Remarkably, the two natural phenomena that seem similar have had contrasting effects on the zooplankton communities.

The deposition of volcanic ash in the lakes affected by the eruption caused a decrease in light and a decrease in sestonic C:P, the same as observed with glacial clay in Lake Mascardi. Before the volcanic eruption, zooplankton communities in lakes Espejo, Correntoso, and Nahuel Huapi (affected by the eruption) were dominated by the copepod *Boeckella gracilipes* and *Ceriodaphnia dubia* as the dominant cladoceran, followed by *Bosmina chilensis* (Balseiro et al. 2007). However, in the summer following the eruption, Wolinski et al. (2013) observed an opposite shift in the zooplankton ensemble to the one observed in Lake Mascardi. Although ash inputs generated a decrease in light penetration and a consequent decrease in the phytoplankton C:P ratio (such as glacial clay in Lake Mascardi), cladocerans disappeared and copepod populations decreased, but the latter only in the lake with the highest ash concentration.

Volcanic particles are very similar in size to glacial clay (Modenutti et al. 2013a), but they have a crystalline structure, mainly composed of silica, which grants abrasive features (Caneiro et al. 2011). Therefore, filter feeders ingest ash as they do with clay, but the damaging effect of ash is much stronger and occurs at lower concentrations than that of clay. Ash was observed to be captured by *D. commutata* during the feeding process, resulting in a gut completely filled with ashes, a strong decrease in food uptake, and physical gut damage (Fig. 3.10) (Wolinski et al. 2013).

Regardless of the better food condition generated by the lower light penetration and the inorganic P provided by the ashes, cladocerans could not deal with the ingestion of volcanic ashes. However, copepods seemed to suffer less due to their ability to select motile cells (Balseiro et al. 2001). While 5 mg $L^{-1}$ of glacial clay is still a beneficial concentration promoting population growth in *Daphnia* (as reported for Lake Mascardi), ash concentration between 2 and 8 mg $L^{-1}$ was sufficiently harmful to affect the survival and reproduction of natural and experimental populations of cladocerans. Indeed, the experiments with 8 mg $L^{-1}$ of ashes caused the complete disappearance of *Daphnia* populations within 5 days (Wolinski et al. 2013). As the sedimentation process occurred, ash concentrations decreased, favoring population recovery, as observed in the following summer (18 months after the eruption). When the lakes recovered their original transparency with low total suspended solids values, the cladoceran populations also returned to their historical abundances.

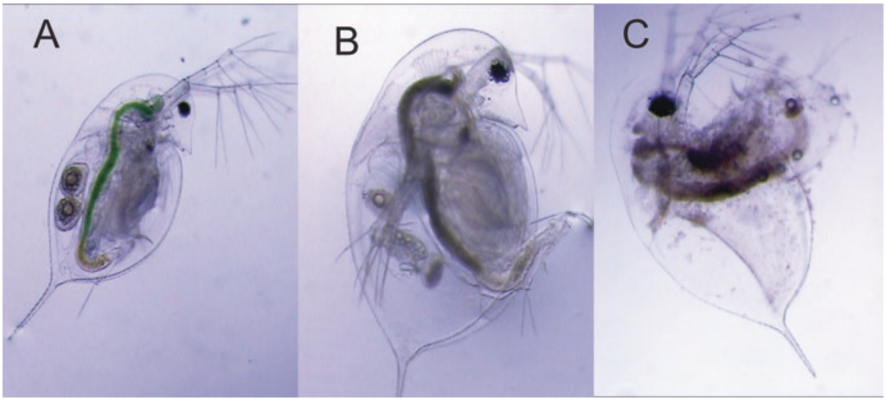

**Fig. 3.10** Microphotographs of *Daphnia commutata* exposed to volcanic ashes in concentration from 0 to 8 mg $L^{-1}$. (**a**) Without volcanic ashes, note the green color of the gut due to algae. (**b**, **c**) *D. commutata* exposed to increasing concentration of volcanic ashes ((**b**) 2 mg $L^{-1}$ and (**c**) 5 mg $L^{-1}$). Note the change in color of the gut in (**b**) that is filled with ashes and the intensive damage in (**c**) (dead *Daphnia*)

## 10 Concluding Remarks

Long-term studies monitoring North Patagonian Andean deep lakes allowed us to analyze how different events affect internal lake processes and eventually how resilient these lakes are to the input of inorganic particles and the consequent light changes. The most sensitive biological variable to these external factors is the location of the DCM in the water column. This is of particular interest as chlorophyll is a very easy-to-measure variable, both in the laboratory or directly in the field by a portable fluorometer. Thus, this variable represents an important tool for lake monitoring since the depth of the DCM reacted quickly and reliably to changes in light extinction. However, important changes can occur considering the composition of the DCM because species react differently to particle interference (as shown for nanoflagellates) (Schenone et al. 2020) and to the addition of Si and P (as shown by the diatom increase after the volcanic eruption) (Modenutti et al. 2013b). Future scenarios of climate change include differences in optical light climate both decreasing and increasing the diffuse extinction coefficient of lake water. Also, possible changes in land use with the increment of new roads and touristic center developments will affect the input of particles into the lakes with a possible loss of transparency. This will change the location of the DCM, and thus this parameter can be an invaluable tool for understanding lake dynamics and future production trends. Therefore, we encourage agencies to consider this parameter as a quick response of planktonic communities to changes in light conditions.

The relationship between light and nutrients is also determinant of the relative dominance of zooplankton taxa requiring different amounts of P in grazer assemblages and allows to predict shifts in grazer composition with changes in light and nutrient supplies (Elser et al. 2000; Hall et al. 2004; Laspoumaderes et al. 2013). How grazer assemblages react to changes in the light:nutrient environment does not only depend on their P requirements but also on the relationship between the type of particles that generate the light attenuation and their feeding strategy. Hence, the decrease in the performance of planktivorous fishes will affect the transference of secondary production to higher trophic levels, thus impacting the trophic web structure.

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# Chapter 4
# Current State and Recent Changes of Glaciers in the Patagonian Andes (~37 °S to 55 °S)

**Lucas Ruiz, Pierre Pitte, Andrés Rivera, Marius Schaefer, and Mariano H. Masiokas**

## 1 Introduction

The Patagonian Andes (also known as the Wet Andes) contain the largest glaciated area in the Southern Andes (Fig. 4.1) and the Southern Hemisphere outside Antarctica (Barcaza et al. 2017; Zalazar et al. 2020). As in other mountain regions of the world, the shrinkage of Patagonian glaciers is driven by climate change (Hock et al. 2019). Patagonian glaciers' retreat and thinning dominate the glacier mass change of the Southern Andes, which is among the most significant contributors to sea-level rise in recent decades (Hock et al. 2019).

The relative importance of the glacier melt contribution to surface runoff varies across the region but generally increases upstream. Glaciers store water in the form of snow and ice and release this water gradually but more actively during the warmer months. Thus, they can be considered autoregulated natural reservoirs, which become especially important water sources in drier seasons and years. The increased melt and retreat will probably trigger glacier hazards, such as glacier lake outburst floods (GLOFs), and a decrease in the stability of mountain slopes nearby retreating glaciers (Deline et al. 2015).

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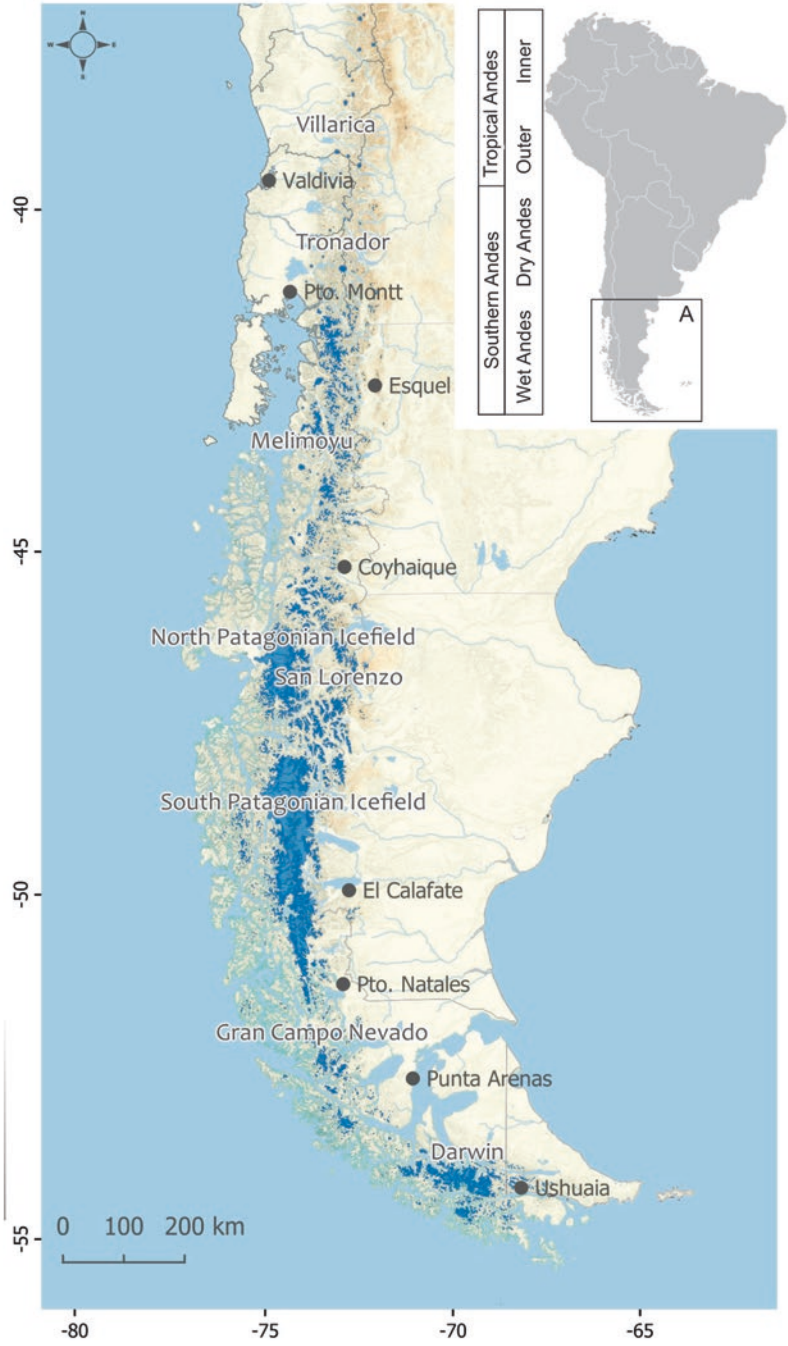

**Fig. 4.1** The Patagonian Andes. Inset shows the location of the Patagonian Andes or Wet Andes in the context of Southern and Tropical Andes. As a reference, the extent of the map is shown (A). Principal glaciers or glacier locations (in blue) discussed in this chapter are highlighted

Although included as one of the ice masses to be considered here, rock glaciers are not indeed glaciers in the sense of being a mass of ice originated by the recrystallization of snow. Rock glaciers are the morphological expression of creeping permafrost (soil or rock remaining below 0 °C for more than two consecutive years) (e.g., Barsch 1996; Berthling 2011). These features are much less frequent and cover a marginal surface compared to clean ice and debris-covered glaciers along the Patagonian Andes. Nevertheless, they have an important ice content and act as water reservoirs, which may play a role in the hydrological regime of specific basins, especially in the driest sectors of the study region. Rock glaciers can also affect water transit time, water chemistry, and discharge timing in these sectors (Giardino et al. 2011).

This chapter addresses the current distribution and characteristics of the Patagonian glaciers as well as their recent changes and hydrological implications. We summarize the recent findings in mass balance and ice dynamics along the Patagonian Andes, highlighting the processes behind glaciers' mass change and differential response to climate change. This chapter finally discusses recent findings about how glacier retreats will impact runoff and other glacier-related hazards.

## 2 The Patagonian Andes

The Patagonian Andes, also known as the Wet Andes (Lliboutry 1956), are located south of ca. 37 °S and contains several peaks with elevations between 3000 and 4000 m.a.s.l. (San Valentín 4058 m.a.s.l., the highest peak in Patagonia, is located in the Northern Patagonian Icefield) (Fig. 4.1). In this review, the Patagonian Andes are further divided in Northern, Southern and Fuegian Andes. The mean annual 0 °C isotherm decreases in elevation from about 3000 m in the Northern Patagonian Andes (at 37 °S) to less than 1000 m in the Fuegian Andes (at 55 °C) (Condom et al. 2007; Carrasco et al. 2008). The prevailing westerlies from the Pacific Ocean and their associated frontal precipitation constitute the main source of precipitation in this region. The north–south orientation of the Andean range creates a strong precipitation gradient, with markedly higher precipitation over the western slopes and drier conditions over the eastern ones (Garreaud et al. 2013; Viale et al. 2019). Some high peaks in the north Patagonian Andes can receive 3–5 m of precipitation per year (Schaefer et al. 2017), and further south, these values increase to 4–7 m on the Patagonian icefields (Lenaerts et al. 2014; Schaefer et al. 2015; Sauter 2020). In this region, precipitation seasonality is gradually reduced southward, with higher amounts concentrated during the winter months in the north, but a more regular precipitation regime throughout the year in Tierra del Fuego (Sagredo and Lowell 2012). The presence of numerous rivers, lakes, and extensive forest cover is also characteristic of the Patagonian Andes and reflects the overall humid conditions in this mountainous region (See Chaps. 1, 3, and 9). These topographic and climatological conditions also represent key factors controlling the size and distribution of glaciers along the Patagonian Andes, explaining, for example, the substantially larger glaciated surface in Patagonia compared with other Andean regions further north. The dominant westerly circulation of moisture also results in a much larger glacier surface area on the western slopes of the Patagonian Andes, which decreases drastically a few kilometers east of the main divides (Masiokas et al. 2015).

## 3 Glacier Distribution in the Patagonian Andes

A regular repetition of glacier inventories is crucial to describe their distribution, determine the importance of ice masses in the hydrologic cycle, and assess the ongoing ice mass changes at local and regional scales. Updating and improving the spatial resolution of glacier inventories is critical to outline environmental policies for glacier protection and monitoring programs and to develop mitigation and adaptation strategies in response to climate changes (Johansen et al. 2018).

A comprehensive understanding of the glacier distribution along the Southern Andes and their main characteristics was obtained through the pioneering work of Lliboutry (1956) and regional and local inventories (Aniya et al. 1996; Rivera et al. 2007; Paul and Mölg 2014; Masiokas et al. 2015). However, for many years, the lack of completed and quality-controlled glacier inventories for the Southern Andes impeded a detailed analysis of glacier distribution along the region and the inclusion of glacier data in local and regional studies. Recently, thanks to advances in remote sensing techniques, which have proven to be useful for glacier inventories worldwide (Raup et al. 2007; The Randolph Consortium et al. 2014), have initially allowed the development of national glacier inventories in Chile and Argentina (Barcaza et al. 2017; Zalazar et al. 2020). These national glacier inventories contain complete, updated, highly detailed and standardized information about ice masses in Southern Andes which are freely accessible online through dedicated web pages.

The glacier inventory of Chile (Barcaza et al. 2017) was based on 35 Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+) of medium spatial resolution (30 m) images between 2000 and 2003. The Argentinean glacier inventory (Zalazar et al. 2020) was based on 178 optical multispectral satellite images (Landsat and Aster) of medium spatial resolution (15–30 m) and 224 high spatial resolution (2–5 m) acquired between 2004 and 2016. Both inventories use a combination of multispectral semi-automatic classification methods to extract clean ice and perennial snowfields and manual digitalization to delineate debris-covered and rock glaciers, wherein the case of Argentina, it was achieved using a combination of medium to very-high spatial resolution images. In both cases (Chile and Argentina), different ice masses were morphologically characterized following the classification scheme of the World Glacier Monitoring Service (WGMS) and the Global Land Ice Measurements from Space (GLIMS) (Raup et al. 2007; Paul et al. 2010). Each ice mass was characterized according to different information, including general data (IDs, geographic location), morphometric details (area, elevation, slope, aspect, length) and information about the satellite imagery (sensor, date) used in the mapping process.

The Randolph Glacier Inventory version 6.0 (RGI 6.0; RGI Consortium 2017) is another glacier inventory used to assess the glacier mass change and glacier volume for the complete Southern Andes (Braun et al. 2019; Farinotti et al. 2019; Dussaillant et al. 2019). The RGI is a short-period (1–2 years) global inventory of glacier outlines released in 2012, with limited resources by a group of international glaciologists to serve the needs of the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (The Randolph Consortium et al. 2014). They aimed to achieve complete coverage rather than extensive documentary details and different sources of information, dates, and qualities were merged in one glacier inventory. Since its release, the RGI has been constantly improved by ingesting local to regional glacier inventories and has become a worldwide reference for glaciological studies (e.g., Farinotti et al. 2019; Zemp et al. 2019; Marzeion et al. 2020). The main advantage of RGI is its worldwide glacier coverage obtained with a systematic approach. However, for studies requiring more details at local scales, the RGI lacks enough resolution, includes many temporal snow areas, excludes debris-covered areas and, due to the automatic delineation of glacier areas, the glacier polygons outlines are highly fractal and, in many cases, are unrealistic. These kinds of discrepancies will be analyzed later in this chapter regarding Patagonian glaciers.

Considering the glacier inventories of Argentina and Chile, the Patagonian Andes contain about 24,074 ice masses covering a total area of 26,100 km$^2$, representing 60% of all the ice bodies and 88% of the ice-covered area of the Southern Andes (Barcaza et al. 2017; Zalazar et al. 2020). The mean elevation of ice masses in Patagonia is much lower than in the Central Andes further north. In the Patagonian Andes, the most numerous types of ice masses are perennial snow patches and glaciarets ($n$ = 14,215) (Fig. 4.2a), followed by mountain glaciers ($n$ = 8720) (Figs. 4.2c–f), rock glaciers ($n$ = 437) (Fig. 4.2b), outlet glaciers ($n$ = 368) (Fig. 4.3), and valley glaciers ($n$ = 334) (Fig. 4.4). Nevertheless, snow patches and glaciarets are substantially smaller and cover less than 511 km$^2$ in total, with rock glaciers covering an even smaller area (less than 26 km$^2$ for the whole study region). Outlet glaciers, which drain the inner areas of the Patagonian icefields, cover the most extensive area (15,400 km$^2$). Finally, small-to-medium-sized mountain glaciers and medium-to-large valley glaciers distributed all along the Patagonian Andes covered more than 7030 km$^2$ and 3125 km$^2$, respectively.

The ice masses in Patagonia are unevenly distributed (Fig. 4.5). Fewer, smaller ice masses exist in the Northern Patagonian Andes (37 °S to 45 °S) compared to those much larger and abundant that can be found further south. In the north, mountain and valley glaciers represent the most extensive ice masses and are usually found on isolated volcanoes and high peaks (Reinthaler et al. 2019). The elevated precipitation levels of the region, the relatively mild temperatures, and the steep average slopes of most glaciers in the region result in high ice mass turnover rates and ice velocities between tens to a few hundred meters per year in the steep areas (Ruiz et al. 2015). Following the strong west–east precipitation gradient, the largest glacierized area and number of glaciers and snow patches are in the west margin. The higher and smaller snow patches and rock glaciers are usually found on the eastern margin of the North Patagonian Andes. According to Condom et al. (2007),

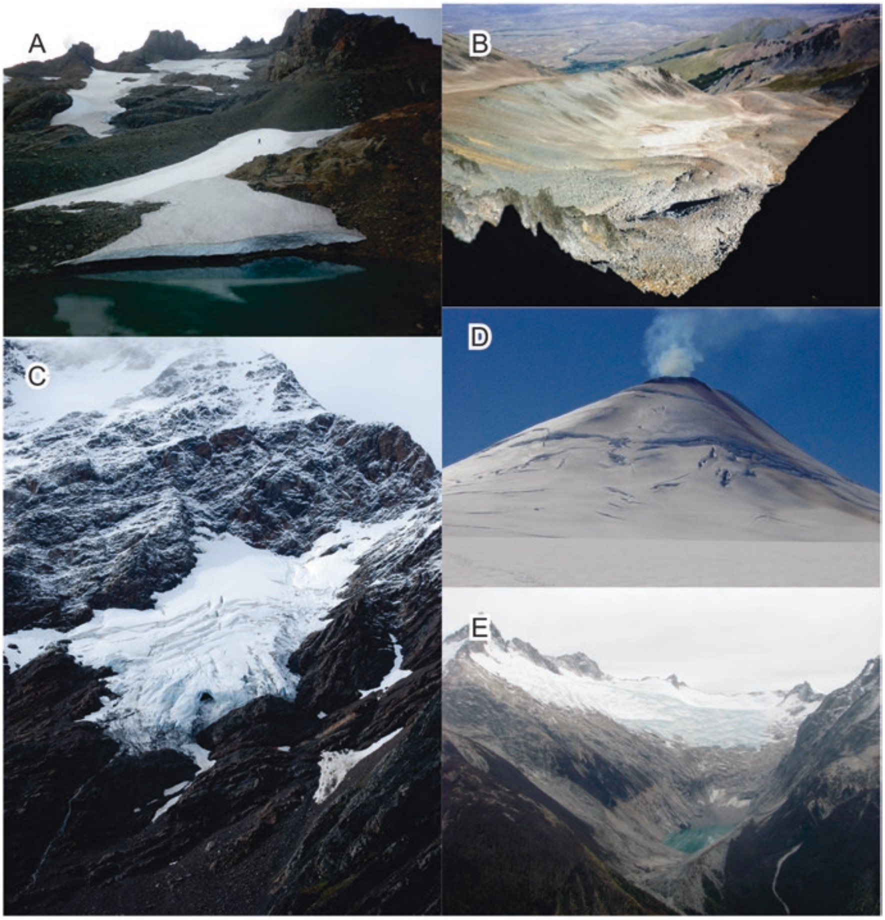

**Fig. 4.2** Examples of small ice mass in the Patagonian Andes. (**a**) Perennial snow patches or glaciarets in the Northern Patagonian Andes of Argentina. (Photo by Lucas Ruiz). (**b**) Small rock glacier in the Northern Patagonian Andes of Argentina. (Photo by Lucas Ruiz). (**c**) Small mountain glacier in the Southern Patagonian Andes of Argentina. (Photo by Pierre Pitte). (**d**) Villarica volcano in the Northern Patagonian Andes of Chile, which is covered by glacier ice cap. (Photo by Andrés Rivera). (**e**) Mountain cirque glacier in the Aysen region of Chile. (Photo by Andres Rivera)

the regional climatic Equilibrium Line Altitude (ELA) for glaciers is lower in the western margin and higher in the eastern one of the Patagonian Andes, where the development of mountain permafrost is favored (Ruiz and Trombotto 2012) (Fig. 4.6a).

The Southern Patagonian Andes (45° to 53 °S) contains the largest concentration of snow patches, mountain glaciers, outlet glaciers, and icefields, including the Northern Patagonian Icefield (NPI) and the Southern Patagonian Icefield (SPI) (Fig. 4.5). The SPI alone is formed by 139 glaciers larger than 5 km$^2$. This extensive icefield currently covers *ca* 12,200 km$^2$ (De Angelis 2014; Meier et al. 2018) and the

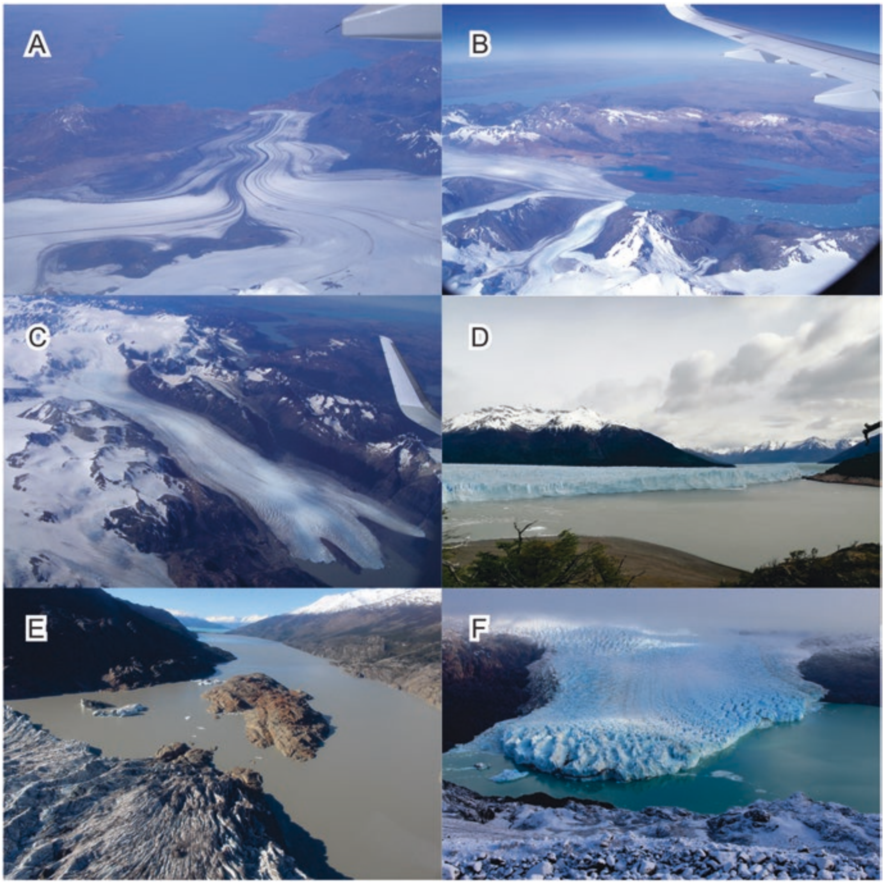

**Fig. 4.3** Examples of outlet glaciers in the Southern Patagonian Andes. (**a**) Viedma glacier, the largest glacier in Argentina flowing from the inner plateaus of the SPI to the Viedma lake. (Photo by Marius Schaefer). (**b**) Conjunction of Upsala and Codo glaciers of the SPI at Argentino lake. Upsala glacier has dramatically thinned and retreated in the last years. (Photo by Marius Schaefer). (**c**) Grey glacier in the SPI. (Photo by Marius Schaefer). (**d**) Front of Perito Moreno glacier as it approaches the Magallanes peninsula closing the channel between the lake Rico and lake Argentino. (Photo by Lucas Ruiz). (**e**) Front of Chico glacier in the northern margin of the SPI. (Photo by Andrés Rivera). (**f**) Front of O'Higgins glacier, one of the largest glaciers in the SPI as it calves into its homonymous lake. (Photo by Andrés Rivera)

NPI is *ca* 3700 km$^2$ (Dussaillant et al. 2018). Other important ice caps can be found at Mount San Lorenzo (*ca* 140 km$^2$; Falaschi et al. 2013) and Gran Campo Nevado (*ca* 200 km$^2$; Schneider et al. 2007b). The most significant amount and extent of ice masses is in the west margins of the Southern Patagonian Andes. The mean elevation of ice masses is lower than further north, and there is also an increase of ice masses elevation from west to east. Similar to the Northern Patagonian Andes, rock glaciers are mostly distributed over the eastern margins (Masiokas et al. 2015; Falaschi et al. 2015) below the regional climatic ELA (Condom et al. 2007) (Fig. 4.6b, c).

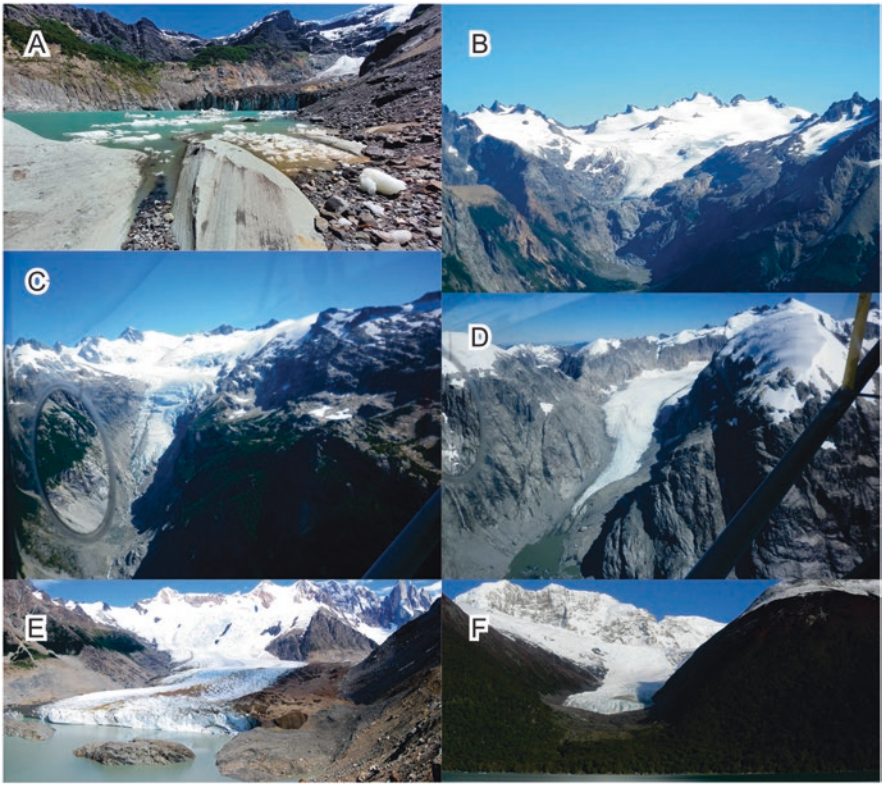

**Fig. 4.4** Examples of valley glaciers in the Patagonian Andes. (**a**) Manso glacier, also known as Ventisquero Negro, is the most voluminous glacier in the Monte Tronador, north Patagonian Andes of Argentina. (Photo by Lucas Ruiz). (**b**) Planchón Nevado valley glacier in the Northern Patagonian Andes of Argentina. (Photo by Lucas Ruiz). (**c**) Esperanza Norte valley glacier in the Northern Patagonian Andes of Argentina, it has one of the longest records of length fluctuation time series of the region. (Photo by Lucas Ruiz). (**d**) Turbio III valley glacier in the Northern Patagonian Andes of Argentina. (Photo by Lucas Ruiz). (**e**) Torre valley glacier in the Southern Patagonian Andes of Argentina. (Photo by Pierre Pitte). (**f**) Seco glacier, in the Southern Patagonian Andes of Argentina. (Photo by Lucas Ruiz)

Further south, in the Fuegian Andes (53 °S to 55 °S), the trend of decreasing ice masses mean elevation continues. The largest ice-covered is Cordillera Darwin icefield (CDI) (*ca* 2300 km$^2$; Bown et al. 2014), although other smaller icefields with outlet glaciers larger than 100 km$^2$ and many mountain glaciers also exist at Isla Santa Inés, Cordón Navarro, and Isla Hoste in the archipelagos of the western margin of Tierra del Fuego (Fig. 4.5). In the eastern margin of the Andes, small mountain glaciers, glaciarets and rock glaciers are developed. In the same way as further

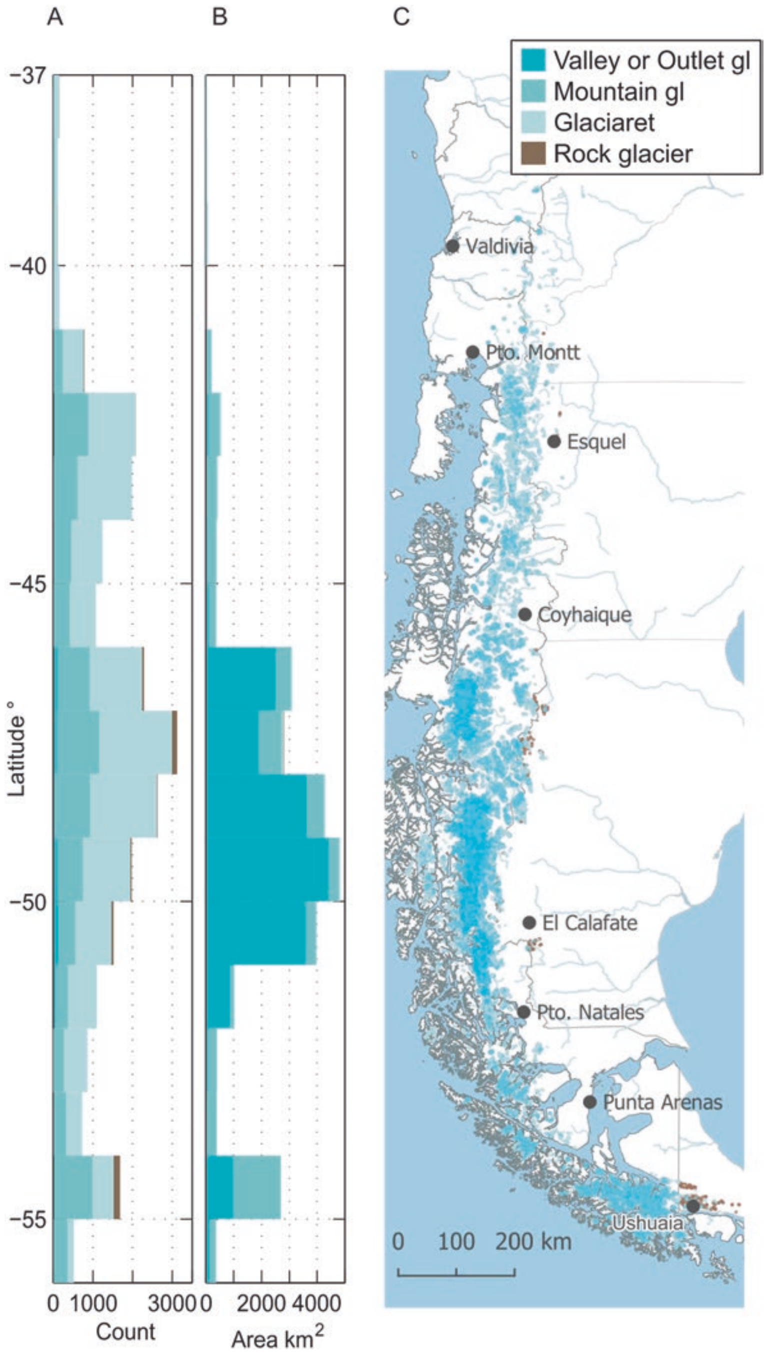

**Fig. 4.5** Distribution of glaciers in the Patagonian Andes. Both number (**a**) and extent (**b**) by glacier type and by 1° of latitude are shown to highlight the uneven distribution of ice masses. (**c**) Map of distribution of glaciers in Patagonia. (Data from Barcaza et al. 2017 and Zalazar et al. 2020)

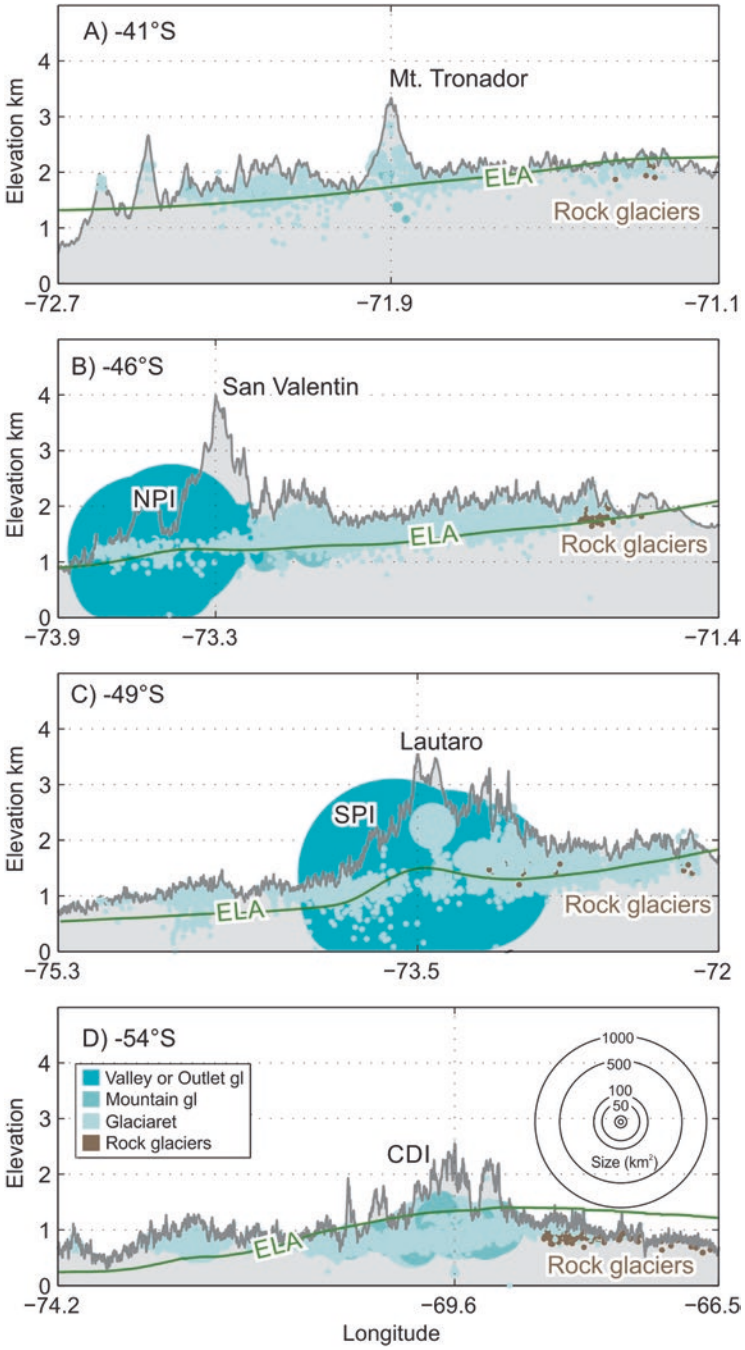

**Fig. 4.6** Across Patagonian Andes transect at selected representative latitudes. Each transect represents the size and mean elevation of ice masses at 2° latitude bands. (**a**) Northern Patagonian Andes at the latitude of Monte Tronador. (**b**) Southern Patagonian Andes at the latitude of the Northern Patagonian Icefield (NPI). (**c**) Southern Patagonian Andes at the latitude of the Southern Patagonian Icefield (SPI). (**d**) Fuegian Andes at the latitude of the Cordillera Darwin icefield (CDI). At each transect, the Regional Climate Equilibrium Line Altitude (ELA) from Condom et al. (2007) is shown

north, rock glaciers are located below the Regional Climate ELA (Condom et al. 2007; Fig. 4.6d).

Although glaciers along this region are mainly clean ice or debris-free (98% of the glaciated area), debris-covered glaciers can still be found due to local conditions such as rock-fall and stagnation. This is the case of valley glaciers at Monte Tronador (41 °S) or at Monte San Lorenzo (47 °S), where rock-falls and avalanches below massive bedrock cliffs or headwalls allow the concentration of debris over the glacier tongues (Falaschi et al. 2013; Ruiz et al. 2017).

When the RGI 6.0 (RGI Consortium 2017) is examined against the glacier inventories of Argentina and Chile, differences can be noted, which need to be considered to put in perspective other studies that use it as input data (Zalazar et al. 2020). The most significant discrepancy occurs in the number and extent of small ice masses (<0.5 km$^2$). Although the RGI 6.0 shows similar latitudinal patterns regarding the number and extent of ice masses, substantial local discrepancies can also be observed between these datasets. The largest differences are found in the Northern Patagonian Andes, where the RGI overestimates the number (>150%) and extent (>100%) of glaciers by more than double of the glacier inventories of Argentina and Chile. However, the difference is much lower in the Southern Patagonian Andes and Tierra del Fuego (<10% in the number of glaciers and <7% in their extent). The difference in the number of glaciers could be attributed to the automatic method used to delimit glaciers and the quality of digital elevation models used as input. Also, the overestimation in the extent of glaciers could be due to inventory misclassified seasonal snow patches as, or part of, perennial ice bodies. As we show below, caution must be taken when considering the results of ice mass distribution or glacier mass change from studies based on the RGI 6.0 or previous versions in the Northern Patagonian Andes.

## 4 Ice Volume in the Patagonian Andes

Glacier ice volume is key to quantifying water resources in mountain regions and their possible contribution to sea-level rise (Farinotti et al. 2017). There are several methods for inferring the total volume of glaciers: volume–area scaling approaches (Bahr et al. 1997), parameterization schemes (Haeberli and Hoelzle 1995) and physical models based on ice-flow dynamics and mass conservation (Farinotti et al. 2009; Morlighem et al. 2011; Gantayat et al. 2014). Recently, there has been an increase in the number of studies using various numerical inversion approaches to recover ice thickness distribution from surface measurements and glacier characteristics (Farinotti et al. 2017). In volume-area scaling approaches, where ice volume is calculated based on an empirical relationship to the glacier area, larger glaciers tend to be thicker (Bahr et al. 2015). Although volume-area scaling approaches are easy to apply, they are potentially misleading if they are not calibrated with representative data. Also, distributed approaches tend to have significant uncertainties and are calibrated to the limited number of existing thickness measurements. Ice thickness measurements along the Southern Andes are scarce, and data are available only for a limited number of glaciers along the Patagonian Andes (GlaThiDa Consortium 2019). First, ice thickness measurements during the 1980s and 1990s were retrieved using gravity or seismic methods in the Northern and Southern Patagonian Icefields by Casassa (1987) and Rott et al. (1998). Rivera et al. (2001) present the first radio-echo sounding ice thickness measurements over the Casa Pangue glacier in Monte Tronador.

Meanwhile, Rivera and Casassa (2002) obtained the first ice thickness measurements in the accumulation area of the Southern Patagonia Icefield using a radio-echo sounding system. The Chilean Water Cadaster (Dirección General de Aguas) conducted a series of radar surveys onboard helicopters on the Patagonian Andes' Chilean side with more comprehensive coverage of the ice thickness for Monte Tronador and the Northern and Southern Patagonian Icefields (Dirección General de Aguas 2014). Zamora et al. (2017) present an airborne low-frequency radar system used to survey the ice thickness of the northern part of the Southern Patagonian Icefield. More recently, Gourlet et al. (2016) and Millan et al. (2019) present the most extending mapping of ice thickness and bed elevation measurements over NPI and SPI by combining airborne gravity data, airborne radar sounder, and lake and fjord bathymetry.

Glacier ice volume or ice thickness distribution has been estimated for the Patagonian Andes by different modeling approaches. Carrivick et al. (2016) derive a first-order estimate of ice thickness and volume for 617 glaciers between 41 °S and 55 °S, inventoried by Davies and Glasser (2012). Farinotti et al. (2019) used different inversion models to recover the ice thickness distribution of all the glaciers in the Randolph Glacier Inventory (RGI) 6.0 (RGI Consortium 2017). Carrivick et al. (2016) estimate that glaciers in the Patagonian Andes, south of 41 °S ($n = 627$; area = 23,475 km$^2$), is 5955 km$^3$ ± 1191 km$^3$ of ice. The consensus estimate for glacier volume for Patagonian Andes ($n = 12{,}712$; area = 27,219 km$^2$) obtained by Farinotti et al. (2019) is 5217 ± 1300 km$^3$ of ice. The amount of water storage in the Patagonian glaciers is so large that it is equivalent to a rise of 15 ± 5 mm in the global mean sea level or three times the volume of all the proglacial lakes in the Central and Patagonian Andes.

Due to the relationship between glacier size and volume, most of the glacier ice location is highly skewed to the western margin of Southern Patagonian Andes, where the largest glaciers are found (Fig. 4.7). Indeed, maximum ice thickness was found in some of the larger glaciers of the SPI, but there is still no agreement on the exact location and magnitude of the highest values in the icefield. For example, Millan et al. (2019) reported maximum ice thickness measurement values of 1650 ± 60 m in the accumulation area of Bernardo glacier (SPI), whereas maximum ice thicknesses modeled by Carrivick et al. (2016) reach 1631 m ± 179 m in the ablation area of Viedma glacier (SPI). In contrast, for Farinotti et al. (2019), modeled maximum ice thickness for this area did not exceed 900 m, with the highest values of 890 ± 230 m in the accumulation area of Upsala glacier (SPI). Millan et al. (2019) calculated a total ice volume of 1124 ± 260 km$^3$ for the NPI and 3632 ± 675 km$^3$ for the SPI, with a combined total of 4756 ± 935 km$^3$. The total

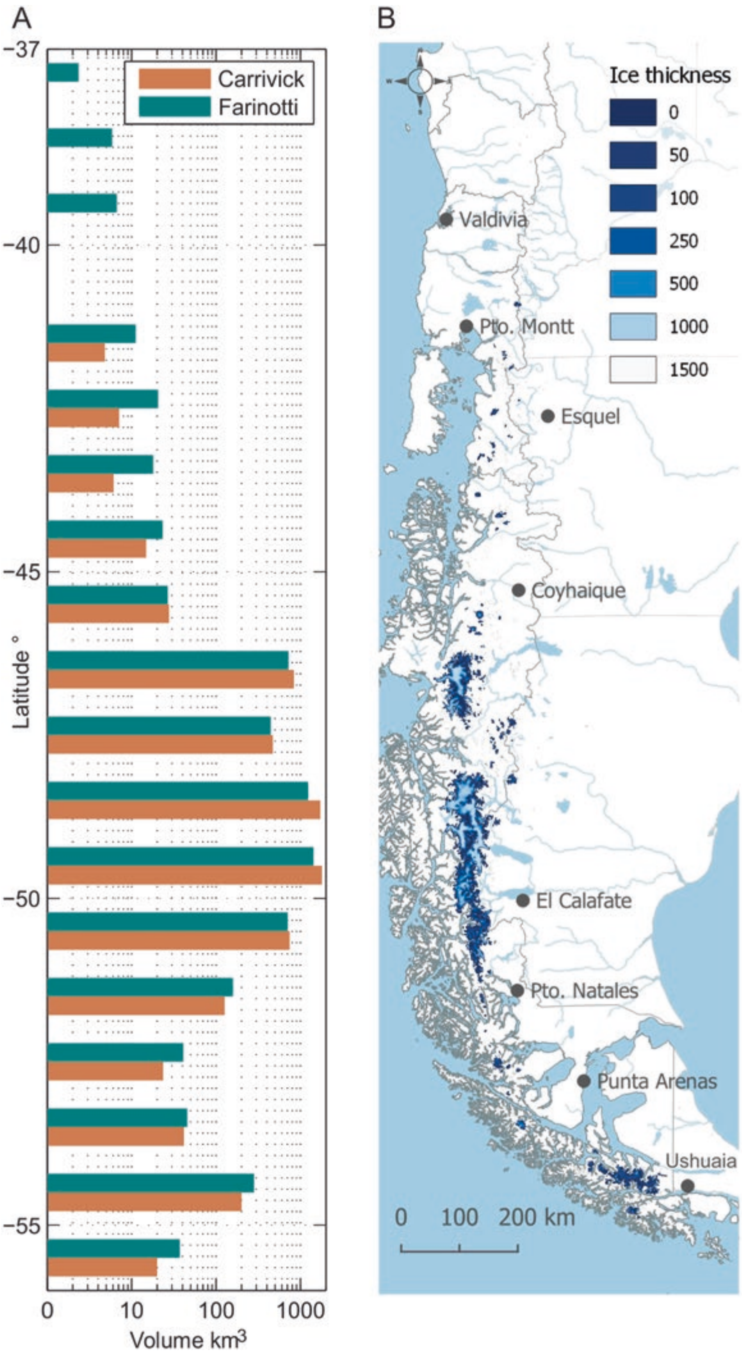

**Fig. 4.7** Glacier ice volume distribution along the Patagonian Andes. (**a**) Ice volume by 1° of latitude from Carrivick et al. (2016) and Farinotti et al. (2019). Due to the uneven distribution and thickness of glaciers, the scale of the plot is logarithmic. Although Carrivick et al. (2016) only considered 617 ice masses due to their size, it captures the general distribution pattern south of the 45 °S, where the largest ice mass is located. Meanwhile in the Northern Patagonian Andes, they underestimate the ice volume. (**b**) Ice thickness map of Patagonian glaciers from Farinotti et al. (2019)

modeled ice volume by Carrivick et al. (2016) was 1234 ± 246 km$^3$ for the NPI and 4326 ± 865 km$^3$ for the SPI. In contrast, Farinotti et al. (2019) estimated a total volume of 4607 ± 1200 km$^3$ for the NPI and SPI together. Although all values agree within their error bars, there are significant differences in the ice thickness distributions. For example, Farinotti et al. (2019) tend to miss some significant topographic features and underestimate some of the deepest glacial valleys and along the plateaus of the icefields.

Further north, Zorzut et al. (2020) performed an independent evaluation of Carrivick et al. (2016) and Farinotti et al. (2019) results for Monte Tronador glaciers against radio-echo sounding measurements and an ice surface velocity inversion model. Monte Tronador's glaciers ice volume best estimates was 4.8 ± 2 km$^3$. However, both Farinotti et al. (2019) and Carrivick et al. (2016) suggested thinner ice (4.3 ± 2 km$^3$ and 2.6 ± 3 km$^3$, respectively). Although the difference in ice thickness estimation between models' results could be attributed to the lack of ice thickness measurements to calibrate Farinotti et al. (2019) and Carrivick et al. (2016) models' outputs, a detailed analysis of their ice thickness distribution maps reveals that glacier inventory used as input data was a significant source of discrepancies.

Differences in mapping and identifying individual glacier basins impact the distribution of ice thickness, especially at ice divides within the central plateaus of the icefields. On the other hand, ice thickness is overestimated where internal outcrops are not well depicted. Discrepancies in the ice thickness distribution have important implications for future projections of ice volume change. Particularly, if thinner and smaller glaciers are depicted for a region (as suggested, e.g., in the analyses of Farinotti et al. (2019)), this will amplify the implications of possible future impacts of climate changes on these glaciers.

## 5 Glacier Changes in the Patagonian Andes

There are different ways to quantify glacier changes, which nowadays mostly rely on remote-sensing techniques. Changes in length or extent (area) of glaciers could be observed on the satellite images or through the simple comparison of historical records (Zemp et al. 2011). However, these changes cannot directly be related to climatic forcing. The response of glaciers is also affected by topographic features (e.g., glacier size, glacier surface slope, glacier bedrock slope), surroundings characteristics (e.g., proglacial lake, type of bedrock sediment), supraglacial features (debris-cover, ponds, or ice cliffs), subglacial hydrology and calving dynamics (in the case tidewater- or lake-terminating glaciers) can play an important role. As a result, changes in glacier size through advance or retreat are an indirect, delayed, and filtered signal to changes in climate (Cuffey and Paterson 2010).

In contrast, the glacier mass changes or mass balance (i.e., the annual change in volume or mass) is a direct and undelayed response to the annual changes in atmospheric conditions (Haeberli and Hoelzle 1995; Cogley et al. 2011). Glacier mass changes can be measured in the field and by remote sensing. In situ measurements consist of ablation measurements with ablation stakes and accumulation measurements using, most commonly, snow pits (Cogley et al. 2011). These in situ measurements are then extrapolated to the entire glacier and provide detailed information but are laborious, and thus only be obtained for relatively few glaciers. Some process such as the loss of mass at the calving front of a lake- or marine-terminating glaciers, which consists of the mechanical breaking off of, sometimes large, chunks of ice (but also includes subaerial melting and subaerial sublimation, and very importantly subaqueous frontal melting), cannot be captured by the traditional glaciological mass balance methods of "stakes and pits." Thus, it is expected that for those glaciers with high active calving fronts, mass changes are different from the surface mass balance. Another widely used approach to estimate glacier mass changes is the geodetic method, where the difference in surface elevation measured at two or more points in time is associated with glacier mass change (Cogley et al. 2011). Significant advantages of this include the possibility that large regions can be investigated simultaneously and that part of the frontal ablation in a calving glacier (the ablation that occurs above the surface of the water body) is also captured. This method is constrained by the availability of suitable elevation data or imagery (Bamber and Rivera 2007; Cogley 2009), but nowadays, many different sources of elevation data are available. Modeling provides another avenue for examining glacier mass change. There are different ways to parameterize the interactions between climate and glacier, which determine the glacier's mass balance ranging from purely empirical parametrizations (such as the degree-day method) to parametrizations that try to quantify the physical processes that take place at the glacier–atmosphere interface. The general performance of these parametrizations crucially depends on the quality and availability of meteorological data (Hock 2003). An important point to note is that adequate parametrizations of glacier–climate interactions are necessary to project the future of glaciers under different climate change scenarios (Marzeion et al. 2020).

## 5.1 Glacier Area Changes

Our knowledge of glacier changes in the Patagonian Andes has increased substantially in recent years, and there is overwhelming evidence for the ongoing, generalized ice mass loss throughout this region. Glacier fluctuation data from the late seventeenth century to the present show that despite several readvances over the past 100–110 years, most of them have been shrinking since the European period called Little Ice Age (LIA) when Patagonian glaciers reached their latest Late Holocene maximum position (García et al. 2020). The date assigned to this maximum advance varies among Andean regions, but it is assumed for the Patagonian Andes that it took place between AD 1650 and 1870 (Masiokas et al. 2009; Glasser et al. 2011), when glaciers advance generated of conspicuous moraines and trimlines through this mountain range. Davies and Glasser (2012) assessed glacier area changes since 1870 for 640 glaciers from 40 °S to 56 °S, finding that between 1870 and 2011, 90% of the glaciers shrank, 0.3% advanced, and 9.5% showed no change. The total area of these studied glaciers covered an area of 26,848.8 km$^2$ in 1870, which was reduced by 15.5% to 2011, when they only had 22,717.5 km$^2$. The overall rate of area loss was twice as rapid for the 2001–2011 period (51.2 km$^2$ yr$^{-1}$) than for the 1870–1986 interval (26.8 km$^2$ yr$^{-1}$), particularly for the NPI and SPI (Davies and Glasser 2012). More recently, Meier et al. (2018) assessed the glacier change between 1870 and 2016 for the Southern Patagonian and Fuegian Andes, considering more than 11,200 glaciers covering an area of 28,091 ± 890 km$^2$ during the LIA. They found a total reduction of 5455 ± 1269 km$^2$ (19 ± 5%), which represents a rate of area reduction of 37 ± 9 km$^2$ yr$^{-1}$ between 1870 and 2016 increasing to 73 ± 42 km$^2$ yr$^{-1}$ between 1986 and 2016. In the Fuegian Andes, the highest rate of change occurred during 1986–2005 and decreased afterward. Both of the studies cited above agree that small glaciers (<5 km$^2$) have the highest relative rate of shrinkage, with large lake or maritime-terminating glaciers contributing most to the total area loss.

There are few exceptions to this general glacier mass loss in the Patagonian region. The most remarkable is the almost continuous advance of Pio XI or Brüggen in the western side of the SPI, which destroyed more than 400 years old trees (Rivera et al. 1997) and formed a prominent frontal moraine thanks to a net advance of 11 km since 1945 (Wilson et al. 2016; Hata and Sugiyama 2021). Perito Moreno glacier in the eastern margin of the SPI also shows an unusual behavior, with only minor fluctuations of the glacier front for about 80 years (Minowa et al. 2015). Nevertheless, most of the SPI glaciers have been retreating (Stuefer et al. 2007; Davies and Glasser 2012). Another glacier that has advanced recently is the Garibaldi glacier in CDI of the Fuegian Andes, which shows an oscillating, quasi-stable front since at least 1945 (Melkonian et al. 2013).

In this context, it is essential to note that the advance or retreat of maritime and lake-terminating glaciers depends, among other factors, on the underlying topography (Benn et al. 2007). Although surge-like behavior has been suggested to explain the advance of Pio XI (Wilson et al. 2016), recently, it has also been stated that sediment deposition in front of the glaciers and the concomitant decrease in water depth may be controlling the glacier dynamics, with a possible influence of recently increasing precipitation (Hata and Sugiyama 2021). The presence of a submarine terminal moraine or a shallower fjord was also proposed to explain the behavior of the Garibaldi glacier (Melkonian et al. 2013). Stuefer et al. (2007) indicate that the shallowing due to the presence of the Magallanes peninsula at the front of the Perito Moreno glacier, together with the hypsometry of its basin (a large accumulation area and a high slope where ELA is located), may explain the oscillatory behavior (small advances and retreats) of this glacier.

Although supraglacial debris cover is not widespread in the Patagonian glaciers, mapping the spatial and temporal changes in debris cover shows that it has increased over time. In the NPI, Glasser et al. (2016) found that debris cover had increased from 4% in 1987 to 8% in 2015. Similarly, in Monte San Lorenzo, Falaschi et al. (2021) found debris-covered increases from 40% to 50% between 1958 and 2020. Although not widely assessed for the rest of the Patagonian Andes, when present, the debris-covered area is increasing in the Patagonian Andes, like in other regions of the Andes (Malmros et al. 2016; Ferri et al. 2020).

## 5.2 *Glacier Mass Changes*

Direct glaciological mass balance observations are limited to a handful of short data series in the Patagonian Andes. Martial Este glacier (0.1 km$^2$) in Tierra del Fuego, with continuous glaciological measurements since 2000, has the longest record of annual and seasonal mass balance in Patagonia, followed by Mocho Choshuenco glacier (5.1 km$^2$) with discontinuous measurements since 2003 (Rivera et al. 2005; Schaefer et al. 2017). Other glaciers in the Patagonian Andes where glaciological monitoring programs started or restarted in 2013 or were sporadically or locally without retrieving the glacier-wide mass balance have been conducted as an insight of the amount of accumulation and ablation (Stuefer et al. 2007; Bravo et al. 2019a, b). Thus, besides their shorter or sporadic records, they help assess the quality of geodetic mass balance measurements or calibrate and validate mass balance modeling approaches (Schaefer et al. 2015).

Different regional to local geodetic mass balance surveys exist for the Patagonian Andes. Two Andes-wide geodetic mass balance studies have comprehensively estimated glacier change for the last two decades along the Patagonian Andes (Braun et al. 2019; Dussaillant et al. 2019). Dussaillant et al. (2019) assessed the mass change of Andean glaciers in two sub-periods, finding that in the Northern Patagonian Andes, ice mass loss rate increased from 0.08 ± 0.25 m w.e. a$^{-1}$ to 0.98 ± 0.25 m w.e. a$^{-1}$ between 2001–2008 and 2010–2017, respectively. Further south, in the Southern Patagonian Andes, the ice mass loss remained high but relatively stable between the two time periods: It changed from 0.85 ± 0.3 m w.e. a$^{-1}$ to 0.76 ± 0.3 m w.e.a$^{-1}$ between 2000–2008 and 2010–2017, respectively. In contrast, the ice mass loss at the Fuegian Andes was 0.76 ± 0.3 m w.e. a$^{-1}$ during 2000–2006 but decreased to 0.35 ± 0.4 m w.e. a$^{-1}$ between 2012 and 2016 (Dussaillant et al. 2019) (Fig. 4.8).

## 5.3 *Causes of Glacier Mass Change*

Mass changes of glaciers, which do not terminate in deep water, such as the majority of ice mass in north Patagonian Andes, depend on the balance between the snow accumulation and glacier surface ablation (Cogley et al. 2011). Glacier surface ablation is driven by the energy fluxes between the glacier surface and the atmosphere. Due to the mild temperature conditions and the rather high wind speeds in the Patagonian Andes, the turbulent flux of sensible heat was highlighted as the most important energy provider to the glacier surface, followed by the net solar radiation (Schneider et al. 2007a, Schaefer et al. 2020). The importance of the turbulent flux

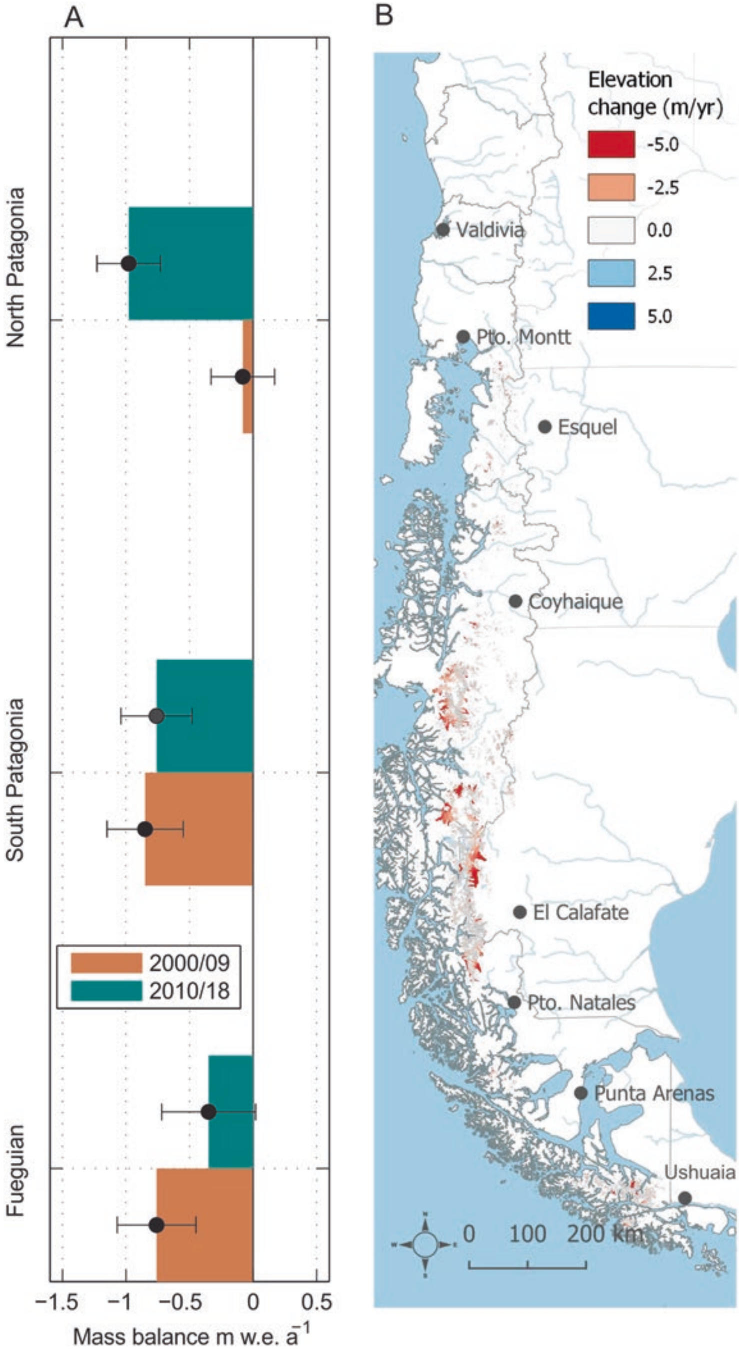

**Fig. 4.8** Glacier mass change in the Patagonian Andes. (**a**) Glacier mass change for the different regions of the Patagonian Andes for two sub-periods (2000–2009 and 2010–2017), from Dussaillant et al. (2019). (**b**) Elevation changes map of the Patagonian glaciers from Dussaillant et al. (2019)

of sensible heat to the glacier surface explains the dependency of the modeled melt rates on the air temperature in the Patagonian Andes as opposed to the Central Andes where the melt rates are rather determined by radiative fluxes (Schaefer et al. 2020). On the contrary, the turbulent flux of latent heat was found to be either positive or negative, depending on the humidity conditions, and the net longwave radiative energy flux which is normally slightly negative. Interestingly, it was found that the days with the highest melt rates in the Patagonian Andes are associated with overcast conditions, where the turbulent flux of latent heat is positive, the turbulent flux of sensible heat is high and the net longwave radiative energy flux is close to zero (Schaefer et al. 2020).

The almost balanced mass budget of Monte Tronador glaciers between 2000 and 2012 suggests that they were probably approaching a dynamic equilibrium (Ruiz et al. 2017) after a hasty glacier retreat between the 1980s and 2000s in the Northern Patagonian Andes (Paul and Mölg 2014). Drastic glacier area changes before the 2000s have been linked (Ruiz et al. 2017) to the notable increase in warm season temperatures in the Northern Patagonian Andes in the late 1970s (Giese et al. 2002). The increase in glacier mass loss between 2010 and 2018 coincides with the megadrought of the Central Andes (Garreaud et al. 2020) and conspicuous volcanic eruptions and large wildfires that decrease the albedo, increasing the melt of snow and ice (Gelman Constantin et al. 2020).

Due to the larger glaciated area, glacier mass loss in the Patagonian Andes is concentrated in Southern Patagonian Andes, with the highest mass loss rates in the NPI and the SPI. Indeed, approximately 80% of recent mass loss from all Andean glaciers originates in these icefields alone (Braun et al. 2019; Dussaillant et al. 2019). Although ice mass loss and glacier front retreat in the Southern Patagonian Andes have been related to tropospheric warming (Rasmussen et al. 2007), climate and ice-melt models applied to the area have shown positive mass balances with an increasing trend between 1975 and 2011 (Schaefer et al. 2013; Lenaerts et al. 2014; Schaefer et al. 2015; Mernild et al. 2017). The causes behind this discrepancy seems to be the overestimation of precipitation rate by climate models used to force the surface mass balance models (Lenaerts et al. 2014; Bravo et al. 2021).

Most of the NPI and SPI glaciers' large glaciers are calving into fjords or lakes (Warren and Sugden 1993; Aniya et al. 1996) and have shown a more complex reaction to climate change than land terminating glaciers (Meier and Post 1987). In these calving glaciers, the initial retreat is often associated with warming air or water temperatures. However, subsequent retreat and acceleration are attributed to the dynamic response of the glacier to changes in the force balance near the front, which initiates positive feedback between thinning and acceleration, increasing the iceberg calving rate (Benn et al. 2007; Sakakibara et al. 2013), making the processes behind the rapid change of frontal ablation and glacier changes quite complex (Benn et al. 2007; Truffer and Motyka 2016). Until very recently, only a few studies were available for individual glaciers in Patagonia to quantify the frontal ablation (Stuefer et al. 2007; Koppes et al. 2011; Collao-Barrios et al. 2018; Bown et al. 2019). Recently, Minowa et al. (2021) quantified the role of frontal ablation in ice mass change for the 38 major calving glaciers in the NPI and the SPI. They found that frontal ablation from 2000 to 2019 represents 34 ± 6% of the total ablation of the icefields, with frontal ablation representing one-fifth of the total ablation in the NPI and close to half of the total ablation in the SPI. They conclude that a substantial increase in frontal ablation at several glaciers drives the observed ice mass loss in the SPI, whereas increasingly negative surface mass balance drives the recent ice mass loss in the NPI. The same pattern is observed in other areas of the Patagonian Andes, such as Monte Tronador or Monte San Lorenzo, where glacier mass loss increased due to calving into recently formed proglacial lakes (Ruiz et al. 2017; Falaschi et al. 2019).

## 6 Hydrological Significance of Glaciers in the Patagonian Andes

The Patagonian Andes represent the source of freshwater for more than 2.4 million people. Thus, the climatic changes in the headquarters of the most important river basins in Patagonia may potentially have a significant societal impact (Johansen et al. 2018). Nonetheless, despite their relevance, many key hydrological processes in the upper river basins are only marginally known due to an overall lack of information and the logistic constraints associated with studying these remote and generally inaccessible areas (See Chap. 9). Due to high precipitation levels in the Patagonian Andes, the hydrological significance of glaciers in this region is generally low, with the glacier ice melt serving as a flow buffer during dry intervals rather than a major source of river flow. Further north, in the Dry Andes region, the contribution of glaciers and other ice masses becomes more hydrologically relevant (Ayala et al. 2020).

Masiokas et al. (2019) found that many river basins along the Patagonian Andes show higher streamflow values in winter, associated mainly with elevated rainfall levels during the colder months. Some rivers also show a second streamflow peak in spring due to the melting of the winter snowpack accumulated in their upper basin areas. The Santa Cruz river, in contrast, shows a distinctive hydrological regime that is strongly influenced by the extensive glaciers located in its upper basin. This river drains ca. 3000 km$^2$ of glacier ice along the southeastern portion of the SPI and shows a single peak in the late summer—early fall largely dominated by glacier melt, with minimum discharges at the end of the winter before the onset of the melting season. This particular seasonal pattern can also be found in other river basins that contain a considerable proportion of their area covered by glaciers. However, as most glaciated basins in Patagonia also receive high amounts of solid and liquid precipitation, their hydrology is more likely dependent on winter precipitation and spring/early summer snowmelt, with a variable influence of glacier ice melt bolstering inputs later in the summer. The hydrological significance of other cryospheric reservoirs such as rock glaciers has not been estimated but may represent a substantial contribution for some small upper basins along the driest, eastern sectors of the Patagonian Andes (For hydrological details, see Chap. 9).

Dussaillant et al. (2019) found that there has been a slight increase in the decadal glacier contribution to streamflow between the first and the second decades of the twenty-first century in the region. For the Baker River Basin draining the southern portion of the NPI, the glacier contribution at a decadal scale increased from 3 to 5%, whereas for the Santa Cruz river draining the eastern sector of the SPI, the decadal contribution increased from 14 to 16%.

Huss and Hock (2018) investigated the future changes in glacial runoff due to future glacier change in several larger river basins of the world, including the Baker and Santa Cruz river basins. Their study focused on the occurrence of the "peak water" during the twenty-first century and the change in the glacier contribution to river runoff for three different Representative Concentration Pathways (RCP's 2.6, 4.5, and 8.5) emission scenarios. For the Baker river, they found that independent of the emission scenario, the glacier peak water has already happened or will occur in the coming years (2015 ± 18 yrs for RCP 2.6, and 2020 ± 16 yrs for RCP 8.5). The late summer glacier contribution at the end of the century in late summer will decrease between 28 ± 10% for a low emission scenario (RCP 2.6) and 40 ± 10% for a high emission scenario (RCP 8.5) due to shrinkage of glaciers in this river basin. For the Santa Cruz river basin, it was projected that the peak water will occur between 2050 ± 18 yrs (RCP 2.6) and 2096 ± 9 yrs (RCP 8.5). Due to the large glacier cover and the projected mass loss, the glacier contribution during the summer months in this basin will continue to increase by 3 ± 5% (RCP 2.6) or 31 ± 11% (RCP 8.5) at the end of the century.

## 7 Glacier as Risk Sources in the Patagonian Andes

Glacial lakes are lakes that are fed by glacier meltwater, including lakes on top of glaciers (supra-glacial), lakes in front of glaciers (proglacial), and glacier-fed lakes that are disconnected from glaciers but close to glacier termini (See Chap. 3). Supra-glacial lakes are often ephemeral and drain when they connect to the englacial drainage system. However, when the lakes reach the glacier bed, supra-glacial lakes can grow and coalesce. When these water bodies collect behind the terminal moraine, they form proglacial lakes and often grow concomitant with glacier recession. Also, ice-dammed lakes can form when a surging glacier blocks a river valley. Proglacial lakes pose a severe risk when the moraine (or glacier) dam fails, and the water drains catastrophically as a glacial lake outburst flood (GLOF), with significant and long-lasting human and physical impacts. Two of the largest river basins in Southern Patagonia (the Santa Cruz and Baker river basins, see above) are affected by the sporadic, rapid drainage of ice-dammed lakes that produce drastic increases in water levels downstream (Dussaillant et al. 2009; Iribarren Anacona et al. 2015). With degrading high-mountain permafrost, there is also an increased probability of rock ice/avalanches from steep slopes reaching glacial lakes and triggering GLOF events (Deline et al. 2015).

Recent glacial lake inventories in the Patagonian Andes (Wilson et al. 2018) show that glacial lakes have increased in number (54% and 44% in the Northern and Southern Patagonian Andes, respectively) and areal extent (50% and 21%, respectively) between 1986 and 2016. Although most of the new lakes have emerged between 1986 and 2000, there was still a considerable increase in the area of glacial lakes between 2000 and 2016 (Fig. 4.9). The distribution of glacial lakes is heavily skewed toward the more glacierized western margin of the Southern Patagonian Andes. The sizes of the individual glacial lakes vary from relatively small in the Northern Patagonian Andes to generally large in the Southern Patagonian Andes. In terms of glacial lake type, most (51% in both regions) lakes present in 2016 were impounded by a moraine dam, followed by rock-bar (48% and 41% in Northern and Southern Patagonian Andes, respectively) and ice dam (0.4% and 7% in Northern and Southern Patagonian Andes, respectively). Those lakes still in contact with glaciers have the most significant potential for future change, with 15% of glacial lakes in the Northern and 26% in the Southern Andes.

The increase in the number and size of glacial lakes in the Patagonian Andes is associated with the widespread thinning and retreat of glacier tongues that initiated after the latest Holocene maximum (the end of the LIA). Although Northern Patagonia has experienced the most significant changes in glacial lakes, the rate of arial growth of these lakes has reduced considerably between 2000 and 2016 (Fig. 4.9). The number of newly emerging glacial lakes in Northern Patagonia during this latter period has declined, too. These changes are likely related to differences in topography, rate of glacier changes, climate change, and the availability of low gradient ice areas. These factors will influence the rate of lake growth and the likelihood of outburst events in each sub-region in the future (Wilson et al. 2018).

Iribarren Anacona et al. (2015) document dozens of GLOFs since the eighteenth century, which have increased in frequency in the last three decades. The frequency increase has been attributed to glacier shrinkage and lake growth and was often triggered by intense or prolonged rainfall and high temperature that increased meltwater production. In addition to observing several non-catastrophic glacial lake reductions and disappearances between 1986 and 2016, Wilson et al. (2018) identified 21 previously unreported GLOFs, including at least eight events since 2000. These authors highlight that the likelihood of future outburst events is highest in Southern Patagonia, which contains the most significant number of moraine- and ice-dammed lakes. Also, GLOFs are expected to increase in the future as the glacier retreats continue along the Andes (Harrison et al. 2018). Although satellite-based inventory and monitoring of glacial lakes allow a better understanding of lake development, future GLOF risk assessments should incorporate projected changes in lake volume, glacier extent, extreme climatic events, and stability-related hazards. Assessing the overall GLOF risk in the Patagonian Andes is crucial for better planning and implementing disaster risk reduction measures and climate change adaptation in the region.

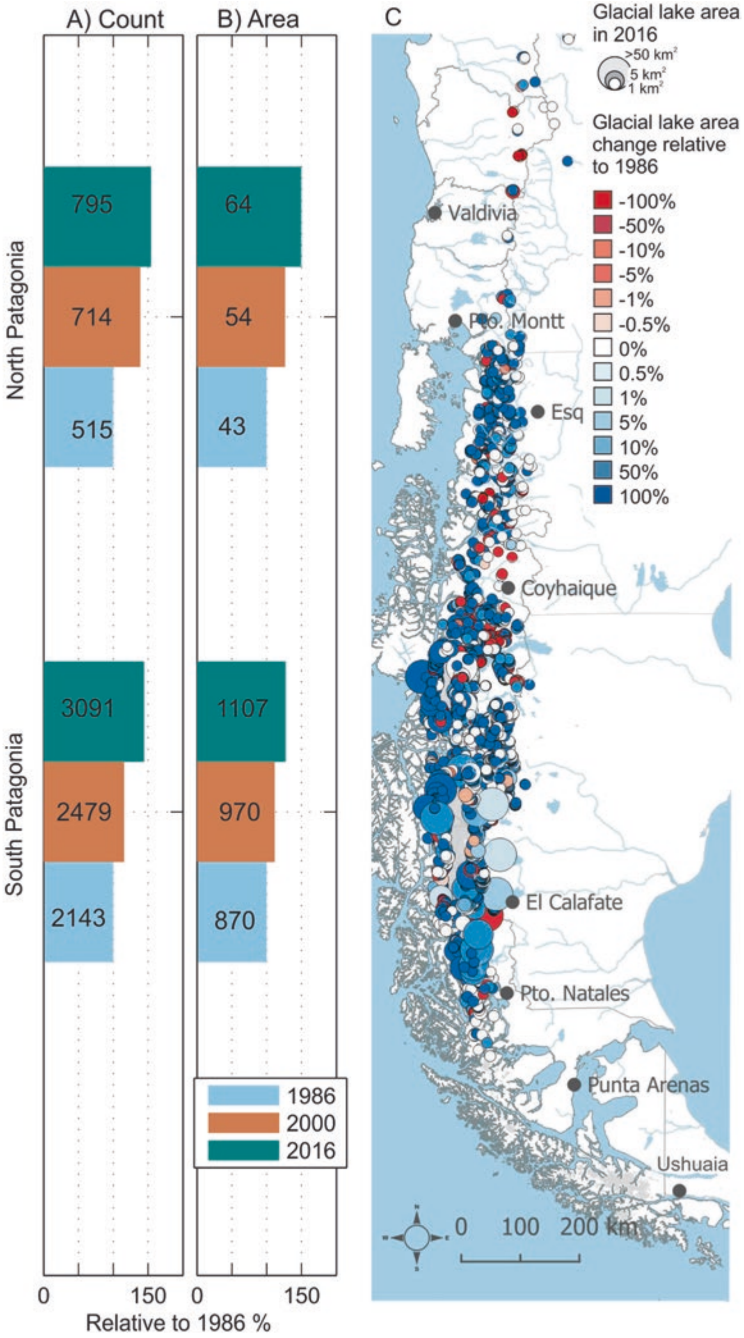

**Fig. 4.9** Glacial lakes change between 1986 and 2015 in the Patagonian Andes. (**a**) Number of glacial lakes between 1986, 2000 and 2016. (**b**) Extent of glacial lakes between 1986, 2000 and 2016. Bars show the relative changes in terms of number (**a**) and area (**b**) since 1986. For each bar the amount and total area in km² are also shown. (**c**) Map of Patagonian glacial lakes changes. (Data from Wilson et al. (2018)). Each dot represents the location of one glacial lake. Note that no glacial lakes were mapped by Wilson et al. (2018) in the Fuegian Andes

# 8 Regulation Measures and Protection of Patagonian Glaciers

Glaciers and other ice masses are critical components of the hydrological cycle of the Andes. Recognizing glaciers' environmental, socioeconomic, and cultural significance and the threat of potentially damaging activities has motivated governments from Chile and Argentina to discuss or enact laws to protect them (Iribarren Anacona et al. 2018). In 2010, social concerns about the impact of mining activities along the Andes and the perception of weak or absent control by governmental agencies promoted the discussion and approval of unique legislation to protect glaciers and the periglacial environment in Argentina (Rojas and Wegener 2020).

At least three revised versions of laws to protect the country's ice masses have been proposed in Chile since 2006. Environmental non-governmental organizations claim that the law must be strict enough to protect glaciers and the ecosystem contributions they provide, whereas mining companies declare that proposed protective measures could undermine the main economic activity of Chile (Iribarren Anacona et al. 2018).

Along the Patagonian Andes, there are no mining activities that could impact glaciers directly. Indeed, most of the glacier-covered area in this region is already located in different natural protected areas (e.g., national, province, or private lands). For example, of the more than 3715 $km^2$ covered by ice in the Patagonian Andes of Argentina, 94% is in a protected natural area. The same in Chile, where more than 90% of the glacier covered area (23,427 $km^2$), is in a protected natural area.

The approval of the "glacier law" in Argentina, but also the discussion of the different projects in Chile, has created new paradigms and opportunities for interactions between science and policy. The definition of common goals and objectives between science and policy has also resulted in different products and activities, such as the glacier inventories of Argentina and Chile, and the implementation of glacier monitoring programs at national scales or for specific mining projects. With no doubt, these recent developments have substantially increased the knowledge and information about Patagonian glaciers in the last decade (IANIGLA and MAyDS 2018).

Nevertheless, a necessary joint problem framing and a more effective interaction between social and physical climate and impact sciences are still required for grounding policy in science. This can certainly help to effectively allocate resources to address the environmental challenges caused by climate change in the Andes and the associated threat to lives and livelihood (Johansen et al. 2018).

## 9 Concluding Remarks

The Patagonian Andes contain the most significant area covered by glaciers in the Andes and the entire Southern Hemisphere outside Antarctica. This alone puts this region in a central position for the development of glaciological studies in South America. However, Patagonian glaciers are also highly relevant from other scientific, environmental, socioeconomic activities, and cultural perspectives, including, for example, studies of climate change impacts on natural resources, sea-level rise assessments, hydrological analyses, the positive impact on tourism, and a great diversity of cultural considerations.

This chapter provides a brief overview of several critical aspects regarding the current state and the recent changes of Patagonia's glaciers (key numbers in Table 4.1). The issues discussed above include descriptions of recently published national glacier inventories representing historical milestones in glaciological studies for Chile and Argentina. As in many other mountainous areas of the world, the increasing accessibility and spatial resolution of remote-sensing (satellite) information have represented a crucial point for developing glacier inventories and many other related analyses with an unprecedented amount of detail.

The ongoing retraction of the ice masses in Patagonia has been documented by numerous studies and is a source of concern not only for glaciologists but for the broader public, who see in this generalized ice mass loss a clear sign of the ongoing climate change that affects Patagonia and the planet.

**Table 4.1** Summary of the current state and recent changes of glaciers in the Patagonian Andes

| Region | Fuegian Andes | Southern Patagonian Andes | Northern Patagonian Andes | Patagonian Andes |
|---|---|---|---|---|
| Number of glacier[a] | 2920 | 14,546 | 6608 | 24,074 |
| Glacier area km$^2$[a] | 3476 | 20,863 | 1761 | 26,100 |
| Glacier volume (km$^3$) | 262[b] / 364[c] | 5735[b] / 4764[c] | 32[b] / 94[c] | 6029[b] / 5222[c] |
| Area change (1870–2011)[d] km$^2$ yr$^{-1}$ | −3.0 | −24.2 | −2.1 | −29.3 |
| Mass change (2000–2018)[e] m w.e. yr$^{-1}$ | −0.48 ± 0.27 | −0.86 ± 0.27 | −0.57 ± 0.22 | −0.78 ± 0.22 |
| Changes in glacial lakes (1986–2016)[f] km$^2$ yr$^{-1}$ | 0.7 | 10.1 | 7.9 | 18.8 |

[a] Number and area of glaciers from Barcaza et al. (2017) and Zalazar et al. (2020)
[b] Glacier volume from Carrivick et al. (2016)
[c] Glacier volume from Farinotti et al. (2019)
[e] Glacier mass change from Dussaillant et al. (2019)
[f] Changes in glacial lakes from Wilson et al. (2018)

The Patagonian Andes have lost nearly 1 meter of ice thickness per year over the entire glacier area (26,100 km$^2$) between 2000 and 2018. Given the exceptionally large area covered by glacier ice in Patagonia, this high and steady ice mass loss of the past two decades has produced an immense amount of meltwater with an already discernible impact on sea-level rise. In fact, Patagonia alone accounts for 16 ± 4% of the sea-level rise estimated from the melting of glaciers in mountain areas (Hock and Rasul 2019).

The meltwater produced by glaciers in most river basins along the Patagonian Andes is, however, usually only a secondary hydrological input for surface flow draining the main river basins. The humid conditions that characterize this region make rainfall and snowmelt the most important contributors to rivers, and thus the seasonal changes in streamflow follow the seasonal patterns of those two major water inputs. Only in catchments where the glacier area reaches a considerable proportion of the total area (usually in the upper basins nearby the glaciers), the glacier melt inputs can reach a discernible hydrological signal and affect the seasonality of the surface flow. In these cases, the peak flow usually occurs toward the end of the summer, when the seasonal snow has already disappeared, but the temperatures are high enough to continue promoting the further melting of glacier ice. The Santa Cruz River in Southern Patagonia is an exciting case study that drains a vast basin with a large part located outside the Andean range, but also ca. 3000 km$^2$ of glacier ice along the eastern portion of the SPI. This extensive glacier cover produces a detectable hydrological signal with a single streamflow peak at the end of the summer and minimum flows in spring at the end of the cold season.

The ongoing, and in some cases dramatic, thinning and recession of the lower reach of glaciers has brought about other issues associated with the risk of destabilization of slopes that were formerly supported by the glacier tongues. This process can trigger a variety of situations that may directly or indirectly impact human lives and infrastructure on-site or further downstream. The retraction of the glaciers has also resulted in the formation of numerous proglacial lakes, with their number and surface area continuing to increase. One of the consequences of this increase in the number and volume of proglacial lakes is the increased risk of sudden floods caused by failure of the moraines or the ice walls that dam these newly formed lakes. Several such events have been reported across the Patagonian Andes, and some of them had severe impacts on the riverbed and nearby infrastructure.

Some pressing issues regarding future studies of glaciers in Patagonia include a better understanding of the ice dynamics of the different glaciers of the Patagonian icefields. As mentioned above, these icefields constitute the largest ice masses of the Southern Hemisphere but are formed by numerous outlet tongues with quite contrasting behaviors. Within the regional and well-documented thinning trend, some outlet glacier fronts have retreated dramatically (in the order of several km in a few decades). Others have remained relatively stable, and some isolated cases have even shown advances overriding deposits from the LIA chronozone. These contrasts in glacier dynamics are certainly challenging from a glaciological perspective but highlight the need to assess the potential impacts of future climate change in the region with caution, considering the particularities of each ice mass within the general picture.

The possible scenarios for glacier changes in the coming decades will depend on the trajectories taken by the main climate variables across Patagonia. These scenarios will also necessarily involve several inherent and complex ice dynamics processes that depend on the size, location, presence of a proglacial lake or debris cover, among others, at each site. A better understanding of the glacier–climate relationships and the ice dynamics in different sectors of the Patagonian Andes can thus help improve our current understanding of these issues but will also prove essential for the development of reliable projections of future glacier changes in this extensive mountain range.

Many aspects of future climate change and glacier response in Patagonia still remain highly uncertain. Social concern about the impacts of human activities in mountain areas of Chile and Argentina has located glaciers and the periglacial environment at the spotlight scene of policy, social, and science debate in Argentina and Chile. Although recent, science ground-based policy measures and initiatives have advanced the knowledge of glaciers and their hydrological role along the region, they must be continued in the future to become truly effective. The regular repetition of glacier inventories and monitoring programs are crucial both to assess the ongoing ice mass changes at local and regional scales. These results and initiatives are also essential to outline solid environmental policies for glacier protection and to develop efficient mitigation and adaptation strategies in response to the ongoing climate changes affecting the Patagonian Andes.

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# Chapter 5
# Biogeographical Patterns of Patagonian Freshwater Microbiota

**M. Romina Schiaffino and Irina Izaguirre**

## 1 Introduction

In ecology, a pattern is a statement about relationships among several observations of nature, which implies some sort of repetition, thus suggesting that some prediction is possible (McArthur 1972). The first approach to explain the structure and functioning of a community is looking for patterns, as for example those of repeated similarities or trends along different environmental gradients over space and/or time. The existence of a particular pattern and their underlying or associated processes depends on the study scale (Levin 1992; Chave 2013). Ecological processes act at a variety of spatial and temporal scales, and they generate patterns at scales that may differ from those at which processes act (Levin 1992).

Biogeography is the study of the spatial distribution of biological diversity (Lindström and Langenheder 2012, and cites therein). The study of biogeographical patterns allows to comprehend the distribution of organisms and their biodiversity over space and time, through the observation, recording and explanation of the geographical ranges of organisms (Pielou 1979; Dolan 2005). These patterns provide insight into the mechanisms that generate and maintain biodiversity (Martiny et al. 2006) and into the ecological and evolutionary processes that shape it (Fenchel

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2003). The arising of biogeographic patterns is frequently scale dependent, and historic legacies (e.g. colonisation), repeated evolutionary patterns (e.g. speciation and extinction processes) and contemporary environmental factors (e.g. pH, salinity) may influence current distribution trends (Dolan 2005; Martiny et al. 2006; Chave 2013).

Particularly, microbial biogeographical patterns provide a valuable framework for understanding the overall ecology of microorganisms and the factors that control their functions and the ecological services they provide in the natural environment (Cotterill et al. 2007; Hanson 2017). Some hypotheses providing a useful context to further explore microbial biogeography have been postulated (Martiny et al. 2006; Lindström and Langenheder 2012). Many studies disputed the idea that 'everything is everywhere' (Baas-Becking 1934) based on the 'ubiquity model' and the cosmopolitan distribution of microorganisms (implying that microorganisms present high dispersal capabilities that erase the effect of past evolutionary and historical events). However, as proposed by the Baas-Becking hypothesis, 'the environment selects', implying that contemporary local environmental changes are responsible for spatial variation in microbial diversity (e.g. Fenchel et al. 1997; Finlay 2002; Fenchel and Finlay 2004). Contrarily, it has been postulated that some microorganisms are endemic to some geographical areas and present restricted distribution patterns (due to the effect of historical events, such as dispersal barriers or past environmental conditions), based on the 'moderate endemicity model' (Foissner 2006, 2008). In this sense, a final hypothesis postulated that the distribution of microbial taxa, like those of macroorganisms, reflect the influence of both past events and contemporary environmental conditions (Martiny et al. 2006). Currently, it is recognised that these mechanisms are not mutually exclusive but interact in the assembly of microbial communities to determine how species diversity and composition vary along environmental and/or spatio-temporal gradients (Lindström and Langenheder 2012). It has also been stated that microbial biogeography differs from traditional biogeography, as historical factors are much less important than biological and physical factors (Dolan 2005), due to short generation times, high population density and growth rates, small body sizes and generally easy dispersion (Dolan 2005, 2006). Although microorganisms exhibit patterns in relative abundance, distribution, diversity and traits across space and time, it remains unclear to which extent they follow macroecological rules initially developed for macroorganisms (Dickey et al. 2021), even if at least some groups are known to follow these patterns (e.g., Martiny et al. 2006 and cites therein, Tell et al. 2011; Schiaffino et al. 2011; Izaguirre et al. 2016; Fernández et al. 2017).

A general biogeographical pattern is the latitudinal (or altitudinal) biodiversity gradient, according to which diversity decreases towards higher latitudes (or altitudes) (Hillebrand 2004). Thanks to large-scale DNA-based screens of microbial diversity (Pace 1997), which disclosed a largely unknown taxonomic diversity of prokaryotic and eukaryotic microorganisms (e.g. de Vargas et al. 2015; Singer et al. 2021), latitudinal diversity gradients have now been also studied for microorganisms (Swanson et al. 1999; Fuhrman et al. 2008; Stephenson and Feest 2013; Lara et al. 2016). Latitudinal gradients of diversity are ultimately dependent on the historical, geographical, biotic, abiotic and stochastic forces (Schemske 2002). Certainly, latitude is a surrogate for a number of primary environmental gradients (e.g. temperature, seasonality) that interact and correlate to each other (Willig et al. 2003). Another evidence for microbial biogeography is the taxa–area relationship. This implies an increase of taxa richness and diversity with increasing sample area of both contiguous and islands habitats or a decrease of taxa richness and diversity with increasing remoteness of a habitat or island (MacArthur and Wilson 1967; Horner-Devine et al. 2004; Bell et al. 2005; Reche et al. 2005). The richness of any region is a consequence of two factors: the richness of each of the smaller areas that compose it and the turnover in species composition (beta diversity) among them (Willig et al. 2003 and cites therein). When considering the beta diversity or similarity of species composition, it typically decreases with increasing distance (such as space, time or environment) (Soininen et al. 2007; Soininen 2010). It refers to the 'distance approach' (Tuomisto and Ruokolainen 2006) and measures the variation or rate in taxa turnover. The distance decay appears to be a product of latitude, study extent and environment, as well as it was a correlate of organism characteristics (Soininen et al. 2007).

Inland waters are excellent scenarios and model systems in which to investigate microbial biogeographical patterns because of their island-like nature, discrete boundaries, hydrological disconnections and their relative heterogeneity (Papke and Ward 2004; Reche et al. 2005).

The Argentinean Patagonian Region (37°–56°S, 64°–72°W) encompasses a vast extension of land with scarce human presence, relatively undisturbed areas and wide heterogeneity of freshwater ecosystems (e.g. Modenutti et al. 1998; Epele et al. 2018; Castro Berman et al. 2020) (See Chaps. 3, 9 and 11). Eastwards from the Andes, the Patagonian climate varies from Andean humid cold to arid, with a strong west-east decreasing gradient of precipitation (from >1000 to <200 mm annual rainfall), mainly concentrated in winters (Barros and Mattio 1977). Besides, strong constant west winds are dominant across the region and mean annual temperature ranges from 12 °C in the northeast to 3 °C towards the south (Paruelo et al. 1998). The climate, geological history, geomorphological features, human activities and the distinctive food web structures (presence or absence of top predators, such as fish; see Chap. 13) reveal strong differences in the Patagonian freshwater systems, affecting species distribution (Modenutti et al. 1998). The Patagonian inland waters are classified according to two main geographical and geomorphological regions: The Andean Patagonia and the Patagonian Plateau (Quirós and Drago 1999; Morello et al. 2018). The Andean Patagonia Region is characterised by large, deep and ultraoligotrophic-oligotrophic lakes, with glacial and fluvial erosion origin (Díaz et al. 2000; see Chap. 3), whereas the Patagonian Plateau is a complex landscape with different types of water bodies that mainly range from mesotrophic to eutrophic: large artificial lakes, permanent natural lakes and temporary waters (Quirós and Drago 1999; Chap. 11). In Tierra del Fuego (Insular Patagonia), some lakes are placed near the mountains and belong to the Andean Patagonia Region, while others are placed in the Patagonian Plateau of the island, among which most of those of the southeastern are humic lakes (Tell et al. 2011) (See Chaps. 3, 9 and 11).

Studies on microscopic algae or phytoplankton from Argentinean Patagonian lakes started about 72 years ago (Guarrera and Kühnemann 1949) and included mostly floristic papers (Tell et al. 2011 and cites therein). The first studies on autotrophic picoplankton, such as picocyanobacteria (Pcy) and picoeukaryotes (Peuk), started around 21 years ago (Zunino and Diaz 2000), while first studies on heterotrophic bacteria (HB) were conducted around 26 years ago (Di Siervi et al. 1995). Even though there is abundant literature related to microorganisms from Patagonian lakes, a lesser number of studies have focused on biogeographical and environmental gradients of prokaryotic (Schiaffino et al. 2011, 2013, 2016a; Saad et al. 2013, 2016; Porcel et al. 2019; Castro-Berman et al. 2020) and eukaryotic (e.g. Tell 1995; Díaz et al. 2000; Maidana et al. 2005; Tell et al. 2011; Saad et al. 2013; Queimaliños and Díaz 2014 and cites therein, Izaguirre et al. 2016; Schiaffino et al. 2016b; Fernández et al. 2017; Saad et al. 2019; Porcel et al. 2020) microorganisms. Some of them have reported decreasing microbial diversity and similarity towards higher latitudes, as well as influences of contemporary environmental conditions on freshwater microbial assemblages (e.g. Schiaffino et al. 2011; Izaguirre et al. 2016), aspects that are addressed in this chapter.

Data from Maritime Antarctic lakes were additionally incorporated in this chapter in order to increase the spatial scale and unravel larger scale biogeographical patterns. The Maritime Antarctic Region is characterised by higher mean temperatures and precipitation than the Continental Antarctic Region, as well as by a large number of freshwater ecosystems that melt out and become ice-free in summer (Laybourn-Parry and Wadham 2014). These Antarctic water bodies are remote environments naturally ultra-oligotrophic or oligotrophic, but there are some meso-eutrophic shallow lakes and hypertrophic ponds influenced mainly by their proximity to bird colonies (Izaguirre et al. 2021 and references therein). Patagonia and Antarctica are separated by the Drake Passage that probably works like an important geographical barrier, providing an ideal scenario to study biogeographical patterns. Still, it has been stated that South America, and in particular Patagonia, could be a source of propagules that can be transported aerially to the Antarctic Peninsula (e.g. Hughes et al. 2004).

Patterns of microbial biogeography have remained poorly described because tools for measuring microbial diversity have emerged in the last years. In addition, in order to better understand microorganism diversity and their biogeographic distribution patterns, the study of prokaryotic and eukaryotic microorganisms would benefit from polyphasic approaches combining microscopy studies with molecular approaches (e.g. Weisse 2008; Jung et al. 2019; Izaguirre et al. 2021). In this chapter, we focused on biogeographical patterns of planktonic prokaryotes and eukaryotes along a gradient of Patagonian and Antarctic lakes, including an overview of present-day knowledge.

# 2 Prokaryotic Assemblages and Their Patterns

## *2.1 Molecular Diversity Patterns*

Through the application of molecular approaches into ecological studies, a large body of research started to support the idea that free living microorganisms exhibit biogeographic patterns (e.g. Martiny et al. 2006; Lindström and Langenheder 2012; Hanson 2017). Available evidence shows that bacterial community composition (BCC) in freshwaters is controlled by dispersal (biogeography) or contemporary environment (habitat quality), or both, and that these controls work across a spatial range from local to regional scales (Crump et al. 2007; Lindström and Langenheder 2012). Biogeographical, altitudinal and environmental patterns in BCC diversity using molecular approaches along gradients of Patagonian freshwater systems were assessed in a few papers (e.g. Corno et al. 2009; Caravati et al. 2010; Schiaffino et al. 2011, 2016a; Nimptsch et al. 2016; Bastidas Navarro et al. 2014).

According to latitudinal patterns, Schiaffino et al. (2011) studied the BCC and dominant bacterial richness of 45 water bodies covering a gradient of ~2100 km stretching from Argentinean Patagonia (45°S) to Maritime Antarctica (63°S). This study was mainly focused on BCC assessed by denaturing gradient gel electrophoresis (DGGE) and band sequencing in order to evaluate the latitudinal variation of bacterioplankton and determine the factors influencing bacterial structure. The obtained results showed that BCC was controlled by a combination of spatial (latitude and longitude) and environmental factors (phosphate, dissolved organic carbon (DOC) and light diffuse attenuation coefficient ($K_d$)). Using the same dataset, here we show that BCC similarity declines with increasing geographical distance (Fig. 5.1a).

Additionally, significant differences were found in BCC between Antarctic and Patagonian regions (based on DGGE band intensity ANOSIM $r = 0.46$ [Bray Curtis] and based on DGGE band presence/absence $r = 0.69$ (Jaccard), both $p < 0.0001$). The bacteria obtained by DGGE band sequencing in the studied lakes from both continents belonged to Cyanobacteria, Bacteroidetes, Actinobacteria, Beta- and Alphaproteobacteria, Firmicutes and Acidobacteria. A general biogeographical pattern supports the idea that diversity decreases towards higher latitudes (Hillebrand 2004). Accordingly, Schiaffino et al. (2011) also found that dominant bacterial richness decreased with increasing latitude, with a significant negative correlation ($r = -0.44$, $p = 0.004$, $n = 41$). A similar pattern was found when including in these dataset bacterial DGGE data from other Patagonian lakes located in the Strobel Plateau (Fig. 5.1b) (Porcel et al. unpublished data).

It has been calculated that about $10^{18}$ viable bacteria annually are transported through the atmosphere between continents (Griffin et al. 2002). Therefore, at large spatial scales (regional or continental) dispersal related to atmospheric transport and deposition of bacteria would be important. However, differences in the ability to support stress or unfavourable conditions during dispersal may be a trait leading to different assembly mechanisms (Lindström and Langenheder 2012 and cites

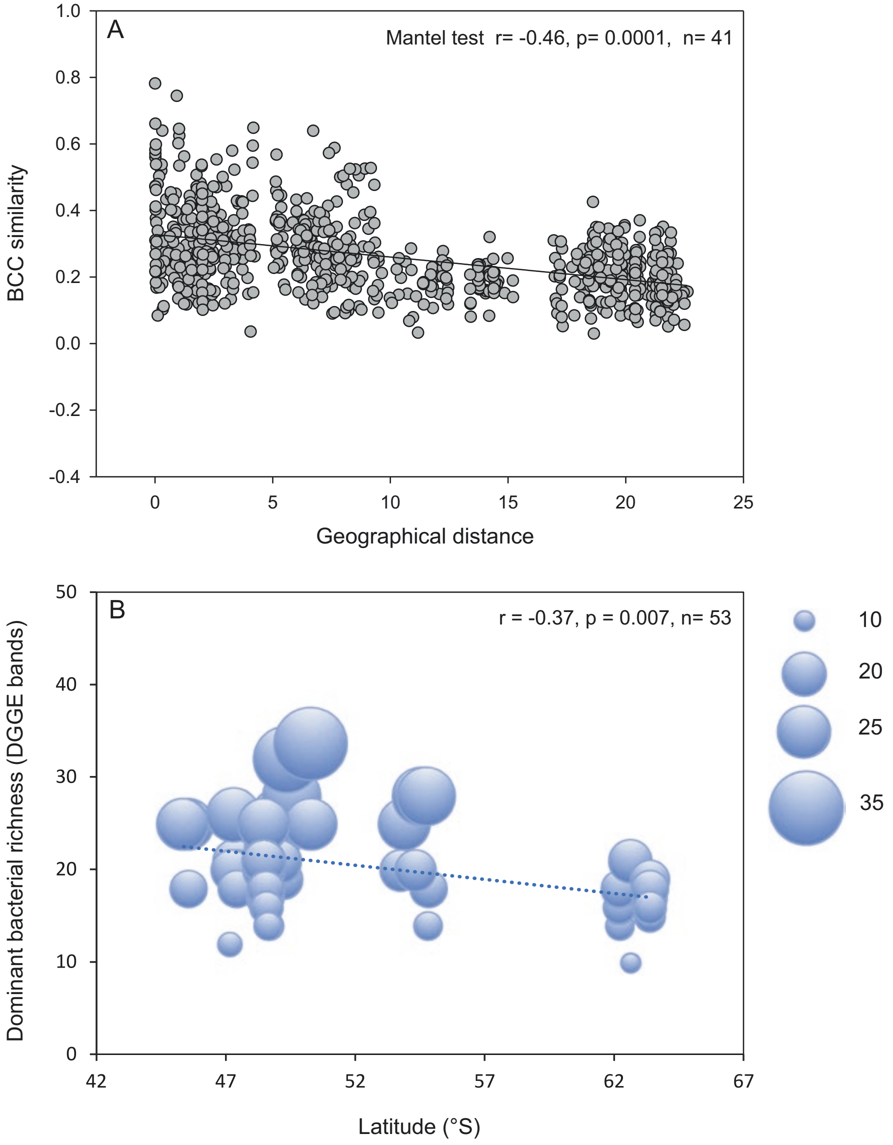

**Fig. 5.1** (**a**) Relationship between bacterial community composition (BCC based on DGGE band intensity) similarity and geographical distance obtained from Schiaffino et al. (2011). Correlation value (Spearman) shows the result from partial Mantel Test, while holding environmental matrix constant. (**b**) Relationship between bacterial operational taxonomic unit (OTU) richness (DGGE bands) and latitude (Spearman rho correlation), including data from literature (Schiaffino et al. 2011; Porcel et al. unpublished data) that used the same set of primers and DGGE run conditions

therein). Accordingly, Schiaffino et al. (2011) found that about 59% of dominant bacterial operational taxonomic units (OTUs) were shared between Patagonian and Antarctic water bodies, while 37% where exclusive of the studied Patagonian lakes and 4% exclusive of the studied Maritime Antarctic lakes.

In a subset of 35 water bodies located in a latitudinal gradient within Patagonia, Schiaffino et al. (2016a) studied the bacterial and archaeal distribution patterns at phylum and class level using catalysed reporter deposition–fluorescence in situ hybridisation (CARD–FISH). The authors found that Alphaproteobacteria, followed by Betaproteobacteria and Actinobacteria groups, globally dominated the bacterioplankton, while Gammaproteobacteria and Archaea groups were less represented (3% and 6% of the community, respectively). Additionally, the lake grouping obtained with CARD–FISH absolute abundance of bacterial groups was consistent with a previous work using bacterial DGGE data band intensity pattern for the same set of Patagonian lakes (Schiaffino et al. 2011): deep oligotrophic lakes clustered together, whereas meso-eutrophic small and shallow water bodies grouped separately (Mantel test, $r = 0.25$, $p < 0.0001$, $n = 29$). Even though both molecular approaches showed that lakes with similar trophic and morphometric characteristics presented similar dominant bacterial composition (DGGE) and abundances at phylum and class levels (CARD–FISH), in this last study with a subset of Patagonian lakes, the latitudinal pattern was not clear, probably because the absence of Antarctic lakes in this dataset and/or the taxonomic resolution provided by CARD–FISH.

Additionally, a study performed by Souffreau et al. (2015) evaluated the BCC in 48 shallow lakes along a ~ 6200 km latitudinal gradient in South America (5–55°S), including some Patagonian ones. The authors assessed the relative importance of environmental and spatial factors in shaping shallow lake BCC from tropical to tundra climate in South America using DGGE. They found significant differences in BCC across regions and that the variation in BCC over the whole latitudinal gradient was due to local environmental variables (3–7%), spatially related environmental factors (6–8%) and to a lesser extent spatial factors (2–3%).

In relation to altitudinal gradients in Patagonian lakes, Bastidas Navarro et al. (2014) studied the BCC from 8 North Patagonian Andes lakes located from 1380 to 1950 m.a.s.l. BCC and diversity were assessed by automated ribosomal intergenic spacer analysis (ARISA) and bacterial community-level physiological profiles by BiologEcoPlate™. These authors found that altitude and resource concentration (DOC and total dissolved phosphorus) were the main variables determining the BCC, which also differed between above and below timberline. Besides, community-level physiological profiles showed a discrepancy with BCC, indicating no relationship between genetic and functional diversity over the altitudinal gradient. However, the different bacterial assemblages found in the studied lakes used the same group of substrates, suggesting that the distinct assemblages keep the same functions.

On the other hand, environmental gradients in Patagonian lakes were assessed by Corno et al. (2009), who investigated the effect of UVR on bacterial diversity and morphology in 9 deep ultraoligotrophic North Andean Patagonian lakes located in different drainage basins using DGGE and band sequencing. The authors found that the overall BCC was similar in all the studied lakes and over depth in each lake and neither the watershed (Pacific or Atlantic) nor the basin (NahuelHuapi, Futaleufú and Manso) was central in determining the dominant BCC. However, their morphological diversity changed, with a higher proportion of bacterial filaments in the upper layers with stronger UVR intensities (305–340 nm). Caravati et al. (2010) carried out one of the first studies of the Pcy biodiversity in Patagonian lakes using a molecular technique (ARISA) in 6 deep ultraoligotrophic structure consisting of 18 OTUs, of which only one was present at all depths of the 6 studied lakes. They observed a habitat specificity of some Pcy and a distinctive vertical distribution of the OTUs. These authors suggested that the high beta diversity of the studied lakes support the hypothesis that microdiversity is higher in glacier-derived lake systems wherein habitat fragmentation due to geographic barriers results in rapid speciation. North Patagonian lakes, although characterised by low nutrient concentration and low productivity, revealed the presence of blooms of cyanobacteria, mainly constituted by *Microcystis* and *Dolichospermum* species (Nimptsch et al. 2016). These authors studied the presence of cyanobacteria and different toxins through molecular approaches in six North Patagonian lakes of Chile (Lakes Caburgua, Villarrica, Calafquen, Panguipulli, Ranco and Puyehue). Using a genomic approach, they reported for the first time *Cylindrospermopsisraciborskii* in one of these lakes (Ranco) and the presence of microcystins in all studied lakes. As a final point, the authors highlighted the effectiveness of molecular methods as a first approach to describe the composition of cyanobacterial blooms and their potential cyanotoxins in North Patagonian lakes.

Some of the aforementioned studies used fingerprinting techniques (DGGE and ARISA) to explore the biogeographical and environmental patterns, and although there are limitations because only the dominant diversity is highlighted, these approaches still remain an excellent, highly reproducible and comparatively low-cost community analysis tool when used appropriately (Neilson et al. 2013).

## 2.2 *Abundance Patterns*

The geographical and environmental patterns of prokaryotic abundances were also analysed in not many papers, covering a latitudinal gradient of lakes across Argentinean Patagonia and including some Antarctic lakes (Schiaffino et al. 2011, 2013, 2016a), an altitudinal gradient of Patagonian lakes (Bastidas Navarro et al. 2014), a trophic gradient of Patagonian lakes (Zunino and Diaz 2000; Saad et al. 2013) and a regime gradient (organic and inorganic turbid, clear vegetated and clear unvegetated) of shallow lakes from Patagonia (Saad et al. 2019; Porcel et al. 2019).

In the aforementioned study performed by Schiaffino et al. (2011), the HB and Pcy abundances from 45 water bodies covering a latitudinal gradient of Patagonian and Antarctic aquatic systems were obtained by epifluorescence microscopy. The reported results showed that total HB and Pcy abundances declined with increasing latitude (HB $r = -0.65$ and Pcy $r = -0.77$, both $p < 0.001$ and $n = 45$). Multiple regression analyses showed that phosphate, $K_d$ and latitude had significant effects on total HB abundances. Besides, abundances of HB and Pcy increased with increasing trophic status of the lake (HB $r = 0.63$ and Pcy $r = 0.57$, both $p < 0.001$ and $n = 45$). In a subset of Patagonian lakes located in a small latitudinal gradient (45°–54°S), Schiaffino et al. (2016a) studied the abundance of HB and Archaea using CARD–FISH. As reported in previous results, the absolute abundance of most bacterial groups was significantly higher in mesotrophic and eutrophic systems than in oligotrophic ones, whereas those of Actinobacteria and the archaeal groups did not change among lake trophic status. The author found that Bacteroidetes made a significantly higher relative contribution to community structure in mesotrophic lakes than in oligotrophic ones. In general, the archaeal group Crenarchaeota (0.5%, $1.4 \times 10^4$ cells $mL^{-1}$) was better represented than Euryarchaeota (0.1%, $4.0 \times 10^3$ cells $mL^{-1}$) along the gradient of lakes. Besides, Alphaproteobacteria was the best represented group with absolute abundances that ranged from $5.0 \times 10^4$ to $3.1 \times 10^7$ cells $mL^{-1}$ and an average relative contribution to community composition of 30.0%. Even though differences were found in the bacterial and an archaeal composition at the phylum level between water bodies located towards the Andes and those towards the plateau (trophic gradient), no differences were observed between continental (Chubut and Santa Cruz) and southernmost, insular (Tierra del Fuego) Patagonian lakes. In another study, the abundance of HB, Pcy and their cytometric group diversity was investigated along a trophic and latitudinal gradient of 32 Patagonian shallow lakes from Chubut to Tierra del Fuego provinces by Schiaffino et al. (2013). Authors found a total of 28 cytometric populations within HB, 14 of phycoerythrin-rich (PE-rich) and 8 of phycocyanin-rich (PC-rich) Pcy. They reported a strong influence of environmental factors and a less marked effect of latitude on picoplankton structure. In particular, the abundance of HB and Pcy decreased towards higher latitudes, while the cytometric diversity did not change. Additionally, the relative contribution of PE-rich Pcy to total Pcy decreased with Chl-a, whereas the relative contribution of PC-rich Pcy and the number of PC-rich cytometric populations increased with Chl-a values.

An overall decreasing prokaryotic abundance pattern with increasing latitude was also observed when gathering the results from the abovementioned dataset from Patagonian and Antarctic lakes (Fig. 5.2).

The HB (Fig. 5.2a) and Archaea (Fig. 5.2b) abundances showed a decreasing trend with latitude (HB $r = -0.59$, $p < 0.0001$, $n = 92$ and Archaea $r = -0.35$, $p = 0.026$, $n = 40$). The same declining pattern was observed for total Pcy abundance (Fig. 5.2c, $r = -0.36$, $p < 0.0001$, $n = 93$) and PE-rich Pcy abundance (Fig. 5.2d, $r = -0.26$, $p = 0.048$, $n = 60$) with latitude, whereas PC-rich Pcy (Fig. 5.2e, $r = -0.15$, $p = 0.251$, $n = 60$) did not show a significant decreasing trend. This lack of correlation could be explained because as previously reported, PC-rich Pcy are generally predominant in more eutrophic and turbid water bodies, aquatic systems typically less represented in higher latitudes.

In relation to the elevation gradient, Bastidas Navarro et al. (2014) observed that HB abundances declined along an altitudinal range (1380–1950 m.a.s.l) of 8 North Andean Patagonian lakes, with values ranging from $6.7 \times 10^5$ to $2.5 \times 10^6$ cells $mL^{-1}$. The authors suggested that this bacterial abundance decrease with increasing

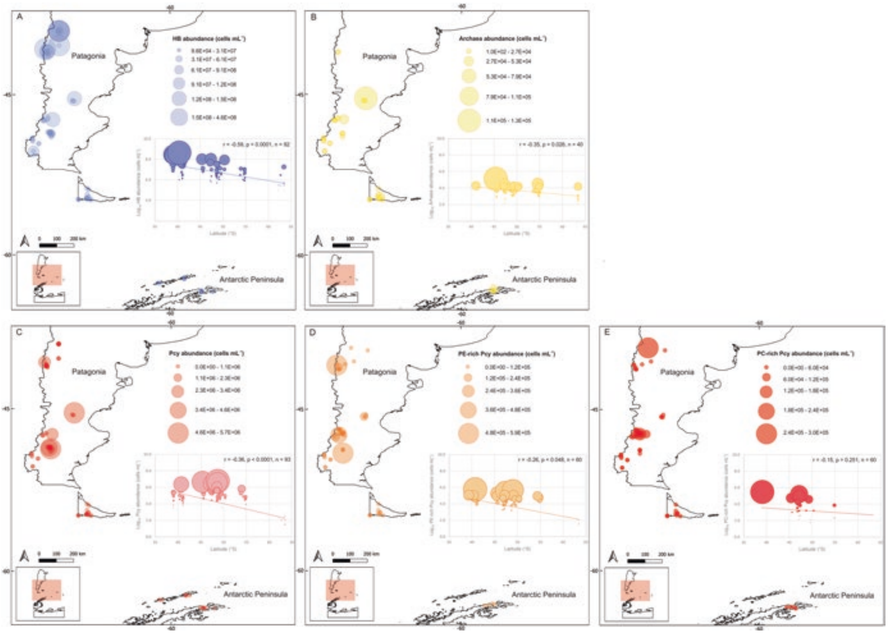

**Fig. 5.2** Maps showing the location of the Patagonian and Antarctic lakes with data of (**a**) heterotrophic bacteria (HB), (**b**) Archaea, (**c**) Picocyanobacteria (Pcy), (**d**) phycoerythrin-rich Pcy (PE-rich Pcy) and (**e**) phycocyanin-rich Pcy (PC-rich Pcy) abundances obtained from literature (HB and Archaea from Schiaffino et al. 2011; Schiaffino et al. 2016a; Saad et al. 2013, 2016; Quiroga et al. 2013; Bastidas Navarro et al. 2014; Gerea et al. 2019; Castro Berman et al. 2020 and Pcy from Schiaffino et al. 2011, 2013; Saad et al. 2013, 2016, 2019; Gerea et al. 2019; Porcel et al. 2019; Castro Berman et al. 2020). Archaea abundances were obtained by CARD-FISH, HB and Pcy abundances by epifluorescence and PE-rich and PC-rich Pcy by flow cytometry. The sizes of the circles on the maps are proportional to the prokaryote abundances. The bubble scatter plots show the relationship between prokaryote abundances versus latitude, and correlations were performed using Spearman rho

altitude would result from the decreasing in total dissolved phosphorus and DOC concentrations above the timberline, resources playing an important role in shaping microbial communities. On the other hand, regarding the environmental gradients, Zunino and Diaz (2000) studied 17 North Andean Patagonian water bodies covering an oligo-eutrophic range, finding that Pcy were numerically more abundant than Peuk. Additionally, PE-rich Pcy dominated in the oligotrophic lakes, whereas PC-rich Pcy dominated in the more eutrophic water bodies. Other authors observed that in North Andean Patagonian ultraoligotrophic deep lakes, around 30% of total chlorophyll-a (Chl-a) can be assigned to Pcy (Modenutti and Balseiro 2002; Callieri et al. 2007). Some other studies on photosynthetic picoplankton (PPP) were also performed in shallow lakes and deep oligotrophic lakes from North Patagonia (Callieri et al. 2007; Bastidas Navarro et al. 2009; Caravati et al. 2010; Callieri et al. 2013) and South Argentinean Patagonia and Tierra del Fuego (Schiaffino et al. 2013). In particular, Callieri et al. (2007) carried out a summer survey in six deep ultraoligotrophic Andean lakes and showed that the PPP was composed exclusively by *Synechococcus* spp. in five of them. Additionally, in the study carried out by Saad et al. (2013) in 12 water bodies located in Southern Patagonia (Tierra del Fuego), it was observed that in general Pcy were more abundant than Peuk, except in humic lakes and beaver ponds. As previously observed, the authors found that PE-rich Pcy were more abundant in the oligotrophic lakes, while PC-rich Pcy in humic and turbid shallow lakes.

The lakes from the Patagonian Plateau (e.g. Strobel and Buenos Aires) have been less studied in comparison with those of the Andean Region, possibly because of their remoteness (Izaguirre and Saad 2014). In a study performed at the basaltic Buenos Aires Lake Plateau (Santa Cruz Province, Argentina) by Porcel et al. (2019) in 16 freshwater bodies, it was observed that the picoplankton structure was associated not only with the trophic gradient of lakes but also to the lake regime and hydrological conditions. As previously reported, the authors found higher abundances of HB (both high and low nucleic acid content obtained by flow cytometry) in more turbid and eutrophic lakes. Clear vegetated oligotrophic lakes showed the highest abundances of PE-rich Pcy, whereas the PC-rich Pcy appeared in clear vegetated, inorganic and organic turbid lakes. Besides, clear vegetated lakes with reductions in the water levels showed an increment in HB, PC-rich and Pcy abundances. Another study conducted by Saad et al. (2019) in 14 freshwater bodies from the Strobel Plateau aimed to analyse the planktonic autotrophic communities in relation to different lake regimes and presence/absence of fish. The authors found that PE-rich Pcy were present in all the studied lakes, but only fish-stocked lakes (phytoplankton turbid) hosted PC-rich Pcy. Additionally, fish-stocked lakes were dominated by cyanobacteria. Autotrophic picoplankton abundances differed significantly among lake types ($H = 7.08$, $p = 0.03$).

## 3 Eukaryotic Assemblages and Their Patterns

### *3.1 Morphological-based and Functional Diversity of Nano- and Micro-phytoplankton*

The geographical patterns in phytoplankton diversity (size fraction >2 μm) were analysed in some papers, covering a latitudinal gradient of lakes along Argentinean Patagonia and also including Antarctica (Maidana et al. 2005; Tell et al. 2011; Izaguirre et al. 2016). These studies were based on the morphological identification of the species by light microscopy, and mainly explored the latitudinal variation in species richness either for particular taxonomic algal groups or for the whole phytoplankton community, and also analysed the influence of the environmental factors on the structure of the algal assemblages. In general, these studies described a decreasing biodiversity pattern with increasing latitude, in accordance with the geographical pattern observed for different groups of organisms (e.g. Willig et al. 2003 and cites therein, Hillebrand 2004; Pimm and Brown 2004), while some exceptions and complex patterns of richness were also found for other microorganisms (e.g. Chown and Convey 2007 and cites therein).

In the meta-analysis carried out by Maidana et al. (2005), data of 85 lentic water bodies were included, covering a transect along the southern end of Santa Cruz province, South of Tierra del Fuego and four sites located in the region of Maritime Antarctica. The study was particularly focused on the diversity of diatoms, reporting a total of 279 infrageneric taxa. A high number of species were shared between Patagonia and Antarctica, whereas only a small percentage of the species found in Antarctica had never been reported in Patagonia. These findings support the hypothesis that most of the diatoms found in the sampled Antarctic areas would have dispersed from southern South America, which is the closest source. The study also showed a geographical pattern in species richness, with a decrease in the number of taxa from Patagonia to Antarctica (latitudinal gradient), identifying a total of 218 species in lakes of Santa Cruz and Tierra del Fuego, and 101 in the four studied Antarctic zones (Deception Island, Potter Peninsula, Hope Bay, Cierva Point). Despite the differences in the environmental conditions between the surveyed water bodies, the similarity in the diatom assemblages was influenced by their geographic position. In a cluster analysis, the species composition observed in the two Patagonian provinces showed more similarity to each other than with the Antarctic places; in turn in Antarctica, the two areas corresponding to the Shetland Islands separated from the two areas located in the continent. This study revealed a high proportion of cosmopolitan species of wide geographic distribution, which was attributed to their capacity to form resting stages and a higher resistance to stress during the dispersion. On the other hand, recent taxonomic revisions of the diatom flora from the sub-Antarctic islands and Maritime Antarctica have revealed that the cosmopolitan nature of non-marine Antarctic diatoms was overestimated (Van de Vijver et al. 2002; Zidarova et al. 2016). Many more recent works have revealed new diatom species endemic to continental Antarctica which have been historically misidentified as cosmopolitan or European taxa (Bishop et al. 2019 and cites therein).

The spatial variation in diversity was also analysed for another algal group (Chlorococcales, Chlorophyceae) along a latitudinal Patagonian transect by Tell et al. (2011). The study was based on two different approaches: a review of the historical records of Chlorococcales in Patagonian freshwater systems and the analysis of phytoplankton samples collected during two spring/summer campaigns (2007 and 2008) in 33 lakes located along a transect from Chubut Province (45.37°S) to Tierra del Fuego Province (63.41°S). The historical analyses accounted for a total of 308 species of this group recorded over a 60-year period. The analysis of the samples collected in 2007–2008 showed a decrease in the species richness of chlorococcaleans with latitude, with a significant inverse correlation ($r = -0.40$, $p < 0.05$). On the other hand, a clear difference in the species richness of this algal group was found between Andean lakes and lakes of the Patagonian plateau, being significantly lower in the former ones ($p = 0.0022$); the contrast in the representativeness of this group in both Patagonian regions seems to be associated to the trophic status of the water bodies, which are mostly oligotrophic in the Andean region and meso to eutrophic in the steppe. Indeed, the relationship with the trophic status was proved with direct significant correlations between species richness and some variables associated with lake trophism (Chl-a, dissolved inorganic nitrogen, dissolved phosphorus and pH). Overall, these results evidenced that the patterns in diversity are influenced by both the geographical position of the water bodies, as well as by their environmental conditions. Regarding the species composition, even in this case 80% of the registered taxa were worldwide distributed, but the assemblages showed more similarity with other cold regions of the Northern Hemisphere than with regions of South America with warmer climates; this is explained by the high dispersion of propagules of micro-algae around the world, which prosper where the environmental conditions are favourable. Accordingly, microorganisms are generally characterised by large absolute population sizes, small sizes and short generation times, and also high dispersal capabilities (by atmospheric transport, migrating birds, marine mammals and human vectors). For micro-algae to establish and develop at a particular place, it first has to arrive, so dispersal is a key factor in their diversity (Broady 1996). The deep analysis of the diversity of Chlorococcales in Patagonian freshwaters allowed the identification of species only reported for Patagonia, and even some probably endemic, which support the 'moderate endemicity model' (Foissner 2006, 2008) for protists. In this sense, several studies have suggested that Patagonia could be considered as an endemism hotspot for protists (e.g. Fernández et al. 2015; Küppers et al. 2016; Coesel et al. 2017).

In a further study, the latitudinal patterns in biodiversity were analysed for the whole phytoplankton community using a polyphasic approach of the diversity (morphologically based, functional and molecular), covering a gradient of 2150 km from Austral Patagonia to Antarctica (Izaguirre et al. 2016). The analysis encompassed 60 freshwater lentic ecosystems with different trophic status, which were grouped into seven different zones, from North to South: steppe of Chubut province; NW of Santa Cruz province; Strobel plateau; SW of Santa Cruz province; Tierra del Fuego; Potter Peninsula at King George Island; Hope Bay at Antarctic Peninsula. Microscopical analyses allowed to identify a total of 321 phytoplankton species, whereas 26 functional groups were registered using the classification of Reynolds et al. (2002), updated by Padisák et al. (2009). The latitudinal analysis of the diversity revealed that both local and regional phytoplankton species richness (LSR and RSR, respectively) decreased with increasing latitude, being the spatial pattern stronger for the RSR; the corresponding values of the regressions performed were $r^2 = 0.188$, slope $= -0.611$, $p = 0.001$ in the case of the RSR; $r^2 = 0.487$, slope $= -0.152$, $p = 0.036$ in the case of the LSR. Using a partitioning analysis of the variance, the study also showed that both spatial and local (environmental) factors are important in shaping the structure of the community. Particularly, lake trophic status was found to exert a high influence on the composition of algal assemblages, since the proportion of taxa potentially phagotrophic were more abundant in oligotrophic lakes than in mesotrophic or eutrophic systems. This was evidenced by a regression analysis that showed that the strictly autotrophs/mixotrophs ratio significantly increased towards higher values of Chl-a as a proxy of lake trophic status. However, although mixotrophy is usually associated with oligotrophic environmental settings, absorbotrophic mixotrophy (osmotrophic mixotrophy with uptake of soluble organic matter found in the environment) is commonly found in meso- to eutrophic waters where organic matter abounds (Selosse et al. 2017 and cites therein).

The influence of lake trophic conditions on the phytoplankton structure was also investigated for another set of Patagonian lakes from Santa Cruz and Tierra del Fuego provinces in relation to DOC and Chl-a (Saad et al. 2016). In this case, the study was focused on mixotrophic species that are capable of having phagotrophic nutrition, but discriminating between taxa according to their predominant type of nutrition, from mainly heterotrophic (such as chrysophytes) to mainly photosynthetic (such as cryptophytes). Although mixotrophic species were represented all along the lake trophic spectrum, primarily heterotrophic taxa attained higher proportions in oligotrophic lakes, whereas primarily autotrophic taxa dominated in eutrophic water bodies. Chrysophytes were best represented in lakes with lower Chl-a and DOC, whereas cryptophytes had higher proportions in enriched lakes. Also, the bacterivory activity of the dominant mixotrophic taxa was analysed in four selected lakes by performing ingestion experiments with fluorescently labelled bacteria, revealing that cell-specific grazing rates were higher in oligotrophic lakes.

## *3.2 Molecular Diversity Patterns*

Diversity patterns of Patagonian freshwater microbiota were also explored using molecular techniques, mainly in the case of the nano and picoplankton fractions, whose identifications are very difficult or impossible by their morphology. The use of different molecular tools has increased the knowledge of the microbial diversity of eukaryotes (Epstein and López García 2008), and particularly the application of high-throughput sequencing technologies allowed to recover even the rarest taxa (e.g. Logares et al. 2014; Lara et al. 2015; Izaguirre et al. 2021).

The distribution patterns and factors driving diversity gradients in microbial eukaryotes were studied along a latitudinal gradient of lakes from Patagonia to Antarctica by Schiaffino et al. (2016b), including a sub-set of 40 lakes of those analysed in the article published by Izaguirre et al. (2016) previously mentioned. The biodiversity of microbial eukaryotes was analysed combining two methods that together allow a better characterisation of the community composition: on the one hand, the fingerprinting technique DGGE that mainly recovers the dominant OTUs which would conform the ‘core biosphere’ and, on the other hand, high-throughput sequencing (IlluminaHiSeq), which allows to obtain the total community composition, including the abundant, the uncommon and the rare OTUs. The analysis was based on the size fraction of 0.2–3 μm. Regarding the latitudinal gradient in biodiversity, DGGE and the more abundant Illumina OTUs showed a clear decreasing richness with latitude; however, for the total microbial richness obtained by IlluminaHiSeq this spatial pattern was not observed. In the same sense, when only the abundant OTUs are considered, Patagonia and Antarctica exhibited significant differences, being the ‘core biosphere’ larger in Patagonian lakes than in Antarctic ones, while the contrary occurred for the ‘rare biosphere’. On the other hand, the ordination method applied (Redundancy analyses) based on community composition versus environmental and geographic variables showed a clear separation between Patagonian and Antarctic lakes.

The study also evidenced high values of beta-diversity both for DGGE and Illumina, mainly due to OTUs turnover. The main driver of beta-diversity along the latitudinal gradient was explored by partial Mantel test and variation partitioning analyses on the more abundant OTUs (DGGE and Illumina), showing a joint significant effect of geographical and environmental factors ($p < 0.05$). Using the same dataset here, we illustrate the result of the Mantel test for the DGGE data ($r = -0.25$, $p = 0.002$; $n = 40$), which shows a declining similarity of the eukaryotic community composition similarity with the geographical distance, while keeping environmental variables constant (Fig. 5.3).

The high-throughput sequencing analyses (IlluminaHiSeq) performed with samples collected in Patagonian lakes along the latitudinal transect allowed to find the first freshwater organism belonging to the family Bathycoccaceae (Chlorophyta; Archaeplastida), a group that only had records of marine species (Lara et al. 2017). This interesting finding led to a deeper molecular study to investigate the role of the two non-mutually exclusive models ‘isolation by distance’ and ‘isolation by

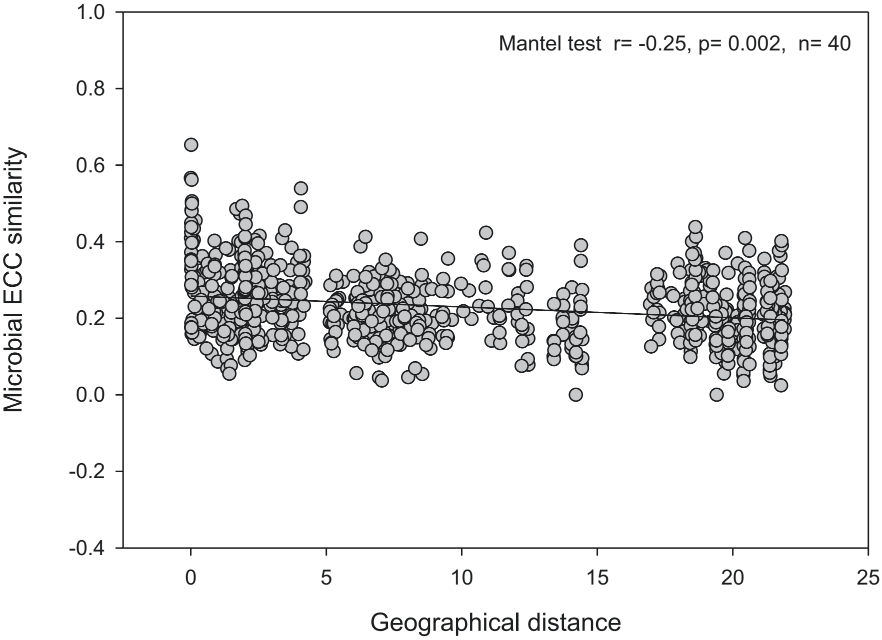

**Fig. 5.3** Relationship between eukaryotic community composition (ECC based on DGGE band intensity) similarity and geographical distance obtained from Schiaffino et al. (2016b). Correlation value shows the results from partial Mantel Test, while holding environmental matrix constant

environment' in shaping the genetic structure of lacustrine populations of this microalga (Fernández et al. 2017). The first model postulates that geographical distance and barriers restrict gene flow among populations, whereas the second model proposes that local environmental conditions select against migrants thus limiting the gene flow among populations (Fernández et al. 2017 and cites therein). The study included samples of a sub-set of six Patagonian lakes located in the provinces of Chubut, Santa Cruz and Tierra del Fuego (Musters, Posadas, Cardiel, del Desierto, San Luis, Yehuin), where freshwater members of the family Bathycoccaceae had been registered. According to the results obtained, isolation by distance accounted for 38.5% of the genetic structure, whereas isolation by environment accounted for 17.7%. As these models (alone or together) could not explain the entire differentiation observed, the unexplained variation in the genetic differentiation was explained in terms of the 'monopolisation hypothesis' (De Meester et al. 2002), which postulates that the genetic differentiation between populations can be explained by a rapid population growth and local adaptation after historical founder effect, resulting in the effective monopolisation of the resources.

Along the Patagonian–Antarctic transect the molecular diversity of the size fraction 3–20 μm (nanoplankton) was also studied using DGGE, observing a strong decrease in the OTUs richness across the latitudinal gradient ($r^2 = 0.39$, slope $= -0.92$, $p < 0.0001$), but no clear pattern in evenness was found (Izaguirre et al. 2016). The standardised correlation coefficient between band richness and latitude was $r_z = -0.73$ ($p < 0.05$). Based on canonical variation partitioning, the study showed that both environmental and spatial factors influenced the molecular diversity of eukaryotes for this size fraction.

The aforementioned works focus on different molecular tools to study biogeographical patterns. In microbial biogeography, taxonomic resolution is key and tools with different resolution power could provide different knowledge and information. High-throughput sequencing approach gives obviously more information than fingerprinting techniques because both PCR-based approaches explore different taxonomic levels and resolutions. Therefore, it is notable that when comparing the microbial patterns obtained for the dominant microbial communities with these two very different approaches, some authors found comparable results (e.g. Pommier et al. 2010; Schiaffino et al. 2016b). Whether or not the same patterns emerge when studying not only the dominant but also the rare microbial composition (using for example high-throughput sequencing technologies) remains an open question, as not many studies using this approach were performed in Patagonian lakes.

## *3.3 Abundance Patterns*

The abundance and the cytometric group diversity of Peuk was investigated along the latitudinal gradient in the same Patagonian lakes from Chubut to Tierra del Fuego provinces (Schiaffino et al. 2013). A total of 41 cytometric populations of Peuk were identified for the 32 studied water bodies. The abundance of Peuk increased with lake trophic status, registering the highest values in mesotrophic and eutrophic environments and the lowest ones in the oligotrophic lakes; mean values (in cells $mL^{-1}$) were $4.0 \times 10^3$, $9.5 \times 10^3$ and $1.3 \times 10^4$ for oligotrophic, mesotrophic and eutrophic lakes, respectively. The relationship between the Peuk abundance and trophic status was proved by a significant positive correlation with Chl-a ($r = 0.62$, $p < 0.001$). On the other hand, the study confirmed a similar pattern to that described by other authors (e.g. Craig 1987; Pick and Agbeti 1991; Vörös et al. 2009), who found a higher contribution of Peuk in less transparent lakes; indeed, in the studied Patagonian lakes, Peuk abundance was positively correlated with $K_d$ ($r = 0.39$, $p < 0.05$).

The study conducted by Schiaffino et al. (2016b) across the latitudinal Patagonian–Antarctic transect showed significantly higher abundances of autotrophic and heterotrophic eukaryotes in Patagonian than Antarctic lakes ($p < 0.001$ and $p < 0.01$ for autotrophic and heterotrophic, respectively). The epifluorescence analyses yielded mean abundances of autotrophic eukaryotes of $7.7 \times 10^4$ cells $mL^{-1}$ in Patagonian lakes and $5.4 \times 10^3$ cells $mL^{-1}$ in Antarctic lakes; for heterotrophic eukaryotes values were $1.1 \times 10^4$ cells $mL^{-1}$ and $2.2 \times 10^3$ cells $mL^{-1}$ in Patagonia and Antarctica, respectively.

In Fig. 5.4, we integrated the available information on phytoplankton obtained in different studies in Patagonian and Antarctic lakes (Díaz et al. 2000; Saad et al. 2013, 2016; Izaguirre et al. 2016; Gerea et al. 2017, 2019; Schiaffino et al. 2016b;

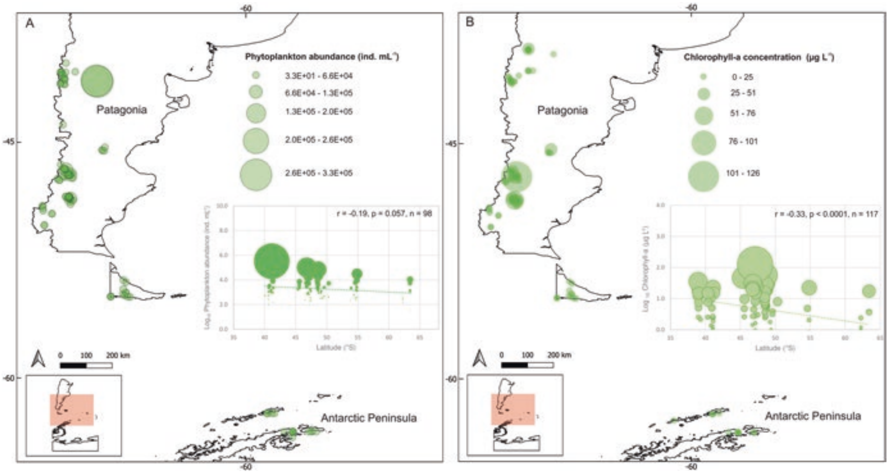

**Fig. 5.4** Maps showing the location of the Patagonian and Antarctic lakes with data of (**a**) phytoplankton abundances and (**b**) chlorophyll-a (Chl-a) concentrations obtained from literature (Díaz et al. 2000; Saad et al. 2013, 2016; Izaguirre et al. 2016; Gerea et al. 2017, 2019; Schiaffino et al. 2016b; Porcel et al. 2019; Castro Berman et al. 2020). Phytoplankton abundances were obtained by Utermöhl and Chl-a by spectrophotometry. The sizes of the circles on the maps are proportional to the phytoplankton abundances and Chl-a values. The bubble scatter plots show the relationship between phytoplankton abundances and Chl-a values versus latitude, and correlations were performed using Spearman rho

Porcel et al. 2019; Castro Berman et al. 2020). With these gathered data, a nearly significant decreasing trend in phytoplankton abundance with latitude was observed ($r = -0.19$; $p = 0.057$; $n = 98$), whereas using Chl-a as a proxy of phytoplankton biomass, a significant decreasing pattern with latitude was found ($r = -0.33$; $p < 0.0001$; $n = 117$).

## 4 Implications for Management and Conservation

Despite covering a small portion of the Earth's surface, inland waters offer enormous benefits to human society, environmental well-being, economic welfare and provide a range of crucial ecosystem services (Finlayson et al. 2005, Chaps. 2, 9, 11, 18), thus the conservation of these systems together with their biological communities is fundamental in order to prevent the risk of their loss.

Most of the phenomena that are observed in the visible and aboveground world are driven directly or indirectly by microorganisms (e.g., Saccá et al. 2017). Microbial communities are vital to the ecosystem functioning, as microbes are greatly abundant and have an immense cumulative mass and activity, and provide essential nature's contribution to people, such as water purification, climate regulation, nutrient cycling, genetic and biotechnological resources (e.g. Ducklow 2008; Chap. 6). Therefore, biogeographical patterns of microorganisms provide a useful framework for understanding the overall ecology of microorganisms, as well as the ecological services they provide in the natural environment (Cotterill et al. 2007; Hanson 2017). Even though most biodiversity and conservation research has focused on the value and importance of macroorganisms, the huge abundance of microorganisms confers them a prime role in providing ecosystem services. Microorganisms play a significant biogeochemical role in aquatic environments (e.g. Cotner and Biddanda 2002), being nutrient cycling key for the regeneration of nutrient in freshwater systems and the continued growth of photosynthetic picoplankton, phytoplankton, benthic algae and macrophytes (Sigee 2005).

Microbes also have adaptive and physiological variability for local resistance and resilience to environmental changes (Allison and Martiny 2008). Given the high bacterial diversity and ability to respond rapidly to changing environments, these communities may be powerful indicators of environmental stressors (Kraemer et al. 2020). In particular, microbial communities from ice ecosystems and peat bogs have been proposed as sentinels of climate change (Vincent 2010; Mataloni 2016). Besides, it has been found that extremophiles are able to produce biomolecules adapted to their unusual living conditions that may represent valuable sources of novel bioproducts (Sanchez et al. 2009).

For all the aforementioned, Patagonian water bodies, as well as Antarctic lakes are unique environments that support a large and exclusive microbial diversity (cosmopolitan and endemic) that deserve better conservation management to support the wide range of ecosystem services they provide. The study of microbial life associated with these particular aquatic environments opens up new perspectives regarding how communities are adapted and thrive in these remote and extreme environments.

## 5 Final Remarks

Overall, in this chapter, we present an overview of the current knowledge of the Patagonian freshwater microorganisms, providing evidence that support the biogeographical patterns of planktonic prokaryotes and eukaryotes along Patagonian water bodies, including Antarctic Peninsula lakes. The literature cited supports the existence of biogeographical patterns of both pro- and eukaryotic microbes, including latitudinal, altitudinal and environmental gradients. Decreasing microbial richness and abundances were observed with increasing latitude, while changes in microbial composition were also reported. Besides, due to the separation between Patagonian and Maritime Antarctic lakes by the Drake Passage, many species were shared between both continents, while others remained exclusive, thus increasing the evidence that supports the endemism for many Antarctic organisms and the urgent necessity for their conservation.

In addition, not only geographical but also environmental variables shaped microbial abundance and composition. Light conditions and trophic status were important in driving the PPP structure. In particular, PE-rich Pcy were predominant in oligotrophic and transparent lakes, whereas PC-rich Pcy and Peuk in eutrophic and turbid lakes. Figure 5.5 summarises the general patterns described in this chapter.

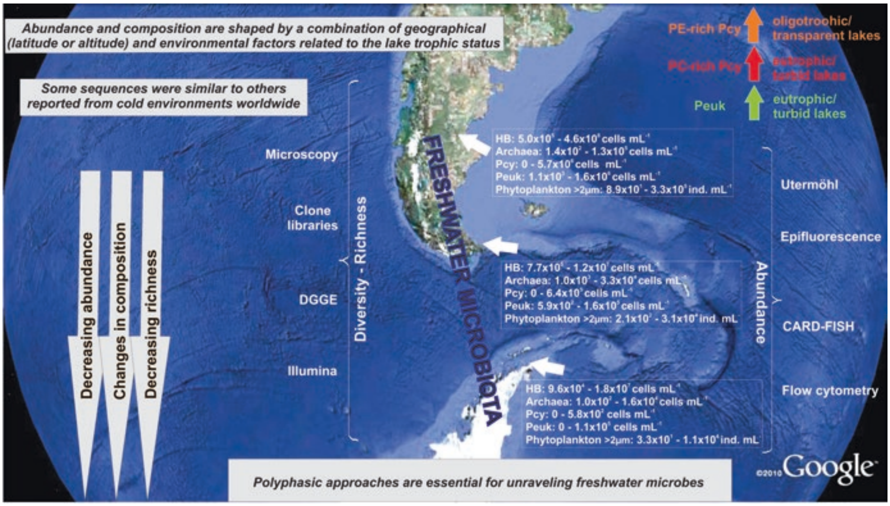

**Fig. 5.5** Figure outlining the principal geographical patterns found for the prokaryotic and eukaryotic microbes along Patagonian lakes, including some Antarctic ones. Image obtained from the Google Earth program (https://earth.google.com)

Undoubtedly, the application of polyphasic approaches is crucial for understanding microorganism diversity and their biogeographic distribution patterns. Even though important knowledge has been attained with the studies already performed in Patagonian water bodies and robust microbial biogeographical patterns have been reported, studies applying novel molecular technologies (such as high-throughput sequencing) are fundamentally needed to deeply explore microbial biogeography in these remote inland waters.

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# Chapter 6
# Diversity Patterns Across Aquatic Communities From Peat Bogs in Changing Environmental Scenarios

**María V. Quiroga, Valeria Casa, Patricia E. García, Gabriela C. Küppers, and Gabriela Mataloni**

## 1 Introduction

### *1.1 The Importance of Peatlands*

Peatlands are a particular wetland type with the capacity of accumulating dead organic material (peat) under permanent water logging conditions facilitated by flat landscapes, low temperatures and abundant, evenly distributed precipitations, met mainly at high latitudes (Joosten and Clarke 2002). They represent 3% of the earth surface and 50–70% of the world wetland area (Domínguez Díaz and Vega Valdés 2015). These environments can have a low decomposition rate, and thus accumulate organic carbon in the peat. Although peat accumulates at an average rate of 1 mm $yr^{-1}$, this process can span a timescale of thousands of years (Harbert and Cooper 2017), and thus the thickness of the peat layer can surpass 10 m over the centuries (Iturraspe 2010). See Table 6.1 for a glossary.

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**Table 6.1** Definitions of terms related to peatlands

| Term | Definition |
|---|---|
| *Wetland* | 'Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres'[a] |
| *Peatland* | Wetland ecosystem subjected to waterlogged conditions dictated by cold weather and abundant, uniformly distributed precipitation, where primary production rate exceeds decomposition rate, leading to the accumulation of organic matter known as 'peat'. |
| *Peat* | Accumulation of dead organic matter with incomplete decomposition. |
| *Mire* | Active area of peat formation and accumulation within a peatland. According to its water supply and trophic state they can be classified in fens and bogs. |
| *Fen* | Minerotrophic mire with a contribution of underground water from the mineral soil |
| *Bog* | Ombrotrophic mire, isolated from groundwater, fed only by precipitations. Most of them have a raised shape and are also called *peat bogs*. |

[a] Ramsar Convention on Wetlands

A healthy peatland sequesters carbon dioxide from the atmosphere and transfers it into the peat (Limpens et al. 2008; Iturraspe 2010). Therefore, one of nature's contributions to people (NCP, IPBES 2019a) provided by peatlands is the mitigation of climate change (Iturraspe 2016), playing an important role in the global balance of greenhouse gases: carbon dioxide ($CO_2$) and methane ($CH_4$). Despite the uncertainties regarding carbon inventory of terrestrial environments (Nichols and Peteet 2021), these wetland types could store up to 30% of the global soil carbon as peat and about 10% of the freshwater (Joosten and Clarke 2002; Limpens et al. 2008). Among other benefits, peatlands can support high biodiversity, contribute to the regulation of floods while preserving the soil from erosion processes, improve the water quality of the basins and provide freshwater for different uses (see Chap. 10; Blanco and de la Balze 2004).

### *1.2 Peatland Succession*

Regarding classification of peatlands, a 'mire' is the active area of the peatland where peat is currently being formed. A minerotrophic mire, also known as 'fen', is a flat or depressed area nourished by mineral soil groundwater (Rydin and Jeglum 2006). Under particular environmental conditions, these peatlands can evolve over time through a process of system succession. Their final development might consist of an ombrotrophic mire, also known as 'bog', which is a dome-shaped elevated area isolated from groundwater, generally dominated by Sphagnaceae mosses (Roig and Roig 2004). Here, precipitations are the only source of nutrient inputs, and moss-driven acidic pH generates dystrophic conditions. The peat bog landscape usually consists of a terrestrial matrix dominated by *Sphagnum* mosses and hosting several water bodies with variable morphometric features. The ponds gradually fill up with sediments and mosses, turning smaller and shallower, until the lentic aquatic habitat disappears and becomes part of the *Sphagnum* matrix. The development of these aquatic systems towards terrestrial conditions modifies their physical and chemical features (Mataloni 1999).

## *1.3 Tierra del Fuego Peat Bogs*

The formation of peat bogs across Patagonia took place in the mountain valleys after the retreat of the glaciers (Iturraspe 2016). In Argentinean Patagonia, 95% of peatlands are concentrated in Tierra del Fuego Province (Rabassa et al. 1996), mainly in Mitre Peninsula, and represent 12.5% of the surface of the Tierra del Fuego Island (Iturraspe 2010). Since the implementation of the territorial planning/land-use zoning of Tierra del Fuego peatlands, which restricts peat exploitation to a particular area, these environments are mainly threatened by urbanisation and climate change (see Sect. 11.4.5; Iturraspe 2016).

The constant growth of Ushuaia population causes an increase in pollution and a greater demand of mains water, acting in concert with the retreat of the glaciers around the city and the decrease in snow accumulation during winters (Iturraspe et al. 2009). In this complex scenario, it is important to estimate how peatlands may be affected by anthropogenic activities in the near future in order to carry out the necessary actions to prevent and mitigate their consequences as soon as possible. Previous studies of Tierra del Fuego peat bogs revealed highly diverse microorganisms uniquely adapted to these extreme wetlands (Quiroga et al. 2013), such as bacteria (Quiroga et al. 2015), planktonic eukaryotes (Lara et al. 2014), microalgae (Mataloni 1999; Mataloni et al. 2015; Casa et al. 2018; González Garraza et al. 2019; Casa 2020), heterotrophic protists (Küppers et al. 2016; Burdman 2019) and microinvertebrates (García et al. 2017). Mataloni (2016) postulates that changes in the diversity patterns of plankton communities from peat bogs would be valuable early indicators of climate change due to their sensitivity and rapid response to environmental changes.

## *1.4 Peat Bog Communities, Nature Contributions to People and Threats*

Peat bogs hold large quantities of freshwater and are capable of mitigating drought-flood events, contributing to water quality and hydrological balance (Joosten and Clarke 2002; Grootjans et al. 2010). *Sphagnum* mosses have the capacity to take up base cations and liberate protons, causing a progressive decrease in pH, conductivity, total hardness and nutrient concentrations as peat accumulates (Clymo 1964), creating distinct environmental conditions. Therefore, the aquatic habitats within peat bogs can be considered as extreme environments due to their low pH values and dystrophic conditions and host highly specialised microorganisms (Gilbert and Mitchell 2006). Thus, peat bogs are appreciated as biodiversity hotspots (IPBES 2019a). Also, they have a key role in the carbon cycle acting as sinks for atmospheric $CO_2$, but global warming and changes in land use could destabilise $CO_2$ and $CH_4$ fluxes to the atmosphere with a potential positive feedback on climate change (Limpens et al. 2008). The processes that drive $CO_2$, $CH_4$ and dissolved organic carbon (DOC) fluxes in peat bogs are almost exclusively mediated by microbes (Limpens et al. 2008; Andersen et al. 2013). As taxonomic identity of microbes can inform about their functional traits, diversity patterns could be potentially linked to ecosystem functioning. In this context, key NCP (IPBES 2019a) provided by peat bogs would relate with the diversity of microbial communities inhabiting them (Andersen et al. 2013; Mieczan and Tarkowska-Kukuryk 2020). For example, the water quality of the Arroyo Grande River, which provides freshwater to Ushuaia City, could be impacted if the biodiversity and functioning of peatlands within the Andorra Valley change.

In Tierra del Fuego, peat bogs are threatened by anthropogenic activities such as settlements on their margins (see Sect. 11.4.5) that pour their raw wastewaters directly into the wetlands, as Epele et al. (2018) observed strong relationships among urbanisation cover and nutrient concentrations in Patagonian wetland waters. The increased nutrient loading in turn alters its ecosystem structure and functioning (Sánchez-Carrillo et al. 2010). In addition, as lotic systems such as rivers are generally the source of domestic wastewaters, they can increase the conductivity and hardness values typically low in undisturbed peat bogs that host a characteristic microbiota (Quiroga et al. 2013, 2015; Lara et al. 2014; Mataloni et al. 2015; Küppers et al. 2016; García et al. 2017). On the other hand, the main climate-related threats that impact microbial diversity, destabilise the food web structure and alter ecosystem processes in peatlands are the rising temperatures (Mieczan and Tarkowska-Kukuryk 2020) interacting with drought (Jassey et al. 2018; Reczuga et al. 2020) and the terrestrialisation of aquatic habitats driven by a lowering water table depth (Mataloni 1999, 2016). Recently, the response of peatland microbes to warming has been studied using in situ experiments. Experimental warming and precipitation reduction, in relation to peatland groundwater level variation, were shown to (i) reduce microbial biomass (Basińska et al. 2020), (ii) generate changes in microbial structure and enzymatic activities (Delarue et al. 2015) and (iii) lead to increased methanogenic rates potentially exacerbating global warming (Wilson et al. 2021). Gunnarsson, Granberg and Nilsson (2004) highlighted that eutrophication, particularly increased N, and higher temperatures may change peatlands currently dominated by *Sphagnum* mosses into vascular-plant-dominated systems. This change in vegetation cover could lead to the loss of moss-associated microbial organisms, impacting the biodiversity of these particular systems.

## 2 Studies in Rancho Hambre and Andorra Peat Bogs

### 2.1 Study Sites

Rancho Hambre and Andorra peat bogs are located in low valleys (altitude 130–200 m a.s.l.) among the ridges of the SW watershed of the Argentinean Tierra del Fuego (Iturraspe and Urciuolo 2000). The area is cold temperate with a monthly mean air temperature of 4.2 °C and a monthly mean precipitation of 60 mm (González Garraza et al. 2012). Both wetlands are ombrotrophic, elevated, dome-shaped bogs. Their landscape consists of a terrestrial matrix dominated by the moss *Sphagnum magellanicum* hosting several shallow ponds (i.e. <2 m deep, Fig. 6.1).

Rancho Hambre (54°47’S, 68°19’W) is located alongside scenic Route J, ca. 50 km away from Ushuaia City, within the protected area of Tierra Mayor Valley (Instituto Fueguino de Turismo 2021). Its domed central area is subdivided into small catchments by lengths of moss hummocks and surrounded by a peripheral

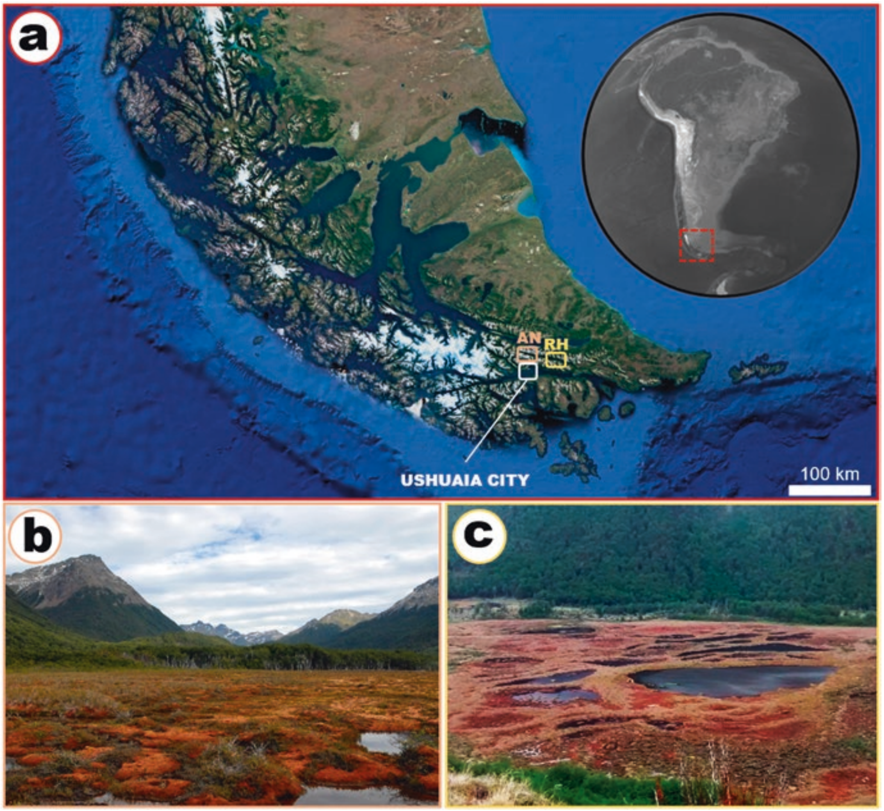

**Fig. 6.1** (**a**) Location of the study area (Tierra del Fuego, Argentina), (**b**) Andorra (AN) peat bog, (**c**) Rancho Hambre (RH) peat bog. The map was generated using Google Earth

drainage network composed of two streams that flow into Lasifashaj River (Grootjans et al. 2010). Despite being a protected area, in recent years, a number of buildings have been irregularly established in Rancho Hambre. These settlements do not have any basic services, such as electricity, mains water or sewage systems, and therefore constitute a threat to the peatland ecosystem (see Sect. 11.4.5). Andorra peat bog (54°45’S, 68°20’W) is located in the Andorra Valley, about 6 km north–west of Ushuaia, within the southernmost Ramsar protected site. This RAMSAR site has an area of 2760 ha and also includes glaciers, lakes and *Nothofagus* forests. Also, the Arroyo Grande River flows through it and is a key drinking water source for Ushuaia City (Grootjans et al. 2010). In view of the delicate balance among these ecosystems’ contributions to people and the present and future use prospects within a global change context, we aimed at analysing the diversity patterns of aquatic algae, ciliate and microinvertebrate communities from these two austral peat bogs in relation to environmental features that could be modified by anthropogenic activities. These diversity patterns were assessed at both local and regional levels, considering (i) temporal variation in ponds from a specific peat bog and (ii) spatial patterns in ponds and mosses from two distant (ca. 50 km away) peat bogs, respectively.

## *2.2 Sampling Design and Analyses*

The local level analysis involved a temporal study of five clear ponds (CP: ponds with a fine sediment covered bottom) located along a transect across Rancho Hambre peat bog. Each CP was sampled on eight occasions during two consecutive ice-free periods (i.e. October–April) from 2008 to 2010 (for details, see González Garraza et al. 2012). Additionally, the superficial, interstitial water of the *Sphagnum magellanicum* matrix was sampled on eight moss sites (SM) in February 2010 (Fig. 6.2a). Nano+microalgae, ciliates and microinvertebrates were sampled at each site as detailed below.

The regional analysis of diversity spatial patterns only involved the microinvertebrates. Here, three types of aquatic environments: five clear ponds (as described above), four vegetated ponds (VP: ponds characterised by having a continuous moss layer growing from the bottom to the surface of the water body) and four moss sites (as described above) were sampled at both Rancho Hambre and Andorra peat bogs on February 2014 (Fig. 6.2; for details see García et al. 2017).

On each sampling event, water temperature, pH and conductivity were measured in situ with multiparametric probes, while total hardness (TH), dissolved organic carbon (DOC) and nutrient concentrations (N and P) were analysed in the laboratory. Detailed methodologies are described in González Garraza et al. (2012) and García et al. (2017).

At each site, one qualitative sample and two replicate quantitative samples of nano+microalgae were taken. The literature used for taxonomic identification and further methodological counting details are described in Mataloni, González

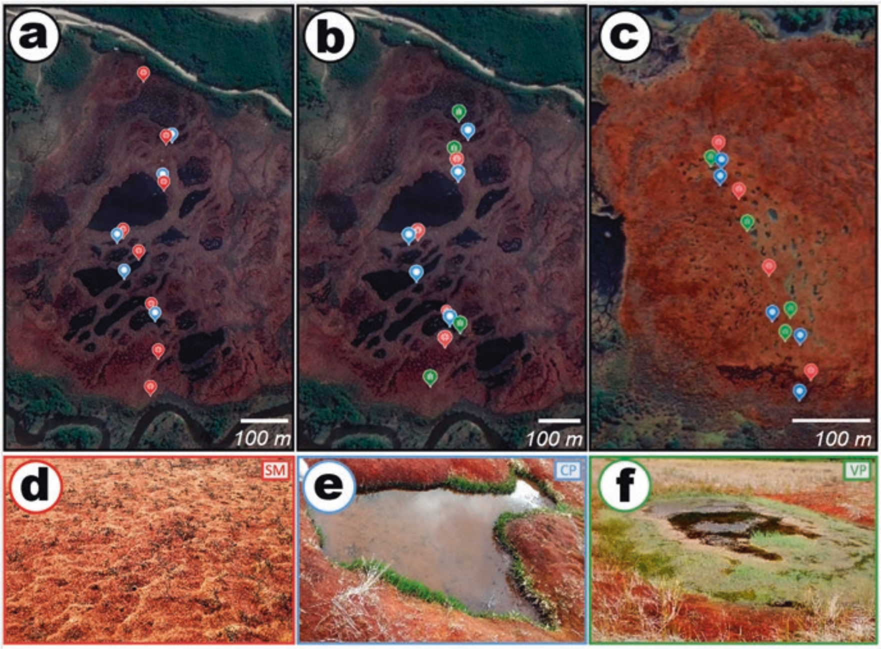

**Fig. 6.2** Sampling sites from: (**a**) Rancho Hambre peat bog during 2008–2010, (**b**) Rancho Hambre peat bog in 2014, (**c**) Andorra peat bog in 2014. (**d–f**) Types of environments sampled (SM *Sphagnum* matrix; CP clear ponds; VP vegetated ponds). The maps were generated using Google Earth

Garraza and Vinocur (2015). For ciliates, two replicate quantitative samples per site were taken. Identifications were based on the comparison of live observations and protargol impregnations of individuals from cultures and Lugol-fixed samples. More details of the sampling methods, quantitative analyses and identification of taxa are amply described in Küppers et al. (2016). Also, Quiroga et al. (2013) and García et al. (2017) give a detailed account of the methods employed for qualitative and quantitative sampling, taxonomic identification and counting of microinvertebrates in the local and regional analyses, respectively. For all communities, only abundant taxa (i.e. relative abundance >3% in any sample) were retained for further analysis.

In order to assess diversity patterns in relation to environmental features, alpha diversity (i.e. Shannon–Weaver index) was evaluated through generalised additive models (GAM; Wood 2017) for each community independently. We used gaussian family (link = 'identity') and implemented cubic regression splines for each environmental predictor. Environmental features tested as predictors were water temperature, pH, conductivity, total hardness and nutrient concentrations (N, P and C forms). A temporal autocorrelation structure of order 1 with days from first sampling at each site as time covariate, corCAR1 (form = ~ time | site), was considered for the Rancho Hambre 2008–2010 dataset. The highest order factors were removed sequentially until we reached the model with the lowest AIC. Model distribution assumptions were checked through quantile–quantile (QQ) plots of Pearson residuals. Indicator taxa were identified with indicator value (IndVal) analysis (Borcard et al. 2018). Samples were grouped based on terciles of environmental features (low, medium and high values), and the IndVal of taxa or associations of up to three taxa was computed using the multipatt function. IndVal index ranges from 0 to 1. Values close to 1 are obtained by a combination of high mean abundance within a group compared to the other groups (specificity, referred to as A) and the presence in most samples of that group (fidelity, referred to as B). For each group, the highest IndVal score was selected. Analyses were performed using *mgcv* (Wood 2017) and *indicspecies* (De Caceres and Legendre 2019) packages in R (R Core Team 2018).

### 2.3 Diversity Patterns of Rancho Hambre Clear Ponds: Potential Impact of Anthropogenic Activities in a Global Change Context

The clear ponds from Rancho Hambre peat bog had been extensively characterised over the recent years. Since Rancho Hambre is an elevated ombrotrophic bog, these lentic systems are fed only by precipitation. Also, some of them have inflow and/or outflow channels but others are hydrologically isolated. This connectivity together with pond size modulates their changes in water level, with smaller, isolated ponds strongly responding to changes in air temperature through evaporation (González Garraza et al. 2012). The interaction among temperature and water level, reflected in the conductivity and nutrient concentrations, generates a seasonal variation of environmental conditions within ponds (Mataloni 2016). This leads to a high spatial and temporal diversity of lentic systems (González Garraza et al. 2012) and their hosting highly diverse microbial communities (Lara et al. 2014; Mataloni et al. 2015; Quiroga et al. 2015; Küppers et al. 2016) and distinct food web structures (Quiroga et al. 2013).

Trends over two annual cycles in the alpha diversity patterns of nano+microalgae, ciliates and microinvertebrates from Rancho Hambre clear ponds are shown in Fig. 6.3. The GAM adjusted $R^2$ was 0.22 for nano+microalgae, 0.40 for ciliates and 0.18 for microinvertebrates, and all smooth terms were significant (all $P$ values <0.04). The Shannon–Weaver index (H') of algae showed negative relationships with temperature and total hardness, while for microinvertebrates it was positively related to conductivity. The H' of ciliates showed a negative trend regarding total nitrogen concentration and a positive relation with temperature until approximately 10 °C, with values oscillating around 1 above that temperature. For each relevant environmental feature, indicator taxa of specific conditions (i.e. low and high terciles) were identified (Table 6.2). For example, *Plagioselmis* sp. and *Stentor* cf. *araucanus* were indicators of high temperature and *Bosmina chilensis* of high conductivity conditions.

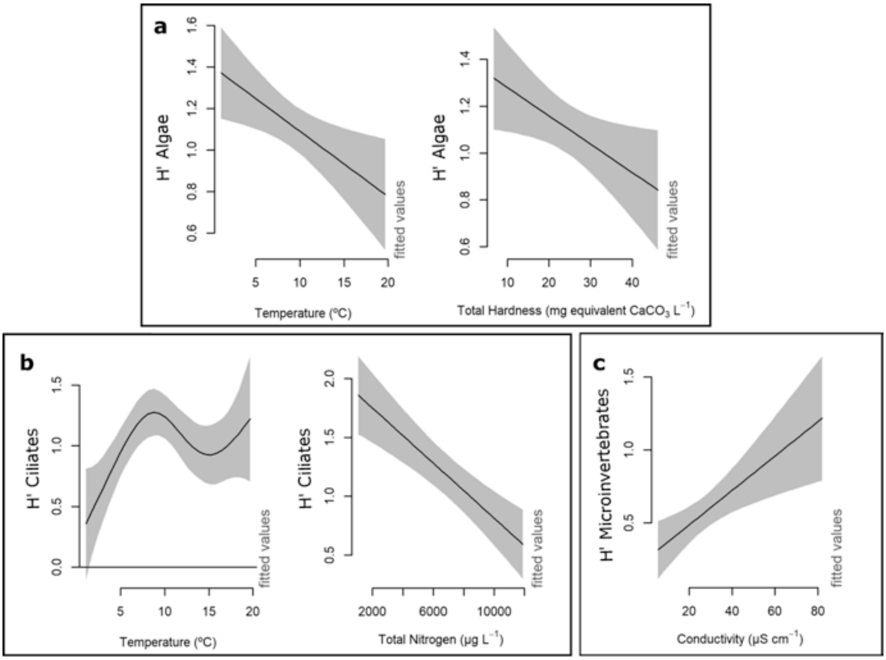

**Fig. 6.3** Relationships between (**a**) nano+microalgae, (**b**) ciliate and (**c**) microinvertebrate alpha diversity (i.e. Shannon–Weaver index, H') and relevant environmental factors in clear ponds within Rancho Hambre (2008–2010). Partial effects of each environmental predictor are shown. Shaded regions represent the 95% confidence intervals of the smooth spline function

Based on these diversity patterns (Fig. 6.3), if the domestic wastewaters poured into the peat bog increase its nitrogen concentration, conductivity and hardness values, this could result in lower diversity of nano+microalgae and ciliates and higher microinvertebrate diversity. Microinvertebrates represent the highest trophic component of these planktonic food webs and can prey on the other two groups (Quiroga et al. 2013). Such trophic relationships are not only driven by taxonomic identity, thus affecting taxonomic diversity, but also by morphological and functional traits, such as cell size, motility and feeding habits among others (e.g. for algae, Kruk and Segura 2012; for ciliates, Šimek et al. 1996; for microinvertebrates, Quiroga et al. 2013). Therefore, understanding trophic links is a complex subject. For microinvertebrates, high conductivity conditions produced more diverse communities, wherein *Bosmina chilensis* was identified as an indicator taxon. This cladoceran coexists with several rotifers including the genus *Keratella* during the austral summer (Modenutti 1994) and could be increasing the microinvertebrate alpha diversity acting as a keystone facilitator (Stachowicz 2001) within its trophic level. In fact, the genus *Bosmina* is a common dweller of the Patagonian inland waters (Menu-Marque and Marinone 1986; Marinone et al. 2006).

Regarding ciliates, they showed higher alpha diversity at low total nitrogen (TN) concentrations, with *Halteria grandinella* (a highly efficient fine suspension feeder)

**Table 6.2** IndVal analyses: best indicator taxa of relevant environmental conditions in clear ponds from Rancho Hambre (2008–2010)

| Variable | Tercile | Range | Group | Indicator taxa | A | B | IndVal (*p* value) |
|---|---|---|---|---|---|---|---|
| Temperature (°C) | Low | 1.10–6.89 | Nano+microalgae | Two *Chlamydomonas* spp. | 0.90 | 0.85 | 0.87 (0.001) |
| | | | Ciliates | *Rimostrombidium hyalinum* | 0.88 | 0.77 | 0.83 (0.010) |
| | High | 10.75–19.67 | Nano+microalgae | *Plagioselmis* sp. | 0.90 | 1.00 | 0.95 (0.003) |
| | | | Ciliates | *Stentor* cf. *araucanus* | 0.72 | 0.69 | 0.71 (0.017) |
| Total Hardness (mg equivalent $CaCO_3$ $L^{-1}$) | Low | 6.81–20.12 | Nano+microalgae | unidentified Chrysophyceae + two *Plagioselmis* spp. | 0.79 | 0.77 | 0.78 (0.002) |
| | High | 26.69–46.20 | | *Eunotia* aff. *exigua* | 0.76 | 0.69 | 0.72 (0.035) |
| Total Nitrogen (μg $L^{-1}$) | Low | 1072–5494 | *Ciliates* | *Halteria grandinella*+ two *Urotricha* spp. | 0.91 | 0.85 | 0.88 (0.001) |
| | High | 7044–11917 | | *Pelagostrombidium fallax* | 0.55 | 1.00 | 0.74 (0.028) |
| Conductivity (μS $cm^{-1}$) | Low | 5.45–20.00 | | – | – | – | – |
| | High | 26.70–82.00 | Microinvertebrates | *Bosmina chilensis* | 0.68 | 0.85 | 0.76 (0.008) |

*A* specificity, *B* fidelity

plus *Urotricha* spp. (raptorial feeders) (Šimek et al. 1996) as indicator taxa. At high TN concentrations, ciliates were less diverse, with *Pelagostrombidium fallax* as indicator taxa. According to Šimek et al. (1996), this ciliate belongs to a group integrated by less efficient fine suspension feeders, coarse filter feeders and detritophages. In addition, *P. fallax* sequesters and temporarily maintains the functional plastids of ingested algal prey, thus combining heterotrophic and acquired autotrophic nutrition modes depending upon nutrient concentration and available algal preys (Modenutti 2014). This shows how changes in environmental conditions influence the abundance of ciliate indicator taxa with different feeding strategies that could further impact the trophic web structure. Furthermore, nano+microalgae were less diverse at higher total hardness values, a potential wastewater-driven scenario in Rancho Hambre, with *Eunotia* aff. *exigua* as indicator taxa. *Eunotia* species are generally found in *Sphagnum*-associated humic habitats (Lange-Bertalot et al. 2011) and appear to prevail in the peat bog ponds even when the algal community reaches its lowest diversity.

Climate change is a threat to wetlands, in general, and peat bogs, in particular (Sect. 11.4.5; IPCC 2019; IPBES 2019b). According to our results (Fig. 6.3), increasing temperatures in Rancho Hambre would lead to drop in algae diversity. Mataloni, González Garraza and Vinocur (2015) had previously reported a negative response of the phytoplankton diversity to temperature, mainly attributed to high summer peaks of the cryptophyceae *Plagioselmis* sp. in the smaller ponds, which showed the highest morphometric-driven temperature values. Our IndVal analysis also identified *Plagioselmis* sp. as indicator of high temperatures, as such conditions allow it to outcompete other algae, thus lowering both the taxonomic and functional diversity of the community, and hence its resilience to further changes (Oliver et al. 2015). Contrarily, a positive temperature–diversity relationship was observed for ciliates. Indeed, at lower temperatures, ciliate communities were less diverse and dominated by small species (10–30 μm sized, Küppers et al. 2016), with the pico-planktivorous *Rimostrombidium hyalinum* as indicator taxa. On the other hand, *Stentor* cf. *araucanus* was identified as indicator of high temperatures. This large (260–630 μm long in vivo, Küppers pers. comm.) mixotrophic ciliate that harbours hundreds of endosymbiotic green algae showed higher abundance in summer and early autumn, particularly in shallow pools, and appears to be related to more diverse communities. Mieczan and Tarkowska-Kukuryk (2020) applied an experimental approach to study the effects of climate warming on the microbial loop of peat bogs. In line with our results, they observed that the increase in temperature led to an increase in species richness and abundance of ciliates, which would potentially result in more diverse communities. As diversity of ciliates and nano+microalgae showed opposite temperature-related trends in Rancho Hambre, further investigations including manipulative experiments would help to understand the potential impact of global warming on the diversity of these interacting communities at the food web level.

## 2.4 *The Sphagnum Matrix Hosts a Distinct, High Biodiversity*

The water environment among *Sphagnum* mosses displays more terrestrial conditions than ponds, a gradual increase in conductivity and decrease in pH characterise the transition from the open water to the drier surrounding mosses (Mataloni 1999; Casa 2020). In agreement, the composition of nano+microalgae, ciliate and micro-invertebrate communities from the Rancho Hambre clear ponds were highly different from those inhabiting the *Sphagnum* matrix (Fig. 6.4) particularly for ciliates, as the two environments shared only 12 out of 52 algal taxa, 3 out of 67 ciliates and 4 out of 23 microinvertebrates. Although this terrestrialisation gradient can span over short distances (3.2 m), resulting in a dramatic drop of the microalgal community richness with strong changes in composition (Mataloni 1999), no differences regarding size, life-form or nutrition mode were observed in nano+microalgae from mosses as compared to ponds.

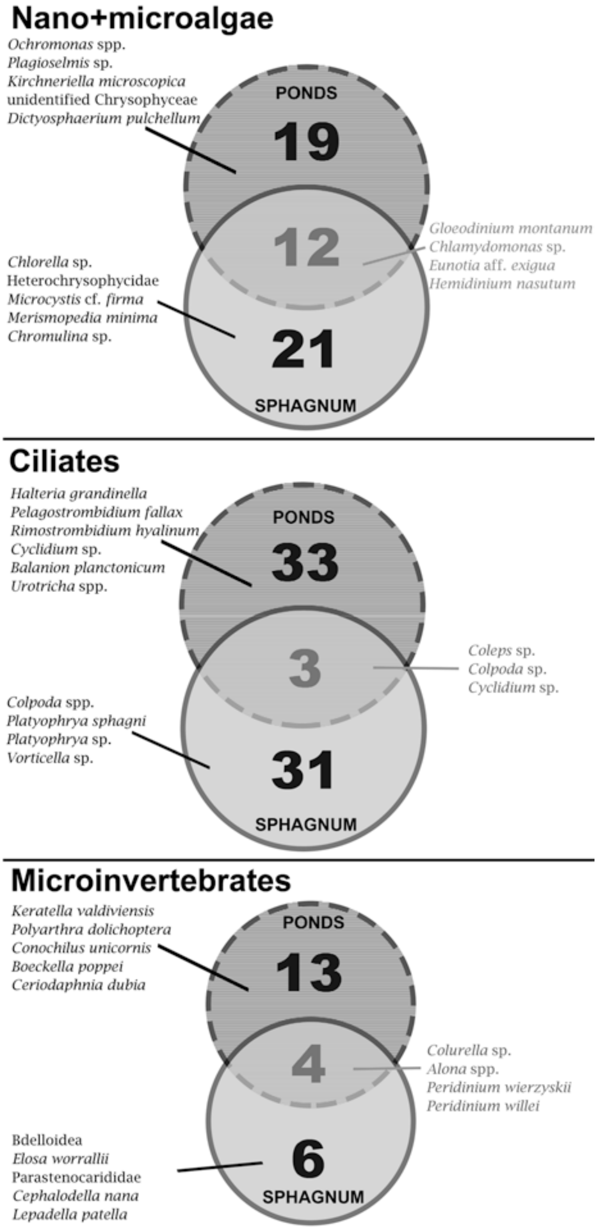

**Fig. 6.4** Venn diagrams showing the number of common taxa among clear ponds and *Sphagnum* mosses in Rancho Hambre peat bog for nano+microalgae, ciliates and microinvertebrates communities. Names: conspicuous taxa in the different habitats

Our analyses showed that ciliate communities inhabiting the *Sphagnum* matrix and the ponds were remarkably different, sharing only three species. Interestingly, the communities from *Sphagnum* matrix resembled those of soils because they were dominated by small- to medium-sized ciliates (less than 60 μm in length), mostly Colpodida and periphytic sessil Peritrichia, while larger euplanktonic ciliates were abundant in the ponds with Peniculia, Prostomatea and Spirotrichea being the most species-rich taxa. This spatial pattern of ciliates size structure was previously observed by Mieczan and Siczek (2010) in a landscape gradient of peat mat – transitional zone – open water zone, with highly abundant small- and medium-sized ciliates in the peat and the highest abundances of medium-sized taxa in the open water. The shifts in ciliate community composition and size-structure are driven by their sensitivity and rapid response to environmental changes (Mieczan et al. 2015; Mieczan and Tarkowska-Kukuryk 2020).

A low diversity of macroinvertebrates has been observed in the peat bogs from Tierra del Fuego, among which the aquatic insects (Diptera: Chironomidae) showed the highest abundances and diversity (Mercado 2004). More recently, García et al. (2017) highlighted that the *Sphagnum* matrix supports highly endemic microinvertebrate species adapted to extreme conditions. Regarding rotifers, Bdelloidea, *Elosa worrallii*, *Cephalodella nana* and *Lepadella patella* dominated the *Sphagnum* mosses, whereas *Keratella valdiviensis*, *Polyarthra dolichoptera* and *Conochilus unicornis* dominated the ponds. In agreement with our present findings, Oloo et al. (2016) observed that Rancho Hambre and Andorra ponds hosted typically aquatic bacterial taxa, while the *Sphagnum* interstitial waters harbour taxa typically characteristic from soils, probably as a result of differences in water chemistry and biotic interactions. Overall, these results highlight the importance of sampling both lentic environments and the *Sphagnum* matrix for assessing aquatic microbial diversity in peat bog ecosystems, in order to include the taxa restricted to a particular habitat type.

## 2.5 The Communities from Different Aquatic Environments: A Regional Study

For this regional study, clear and vegetated ponds and the *Sphagnum* matrix were surveyed at the Rancho Hambre and Andorra peat bogs. Already García et al. (2017) observed that the abundance of microinvertebrate communities inhabiting these aquatic habitats were dominated by rotifers, except for the *Sphagnum* mosses in Andorra, which were dominated by harpacticoid copepods. These authors showed that the aquatic habitat type rather than geographic distance was the key factor explaining differences in species richness and diversity among the microinvertebrate communities. Here, we found that total hardness (TH), dissolved organic carbon (DOC) and pH significantly predicted microinvertebrate alpha diversity (H', Fig. 6.5). In particular for DOC concentration, we observed relatively lower H' mean values below 15 mg $L^{-1}$, and relatively higher H' mean values above that

DOC threshold. A unimodal response was observed for pH, with maximum H’ values around pH 4.3. Remarkably, the Shannon–Weaver index showed a negative relationship with total hardness until approximately 15 mg equivalent $CaCO_3$ $L^{-1}$ and relatively lower H’ values above that hardness level. This GAM model displayed a high adjusted $R^2$ of 0.72 (significant smooth terms, all $P$ values <0.05), and consequently H’-fitted values were really close to the observed ones (Fig. 6.5). These results are in line with Mieczan et al. (2015) mesocosm experiments simulating eutrophication, which showed that DOC and pH significantly affected the assemblage of rotifers and crustaceans.

Additionally, for each relevant environmental feature, indicator taxa of specific conditions (i.e. low and high terciles) were identified (Table 6.3). Regarding pH, we found *Streblocerus serricaudatus* + *Keratella ona* as indicator taxa of high values. The cladoceran *Streblocerus serricaudatus* is a sensitive species that do not dwell aquatic systems with extremely low pH (Walseng et al. 2008), therefore this species could benefit in an increasing pH scenario. According to our results (Fig. 6.5), if raw wastewater disposal increases TH values, we would expect a decrease in the microinvertebrate diversity. Interestingly, the highest microinvertebrate H’ values were observed at low TH conditions with *Bosmina chilensis* identified as the indicator taxon. Thus, this cladoceran appears to be stimulating microinvertebrate alpha diversity at both local (i.e. clear ponds from Rancho Hambre peat bog) and regional scales (i.e. different aquatic habitats in Rancho Hambre and Andorra peat bogs).

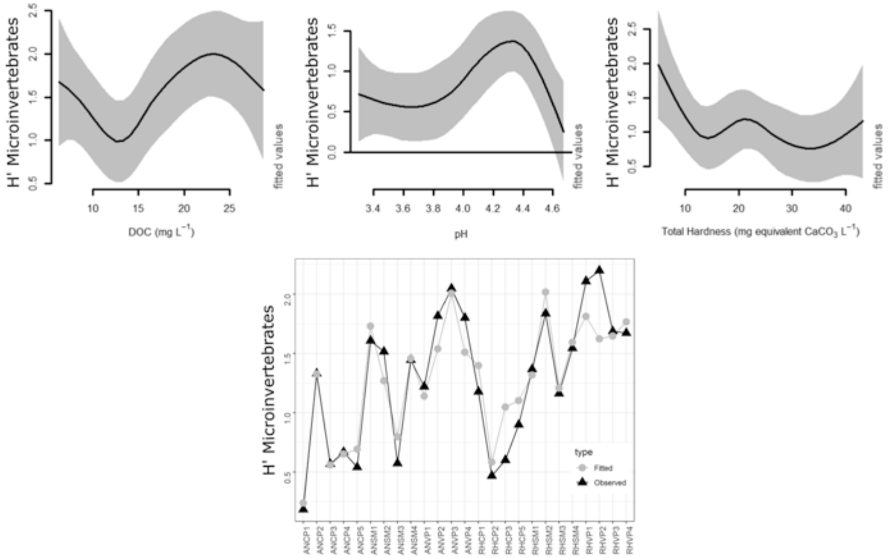

**Fig. 6.5** Above: Relationships between microinvertebrate alpha diversity (i.e. Shannon–Weaver index, H’) and relevant environmental factors in clear and vegetated ponds and *Sphagnum* mosses within Rancho Hambre and Andorra peat bogs. Partial effects of each environmental predictor are shown. Shaded regions represent the 95% confidence intervals of the smooth spline function. Below: Observed microinvertebrate alpha diversity versus GAM-fitted values

**Table 6.3** IndVal analyses: best indicator microinvertebrates of relevant environmental conditions in clear and vegetated ponds and *Sphagnum* mosses from Rancho Hambre and Andorra peat bogs

| Variable | Tercile | Range | Indicator taxa - Microinvertebrates | A | B | IndVal ($p$ value) |
|---|---|---|---|---|---|---|
| Total Hardness (mg equivalent $CaCO_3$ $L^{-1}$) | Low | 4.97–13.99 | *Bosmina chilensis* | 0.88 | 0.78 | 0.83 (0.012) |
| | High | 23.23–43.24 | Cyclopoid nauplii + *Lepadella imbricata* | 0.78 | 0.75 | 0.77 (0.029) |
| DOC (mg $L^{-1}$) | Low | 6.29–12.23 | *Bosmina chilensis* + *Lecane lunaris* | 0.80 | 0.78 | 0.79 (0.039) |
| | High | 18.12–28.78 | Cyclopoid nauplii | 0.96 | 1.00 | 0.98 (0.001) |
| pH | Low | 3.30–3.94 | Harpacticoida | 0.92 | 0.67 | 0.78 (0.008) |
| | High | 4.35–4.67 | *Streblocerus serricaudatus* + *Keratella ona* | 0.90 | 0.63 | 0.75 (0.044) |

*A* specificity, *B* fidelity, *DOC* dissolved organic carbon

## 3 Conclusion

In the Southern Hemisphere, the rising temperature and glacier retreat predicted for Tierra del Fuego region (Iturraspe 2010) threaten to generate drier conditions and decrease water table depth, thus degrading peat bogs through changes in biodiversity that impact its ecosystem functioning (Iturraspe 2010; Mataloni 2016). Parallelly, if the smaller ponds from Rancho Hambre are lost to terrestrialisation, a significant amount of unique and highly adapted biota could undergo local extinctions, greatly diminishing its ecosystem biodiversity. As smaller ponds host unique taxa that do not proliferate in larger ponds, the sum of local extinctions could eventually result in gamma diversity loss (i.e. loss of taxa from the regional pool).

Although peatland microorganisms are known to be driven by vegetation, physical, chemical and hydrological characteristics, we still do not completely understand how their diversity patterns respond to natural and anthropogenic disturbances (Andersen et al. 2013). This study advances the knowledge of Southern Hemisphere peatlands in two ways: (1) By integrating analyses of the nano+microalgae, ciliate and microinvertebrate communities of Tierra del Fuego peat bogs and (2) By modelling their diversity patterns in relation to environmental features sensitive to climate and anthropogenic impacts: temperature, conductivity, total hardness and nutrient concentrations. We found that the local effects of global warming could have contrasting effects, decreasing algal diversity and increasing that of ciliates in clear ponds. Also, the urbanisation-mediated increase of conductivity, total hardness and nitrogen concentration conditions could result in increased microinvertebrate diversity and decreased algae and ciliates H' in Rancho Hambre ponds. At a regional scale, the urbanisation-induced increase of total hardness could potentially decrease microinvertebrate diversity across ponds and the *Sphagnum* matrix. For all the studied communities, we also identified species – or combinations thereof – highly indicative of such changes, which could be used to monitor and early detect impacted ecosystems. With this, we aim at serving governmental institutions to build a decision-making tool for peat bog NCP conservation actions, such as imposing restrictions to urbanisation in order to preserve these wetlands.

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# Chapter 7
# Extreme Freshwater Ecosystems in Patagonia: The Copahue-Agrio System

**Pedro Temporetti, Gustavo Baffico, Mónica Diaz, Guadalupe Beamud, Daniela Nichela, Juan Cabrera, and Fernando Pedrozo**

## 1 Introduction

Numerous lakes in Andean Patagonia are of glacial origin, with great size and depth, and are distributed in a vast area of southern Argentina. Although these lakes are very similar in chemical composition (low salinity, nutrient content) and chlorophyll concentration (Pedrozo et al. 1993), they have different phytoplankton compositions (Diaz et al. 2007) and can be classified as neutral and ultra-oligotrophic lakes. In the arid region of Patagonia (plateau), lakes are shallow, alkaline, and eutrophic (Diaz and Pedrozo 1996). The extremely acidic natural Lake Caviahue stands as a unique case in South America. This lake, located in an old volcanic caldera of the Copahue–Las Mellizas effusive complex in the Province of Neuquén (Pesce 1989), is a particular case, as it has high nutrient concentration and low planktonic diversity and biomass (Pedrozo et al. 2001), which is why it can be considered as oligotrophic.

Acidification of water bodies has at least three origins (Geller et al. 1998): (1) the contribution of sulphuric and nitric acid through acid rain, (2) acid mine drainage in soils rich in pyrite, or (3) contributions of volcanic origin. The last two processes lead to extremely acidic lakes with pH values below 4.0. This causes an increase in solubility that implies high concentrations of inorganic chemical species, for example, Fe and $SO_4^{2-}$ in mine lakes in Germany (Friese et al. 1998), and others such as Al, Cl, and F (Stumm and Morgan 1996; Geller et al. 1998). In addition, potentially

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toxic concentrations of trace elements such as Cu, Pb, Cr, Zn, and Cd can be found (Gammons et al. 2005; Cabrera et al. 2020a).

The progressive acidification of aquatic environments due to anthropogenic factors (atmospheric inputs, mining) is often associated with disturbances that damage ecosystems (Geller and Schultze 2009). One of these alterations involves biodiversity losses and changes in the trophic structure of water bodies (Moiseenko 2005). Naturally acidic aquatic systems (e.g. from volcanic activity) are inhospitable environments with low diversity (few bacterial, fungal, and algal species (Gross 2000)) and simplified biological community structure (Albertano 1995, Beamud et al. 2010).

From the trophic point of view, it is crucial to know which factors control eutrophication in Lake Caviahue acidic waters and the amount of nutrients that are exported by River Agrio to define management criteria in the Upper Basin of River Negro, main water resource of the Alto Valle region and one of the most important hydrographic basins in Argentina (see Chap. 9). Lake Caviahue receives nutrients from both natural volcanic (mostly P) and anthropogenic (containing both N and P) sources and has a range of biological and chemical characteristics that differ substantially from standard trophic models. Therefore, the protection or remediation strategies currently available for use in circumneutral lakes, and which largely feature input nutrient reduction schemes, must be carefully evaluated for application in acidic environments.

Studies of extremely acidic lakes are restricted to volcanic lakes in New Zealand and Japan (Ivanov and Karavaiko 1966; Satake and Saijo 1974). Schindler et al. (1985) and Geller et al. (1998) have studied acidic lakes of anthropic origin (acid rain and mining, respectively). Varekamp et al. (2000) compiled information on volcanic lakes and explored their geochemistry. Many of these works reflect the geographic distribution of volcanic crater lakes (Geller et al. 1998), generally located in remote areas.

In this chapter, the information compiled over 20 years in River Agrio-Lake Caviahue system is analysed in relation to water composition, nutrient load, lake trophic state, algal communities (phytoplankton and epilithon), the geochemistry of the sediments and their role in the dynamics of metals and nutrients. We describe (a) the influence of the feeding basin on the chemical composition and extreme pH of the lake; (b) the limnological characteristics and the changes observed throughout the basin as a consequence of pH fluctuations over time; (c) temporal variations (daily, intra-, and inter-annual fluctuations) and zoning along the water column; d) the behaviour of Lake Caviahue as a Fenton reactor; and (e) the potential use of algae isolated from Lake Caviahue as a bioindicator of the remediation of soils contaminated with aromatic hydrocarbons (PAHs).

## 2 Study Area

The River Agrio-Lake Caviahue system covers three well-defined areas: the Upper Agrio River (UA), Lake Caviahue, and the Lower River Agrio (LA). The region (Fig. 7.1) is located in the Andes between 37–38 °S and 71–71.2 °W of the Province

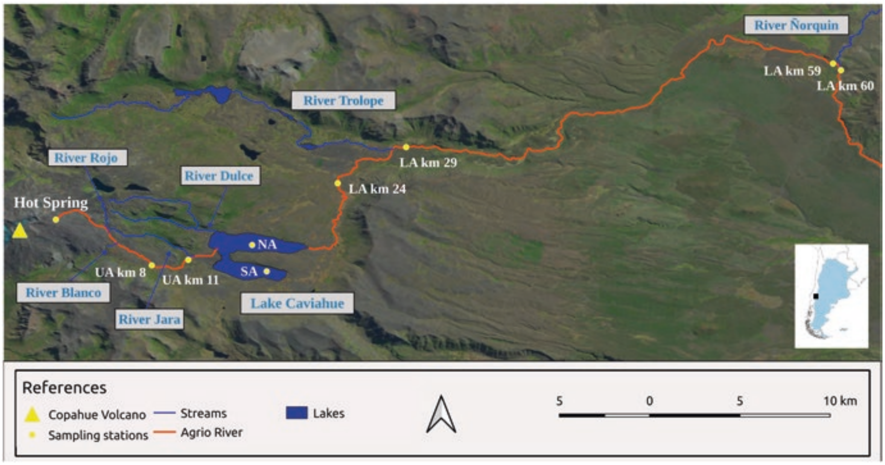

**Fig. 7.1** Location of the study sites (yellow dots) along River Agrio (orange). UA Upper Agrio, LA Lower Agrio, Km distance from Mt Copahue hot springs. The sampling sites (yellow dots) in the lake are NA (North arm) and SA (South arm). River Agrio main affluents are also shown (blue)

of Neuquén (Department of Ñorquín). The predominant vegetation around Caviahue town is the Araucaria or Pehuén (*Araucaria araucana*) forest, surrounded by stunted shrubs of lenga (*Nothofagus pumilio*) and ñire (*Nothofagus antarctica*). The andesitic stratovolcano Mt Copahue lies within the 20 × 15 km large caldera formed by the Copahue–Caviahue effusive complex during the Pliocene-Holocene (Pesce 1989). The volcanic cone is the highest point in the basin (2965 m.a.s.l.) and has a small crater lake.

The UA source is at 2740 m.a.s.l. on the eastern slope of the Copahue Volcano (Fig. 7.1) and is located approximately 200 m below the crater rim (Agusto and Varekamp 2016). The surface of the volcano is covered by a layer of ice and snow that has been decreasing in thickness since July 2000 and December 2012 eruptions, with eruptive pulses in January 2013, March 2014, April 2015, and March 2018. The UA hot springs (Fig. 7.1) are a series of acidic hydrothermal vents on the east side of the volcano. The length of the UA is 13.5 km with 8.4% slope, and there are several waterfalls. Its main tributaries are River Jara and River Rojo on the north bank and River Blanco on the south bank. The UA (1.10 $m^3 s^{-1}$) forms a delta as it empties into Lake Caviahue, providing water with pH between 0.78 and 3.50 (20-year range). A second river, River Dulce (average flow 1.23 $m^3 s^{-1}$) also discharges into the lake with a pH of about 6.5 (Cabrera et al. 2020b).

Lake Caviahue is a large lake carved by glaciers. It is located inside the Caviahue caldera (Agusto et al. 2016) at 1600 m.a.s.l. within the Copahue–Caviahue Provincial Park (37° 50 ‘S and 71° 06’ W). It is an extremely acidic lake (pH 2.0–3.0) (Pedrozo et al. 2001, 2008) due to the influence of the UA. The lake has a horseshoe shape open to the east, presenting two arms: North Arm (NA) and South Arm (SA). Morphometry was described by Rapacioli (1985). Water residence time was estimated between 2.6 (Rapacioli 1985) and 3.5 (Varekamp 2003, 2008) years, based on hydrology and mass balance of conservative elements, respectively.

The last two vigorous eruptions of the Copahue Volcano occurred in July 2000 and December 2012. They were characterised by the emission of ash, lapilli, and bombs that covered the town of Caviahue (975 inhabitants, located on the western bank of the lake). The ashes reached a maximum distance of approximately 100 km from the volcano. Since December 2012, the volcano has remained active. Therefore, ashes were spread all over the basin, and the wind has blown them away and deposited them in the lake. Lake Caviahue effluent is the Lower River Agrio (LA km 24) (4.5 $m^3 s^{-1}$, Rapacioli 1985). At its source, pH is close to 3.0. But after it joins with River Trolope (neutral pH) and leaves the high basin through a canyon that includes the Salto del Agrio waterfalls (LA km 29), pH rises above 3.5 and Fe begins to precipitate massively.

The first indigenous human group to inhabit the Caviahue–Copahue region were the Pehuenches, ethnically associated with the Huarpes from southern Mendoza. Later, the Mapuches had a strong presence since the nineteenth century. Finally, the Criollo-European group occupied the area with the support of the state army (Baschini et al. 2018; see also Chap. 16).

The knowledge of thermal baths and of the healing properties of mud therapy was passed down from generation to generation in the Mapuche community and it was they who named the most emblematic places in the region (Lonac 2018). Nowadays, the Caviahue–Copahue destination is not only renowned for health tourism in the Copahue Thermal Complex but for other natural and cultural attractions such as the Ski Centre, the Copahue Volcano, UA waterfalls, the Salto del Agrio waterfalls on the LA, trout fishing in nearby freshwater bodies (e.g. Trolope, Hualcupen, and Rincón lagoons), and provincial festivals of Craft Beer, *Piñón* (*Araucaria araucana* seed), and Loom.

From spring to early autumn, it is possible to see the 'veranada', the herding of hundreds of goats to higher grazing grounds near the urban area of Caviahue by the Mapuche community Millaín Currical (https://www.interpatagonia.com/caviahue/paseos.html).

## 3 The Upper Agrio River (UA) Basin: The Source of Acidity

UA basin water data of the January 2000 sampling, before the July 2000 eruption, showed that the crater lake was hyper-acidic (pH = 1.16) and its temperature ranged from 5 to 54 °C (Varekamp et al. 2001) due to the influence of both glacier melting and volcanic activity. In that year, the UA hot spring site started flowing 100 m below, on the eastern slope of the Copahue Volcano. This hyper-acidic site (acidity = 1.5 mM) recorded temperatures of up to 82 °C (Pedrozo et al. 2001) depending on the year. Pedrozo et al. (2002) recorded pH values as low as 0.37, relating to the fact that 70% of this water is of magmatic origin. Conductivity had extremely high values (567,488 mS $cm^{-1}$ at 81.3 °C), with very high concentrations of $SO_4^{2-}$ (45,200 mgL), Ca (1230 mg $L^{-1}$), Fe (806 mg $L^{-1}$), and Al (2520 mg $L^{-1}$). Total Phosphorus (TP) concentration was 38 mg P $L^{-1}$, 99% present in dissolved form

(SRP). Ammonium (0.87 mg N-$NH_4^+$ $L^{-1}$) was the main inorganic N species, representing 87% of the Dissolved Inorganic Nitrogen (DIN).

A few hundred meters from the hot springs, UA water cools down maintaining its low pH. While descending the slope, it receives input from tributaries with different chemical characteristics. This leads to a slight increase in pH (Geller et al. 2006; Pedrozo et al. 2008). However, at its mouth in Lake Caviahue, 14 km downstream from the hot springs, the UA maintains a pH ranging between 2 and 3, and high concentrations of dissolved ions (Baffico et al. 2004; Cabrera et al. 2016).

Due to its volcanic origin, the UA is highly influenced by magmatic activity, which defines its main chemical characteristics (low pH and high ion concentrations). In turn, the UA is diluted as a consequence of the higher freshwater inputs (due to ice and snow melting) that occur in winter and spring, which leads to flow increase (Fig. 7.2a). This dilution effect modulates the original characteristics of the UA, causing significant patterns of seasonal variation in pH and conductivity (Fig. 7.2a, b). Downstream, the discharge increases due to snowmelt and the

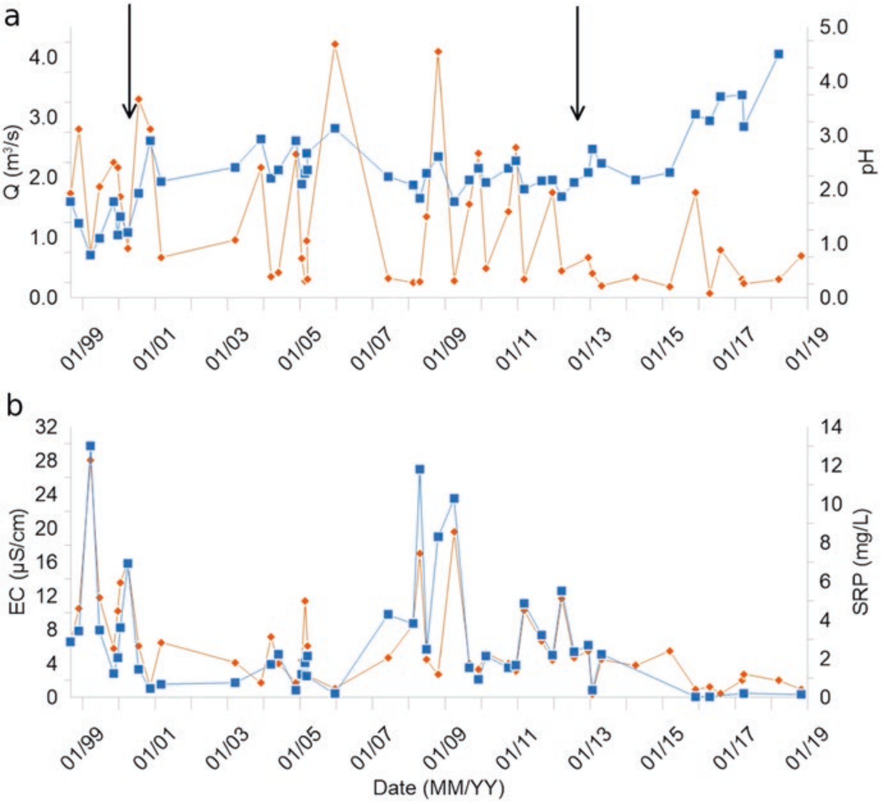

**Fig. 7.2** Variations from 1998 to 2018 at UA Km 11 in (**a**) Flow (Q, orange) and pH (blue), and (**b**) EC (electrical conductivity, orange) and SRP concentration (blue). The vertical arrows indicate the Copahue volcano eruptions of July 2000 and 2012

confluence (Fig. 7.1) with freshwater streams (River Rojo pH ~ 4.0, River Blanco pH ~ 6.0, and River Jara pH ~ 6.0), all of them low in dissolved salts. Due to this dilution, the acidity, conductivity, and transported elements are drastically reduced in the UA before reaching the lake (Geller et al. 2006; Agusto et al. 2012), accounting for a 5 to 20-fold dilution in nutrients and a 13- to 50-fold dilution for other elements like aluminium and sulphur (Pedrozo et al. 2008), which also depends on the degree of hydrological connectivity (Cabrera et al. 2020b). At the discharge to Lake Caviahue, the UA has recorded a range of pH from 0.78 to 3.50 and conductivity from 28.0 to 0.95 mS cm$^{-1}$ over 20 years. Nevertheless, a strong daily cycle is observed in UA because the flow is influenced by snowmelt on the upper basin (Baffico et al. 2004; Pedrozo et al. 2010).

UA flow (Q), pH (Fig. 7.2a), conductivity, SRP concentration (Fig. 7.2b), and other compounds (e.g. N, $SO_4^{2-}$, Fe, Si, Ca) showed a great inter-annual variability (Pedrozo et al. 2008). On the other hand, Pedrozo et al. (2010) also observed a daily variability (7/4/2000, dry season) for example in SRP concentration (ranging from 6.6 to 8.0 mg P L$^{-1}$). UA pH, conductivity, and nutrients were monitored throughout the years 1999–2018 at different frequencies. TP loads were estimated (Pedrozo et al. 2008) from the flows and TP concentration measured between 1999 and 2005. SRP had a close positive correlation ($R^2 = 0.7$; Pedrozo et al. 2010) with conductivity and an inverse correlation with flow. The latter followed a hysteresis pattern due to the effects of the same flow on the concentrations of salts and SRP. This was defined by (1) whether it was in the low or rising water period and (2) due to the existence of a chemical composition baseline given by the contribution of acidic waters from the volcano that feeds the UA. The concentration of ammonium (70–130 µg N-$NH_4^+$ L$^{-1}$) was higher than that of nitrates (30–50 µg N-$NO_3^-$ L$^{-1}$), reaching 800 µg N-$NH_4^+$ L$^{-1}$ in the hot springs (Pedrozo et al. 2001) and as a result, N:P molar ratios were very low (0.1). The exceptions were the UA tributaries with neutral or low pH such as River Blanco and River Jara (N:P = 1.50 and 10.5, respectively). Between 2000 and 2005, P load decreased 10 times whereas pH increased from 2.0 to 3.0. Between 2012 and 2018, a second period of decrease in P load, and an increase in pH from 2.31 to 4.50 was observed (Fig. 7.2a) after main eruption events (see arrows in Fig. 5.2a). These results were experimentally endorsed by Pedrozo et al. (2008), who reported a decrease in SRP when pH values approached neutrality with the addition of an alkaline standard solution ($CaCO_3$).

The UA can be considered an extreme environment, which explains why the epilithic algal community of this entire stretch of the river reveals very low diversity, with *Cyanidium caldarium* in the springs and *Euglena mutabilis* in the river mouth as its main representatives. Both species are frequent in sites with extreme characteristics from different parts of the world (Aguilera et al. 2007; Toplin et al. 2008). Likewise, the epilithic biomass values in the UA are relatively low (maximum of 2 mg Chl am$^{-2}$) and, in winter, practically no biomass is registered. On the other hand, algal development in the UA is subjected to the catastrophic effects of volcanic eruptions. After the eruption of the Copahue Volcano in July 2000, in addition to the direct effect of ash fall (shading and abrasion by transported solids) (Chap. 3), there were changes in the chemical characteristics of UA (a slight increase in pH and a slight decrease in conductivity). So much so that the epilithic algal community sampled in January 2000, represented by *Gloeochrysis turfosa* as the dominant species at that time (Baffico et al. 2004), completely disappeared and no such species has been recorded in the river since then.

## 4 Lake Caviahue: A Lake of Variable pH

Lake Caviahue is a less transparent water body (Secchi depth, DS = 2.3–5.5 m) than other large glacial lakes in Andean Patagonia (e.g. Lake Nahuel Huapi, DS = 22 m, see Chap. 3), with the lower limit of the euphotic zone (1% of the surface PAR irradiance) located at a depth of 16 m (Beamud et al. 2010). Like other deep lakes in the region, it is monomictic but the temperature in the hypolimnion is 2 °C lower, which may be due to its location at least twice as high as (1600 m.a.s.l.) those of most large Andean lakes.

Variation in pH is controlled mainly by the UA, on which the dilution due to the neutral tributaries and the urban wastewater are superimposed. The lake waters are also rich in TFe (20 mg $L^{-1}$), $SO_4^{2-}$ (478 mg $L^{-1}$), $Ca^{2+}$ (28.4 mg $L^{-1}$), and $Al^{3+}$ (40.0 mg $L^{-1}$) because of their acidity. In 2018, Al concentration represented two times that of Fe, and P was found mainly dissolved (90% of TP = 0.5 mg P $L^{-1}$). The concentration of dissolved inorganic nitrogen in the lake ranges between 30 and 150 μg N $L^{-1}$, mainly as ammonia, which is provided by the UA (Pedrozo et al. 2001, 2008). Moreover, total organic nitrogen varies between 32 and 91 μg N $L^{-1}$ (Pedrozo et al. 2008). These values are within the range of those found by Diaz et al. (2007) for neutral Andean lakes (24–223 μg N $L^{-1}$) but they are low compared to those observed by Nixdorf et al. (1998) for acidic mine lakes in Germany (430 μg N $L^{-1}$).

The vertical distribution of nutrients in Lake Caviahue is homogeneous throughout the year. Over the past 22 years, Fe concentration, particularly, fluctuated between 16 and 24 mg $L^{-1}$, the average concentration being around 20 mg $L^{-1}$ (Varekamp 2008; Cabrera et al. 2016; data not published). These variations, like those of other nutrients (ammonium 70 μg N $L^{-1}$), respond mainly to the hydrogeothermal processes of the Copahue Volcano and determine the total metal content in the lake (Varekamp 2008; Cabrera et al. 2016). Lake Caviahue also receives River Dulce (Fig. 7.1) with low conductivity (270 μS $cm^{-1}$), and low or undetectable element and nutrient concentrations, in the range of Andean Patagonian freshwater rivers (Pedrozo et al. 1993).

The multi-elemental relationship regarding that of P (main elements that control the conductivity of the lake) is as follows:

$$\text{Upper Agrio River}: \quad Cl_{639}\mathrm{S}_{510}Fe_{193}\mathrm{Mg}_{190}Al_{160}Ca_{121}Na_{97}\mathrm{K}_{24}\mathrm{N}_{0.3}:\mathrm{P}_1 \tag{7.1}$$

$$\text{Lake Caviahue}: \quad Cl_{137}\mathrm{S}_{301}Fe_{22}\mathrm{Mg}_{37}Al_{90}Ca_{42}Na_{37}\mathrm{K}_{11}\mathrm{N}_{0.3}:\mathrm{P}_1 \tag{7.2}$$

$$\text{Neutral Andean Lakes}: \quad Cl_{62}\mathrm{S}_{128}Fe_{1.3}\mathrm{Mg}_{147}Ca_{511}Na_{246}\mathrm{K}_{36}\mathrm{N}_{3.3}:\mathrm{P}_{1} \tag{7.3}$$

The N load (on a molar basis), mainly in the form of $NH_4^+$, was three times lower than P (as SRP). Compared with the multi-elemental ratios for neutral Andean lakes (3) (Markert et al. 1997), the element proportion in solution is several times higher at UA km 11 (1) due to its acidity. Likewise, the low N:P ratio in Lake Caviahue (2) coincides with the N deficit with respect to P in Andean Patagonian lakes (Diaz et al. 2007).

A zonation of SRP and $Fe^{3+}$ is observed (Fig. 7.3a) in the water column of the lake. The field and experimental results suggest that Fe has a vertical zoning linked to light and that the increase in $Fe^{3+}$ enables P to be precipitated with Fe. Analog processes have been described by Parker et al. (2008) at LA km 59 (Fig. 7.3b), where the daily cycle of $Fe^{3+}/Fe^{2+}$ occurs under light/dark natural conditions. Under the acidic and anoxic conditions of the lake bottom, the eventual presence of $Fe^{\circ}$ would further facilitate the reduction of nitrate. Therefore, the photochemical cycle and the presence of $Fe^{2+}$ would affect the distribution of SRP and dissolved inorganic nitrogen.

The two eruptions mentioned (July 2000 and December 2012) were strongly reflected in the chemical conditions of Lake Caviahue (Pedrozo et al. 2008). In the 5 years after the first event, conductivity decreased from 2.5 to 1.3 mS $cm^{-1}$. At the same time, P concentration was also reduced by half, accompanied by a gradual increase in pH (from <2 to about 3). The second eruption also produced a change in the lake pH (> 3.0) (Fig. 7.4). Upon reaching this pH, an orange layer appeared on the rocks on the lake shore (Fig. 7.5b, c). The precipitates were orange iron oxohydroxides (particularly schwertmannite) and were described by Varekamp (2015), evidencing a highly contrasting situation with respect to the previous two decades (Fig. 7.5a). As a consequence of this precipitation, a reduction in dissolved P was observed. Nevertheless, the average N:P ratio of the lake varied between 0.03 and 0.11 indicating a clear nitrogen deficiency in water. These values are low compared to those cited for mining lakes in Germany (92:1 and 1200:1), where nitrogen is not the factor limiting algal growth (Nixdorf et al. 1998).

Both eruptions resulted in primary pH control in Lake Caviahue. This is a unique situation affecting an Andean Patagonian lake because the vast majority of lakes, if not all, have a constant pH and a buffered system. Between 2000 and 2008, chemical compounds retention decreased by one half. On the other hand, the solubility index clearly showed that most elements remained in solution, although pH increased by around one unit until 2005. In the second phase of pH increase (Fig. 7.2), there was a dramatic decrease in P along with other elements such as Fe and Al (Agusto et al. 2012). In fact, a replacement process occurred. After both eruptions, the lake presented more diluted characteristics, which is why pH increased while conductivity and element concentrations decreased.

In terms of its trophic status, Lake Caviahue can be categorised as an oligotrophic water body due to its average Chl *a* concentration (0.55 mg Chl *a* $m^{-3}$) and, at

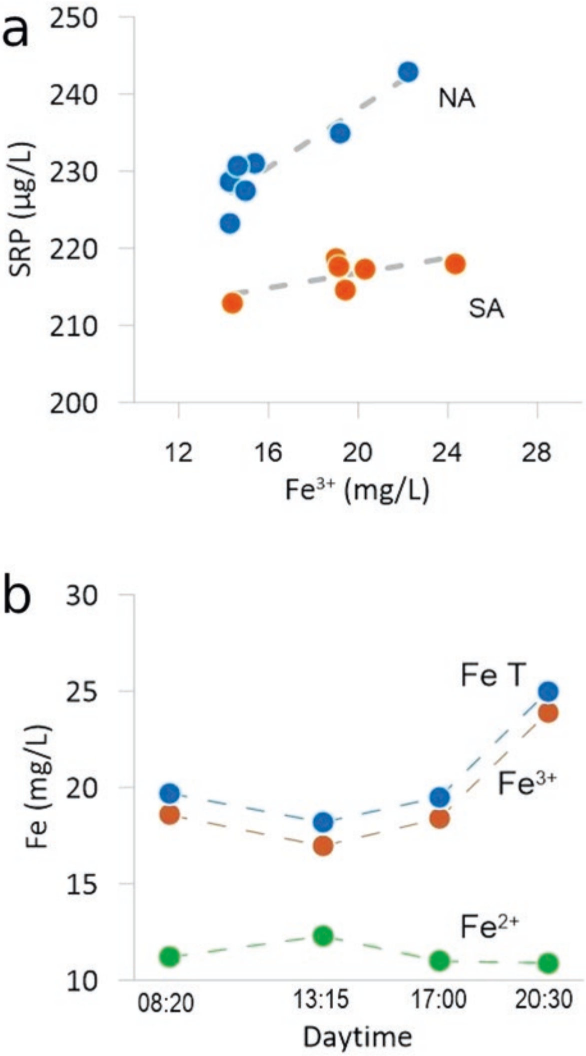

**Fig. 7.3** (**a**) Variations of SRP and $Fe^{3+}$ in the water column of Lake Caviahue at 0, 5, 20, 30, 40, 60, and 85 m depths in North Arm (NA) and 0, 5, 20, 30, 40, and 65 m in South Arm (SA). (**b**) Diel variations in the concentrations of total dissolved Fe, $Fe^{2+}$, and $Fe^{3+}$ in the lower Rio Agrio (LA km 59)

the same time, as mesotrophic considering the average phytoplankton abundance (12,600 cells $mL^{-1}$) or as eutrophic taking into account TP concentration in water (Pedrozo et al. 2001). Thus, a single criterion cannot be applied to define Lake Caviahue because there is no correspondence between nutrients (high TP) and pigments (low Chl *a*) or biomass.

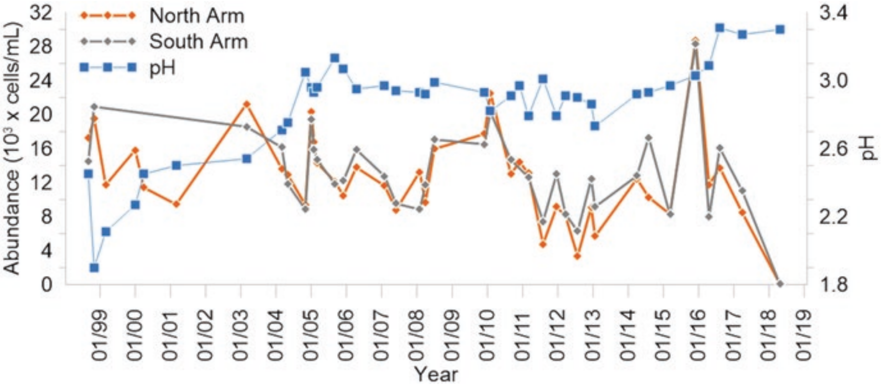

**Fig. 7.4** Phytoplankton abundance and pH variations from 1999 to 2019 in NA and SA arms of Lake Caviahue

## 5 Phytoplankton

Water bodies in volcanic areas are inhospitable: low pH, limited supply of carbon dioxide for photosynthesis and high concentrations of heavy metals. To survive in these extreme environments, algae have to maintain a neutral cytosolic pH within the cell against the high $H^+$ concentrations in water by active proton pumping (Gross 2000). In Lake Caviahue (pH 2–3), algal cells have to deal with proton gradients across the plasma membrane of 1:100000 to 1:10000, respectively.

Lake Caviahue plankton has very low diversity: only one rotifer, *Philodina* sp. (Bdelloidea) and only one phytoplanktonic chlorophyte, *Keratococcus rhaphidioides*, are the dominant species. *K. rhaphidioides* was present in the lake at all depths, in both arms, throughout the year, during the 20-year study (Fig. 7.4), during which there was no seasonal phytoplankton succession (Diaz and Beamud 2014). Another four algal species were occasionally found coming from the UA and the basin wash, but they never represented more than 1% of total abundance: *Watanabea* sp., *Euglena mutabilis*, *Palmellopsis* sp., and *Chlamydomonas acidophila*.

Between 1998 and 2018, volcanic eruptions caused geochemical processes that increased the lake pH from 1.9 to 3.3 (Fig. 7.2a) and, as a consequence, there was a progressive decrease in phytoplankton abundance. After the 2000 eruption, the lake pH remained constantly 2.6 until 2003, when it increased to pH 3.05, during which period phytoplankton abundance halved (from 20,000 to 10,000 cells $mL^{-1}$). The same happened after the 2012 eruption: pH remained a constant 3 until 2014, when it started to increase and reached 3.3 in 2018. Due to this pH increase, $H^+$ concentration decreased by an order of magnitude (from 0.01 to 0.001 M) and produced Al dissolution, Fe precipitation (Stumm and Morgan 1996) and a consequent reduction of available P and N for algal growth, affecting the *K. rhaphidioides* population, whose abundance dropped from 29,000 to just 80 cells $mL^{-1}$ (Fig. 7.4). Fenton processes were also triggered (see Sect. 8.1),

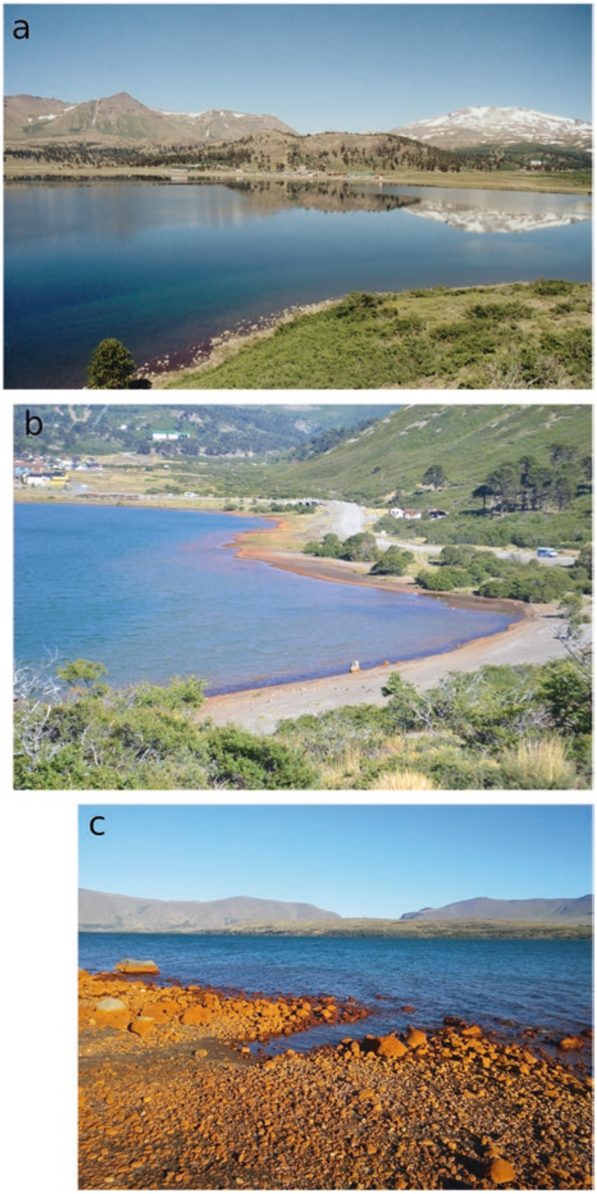

**Fig. 7.5** Views of the Lake Caviahue shore in 2000 (**a**) and 2018 (**b**, **c** detailed view). In (**b**, **c**), Fe mineral deposits are observed with pH increase

which led to the production of hydroxyl radicals ($^{\bullet}$OH) that can cause oxidative damage between the plasma membrane and the cell wall. This would explain growth inhibition correlated with pH increase, as was observed in the lake and in the laboratory bioassays (Cabrera et al. 2021).

From the experiments carried out in the lab and in situ over the years (Beamud et al. 2010, 2014; Schultz 2016; Baffico et al. 2018), it is possible to infer that *K. rhaphidioides* is tolerant to conditions such as low light intensities (12–70 μmol photons $m^2$ $s^{-1}$), very low pH (between 2.0 and 4.0), and high concentrations of different elements, for instance, Fe, Al, and Mn (Cabrera et al. 2021). Besides, the alga is reported as a mixotrophic species (Beamud et al. 2014) with the ability to grow with both organic and inorganic carbon and nitrogen, and phosphorus sources (under both axenic and non-axenic conditions). Laboratory assays showed that algal growth was stimulated with the addition of $CO_2$, glucose, acetic acid, aspartic acid, ammonium, urea, leucine, arginine, glutamine, phosphate, and glycerophosphate. The algal growth experiments carried out in situ in the hypolimnion showed mixotrophic assimilation of organic nutrients (nitrogen and carbon). This result could explain the presence of the algae growing in the dark as a strategy in the absence of inorganic carbon, as it occurs in low pH environments such as Lake Caviahue (Diaz and Beamud 2014). Supporting this hypothesis, low molecular weight carbohydrates were found both in the water column–sediments interphase and the sediment interstitial waters (Cabrera, unpublished). Samples taken with corer at 70 and 90 m depth (Fig. 7.6a, b), containing *K. rhaphidioides* were observed under a microscope with transmitted light (Fig. 7.6c) and with epifluorescence (Fig. 7.6d), showing an apparent good condition of chlorophyll ***a***.

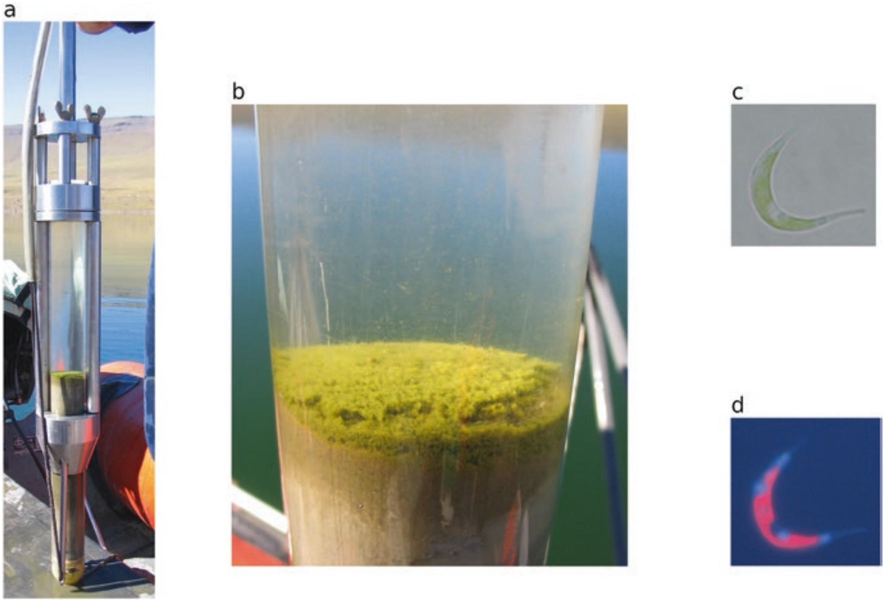

**Fig. 7.6** Sediment core from Lake Caviahue (depths: 70 m in SA and 90 m in NA) containing the acidophilic algae *K. rhaphidioides* (**a**, **b**). Photo of the alga under a transmitted light microscope (**c**) and under an epifluorescence microscope (**d**)

An observation matrix was used to describe the relationships between the variables measured between 1998 and 2018 in Lake Caviahue. It was based on 271 samples and eight variables: biological (*K. rhaphidioides* abundance) as well as physical (pH, conductivity, water temperature, transparency Secchi disk) and nutrients ($NH_4^+$, SRP, and $CO_2$). A principal component analysis (PCA) was performed with these values. This analysis (Fig. 7.7) indicated that the first two axes explained 44% of the total data variation. The first axis showed an environmental gradient of phytoplankton abundance, pH, SRP, and conductivity values. The second axis showed a gradient related to $CO_2$, temperature, and $NH_4^+$. Transparency was important for both gradients.

Lake Caviahue was analysed as a sentinel of atmospheric deposition of nitrogen and carbon hypothesising that the physiological state of *K. rhaphidiodes* could be the indicator parameter (Baffico et al. 2018). Chlorophyll per cell and the ratio between in vivo fluorescence and chlorophyll concentration were measured. The study involved the analysis of the development and physiological state of phytoplankton in different seasons in relation to nutrient concentration and dynamics in the lake. During the study, a very low N:P ratio <1 and $CO_2$ concentrations of 0.5 to 0.9 mg $CO_2$ $L^{-1}$ were registered. At pH values below 4, $CO_2$ concentration is at air equilibrium and virtually all DIC is present in the form of $CO_2$ (Diaz and Maberly 2009). N and $CO_2$ concentrations were indicative of nutrient limitation for phytoplankton growth. Laboratory experiments with nitrogen enrichment confirmed that an increase in nutrient content enabled a better physiological state of algae. Under the projected scenarios of climate change, the increase of available N through the increase in deposition, and the increase of dissolved inorganic carbon as a consequence of higher atmospheric $CO_2$ will compensate for the natural nutrient

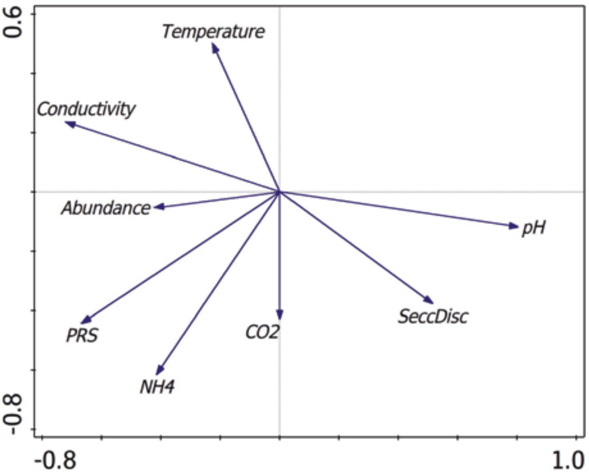

**Fig. 7.7** Two-dimensional plot of the PCA analysis showing the relationship between the physical and chemical variables and algal abundance recorded in Lake Caviahue

constraints observed in lake phytoplankton. However, these possible changes will also be regulated by the magmatic activity of the Copahue Volcano.

## 6 Water–Sediment Interactions

Lake Caviahue sediments, pore water, and water–sediment interface present characteristics of an environment influenced by volcanic activity. The pH in the water–sediment interface and in pore water near the surface (2.4 and 2.6, respectively) was similar to water column pH (2.9). Besides, the pH in surface sediments was on average 2.6 and increased with sediment depth to approximately 4.0. In addition, although Electric Potential (ORP) in the water column was +680 mV (highly influenced by the dissolved iron), in sediments it was always negative (between −20 mV and −250 mV) in both arms and likewise increased (negatively) with depth (Cabrera et al. 2016).

Lake Caviahue sediments, in both arms, have a homogeneous texture. The fine clay and silt fractions are predominant (5% and 43%, respectively). These sediments are characterised by having fine laminations (0.5–1 cm thick) (Fig. 7.8), which indicates the lack of bioturbation, due to the absence of a macrozoobentonic community. The geochemical composition was similar to that of andesitic rocks. $SO_3$ presence in these sediments, probably due to S availability in the volcanic hydro geothermal complex, makes them different from sediments in other Patagonian lakes.

Regarding nutrients, especially TP, the concentrations varied between 1163 and 1074 µgP/g dw, while TC varied between 2.4 and 13% in both arms (Temporetti et al. 2013). The low C/P ratio (<10) suggests that P could be efficiently retained in the sediment while organic carbon would be rapidly mineralised due to the high microbial activity. Although the mechanisms for controlling P in sediments are very well known in freshwater lakes, little is known for acidic waterbodies. Moreover, pH and ORP are factors that affect P release and have been the most studied (Wang et al. 2005; Wu et al. 2014). An increase in pH can cause the release of P bound to Fe and Al complexes due to site-specific competition between hydroxyl ions and P bound to these metal oxohydroxides (Kim et al. 2003; Li et al. 2016).

The link between TP and TC was assessed based on the contents in the different strata of core sediments. It was observed that both total concentrations increased with the age of the stratum, having a significant correlation ($p < 0.05$, $r^2$ of 0.95 for NA and 0.96 for SA) between both variables (Cabrera et al. 2016). The fractions of P, TP, and TC content in the cores indicated that organic matter was the main factor that controlled P retention in the sediments of this environment. This was also described by Cabrera et al. (2016), who found that P was highly correlated to the content of humic substances, that is, the recalcitrant OM fractions. In general, DOM present in the lake is mostly of autochthonous origin (Cabrera et al. 2020b).

One of the main reasons for the high TP concentrations found in the deeper layers of the sediment and TP association with organic matter is an increase in

**Fig. 7.8** Detailed view of sediment core from Lake Caviahue showing laminations unchanged (red arrow), which indicates the lack of bioturbation

suspended material (ash) in the lake, with a decrease in water transparency and in total phytoplankton biomass (e.g. April 2000 = 2.5 mg $L^{-1}$; February 2001 = 0.3 mg $L^{-1}$) caused by the two main eruptions of the Copahue volcano and the eruptive pulses in January 2013, March 2014, April 2015, and March 2018. These eruptive events and the consequent phytoplankton mortality implied the release of P to the sediment, in addition to other nutrients, upon sedimentation.

Cabrera et al. (2020a, b) found that the readily extractable fraction and the total sedimentary content of Fe, Pb, Cr, Cu, Mn, Cd, and Zn in sediments were very low compared to those in circumneutral lakes and below the toxicity thresholds established by the EPA and the Canadian Council of Ministers of the Environment (CCME) for the aquatic biota. In addition, the soluble chemical species were modelled, and non-toxic forms were predicted. Therefore, the sedimentary metal content poses no risk to the benthic biota. Metals in sediments are usually dragged to the bottom by co-precipitation with iron oxides and carbonates in oxic waters, and their formation is not favoured in Lake Caviahue due to its low pH, thus explaining the low trace metal content in its sediments.

One of the main environmental problems that affect water bodies is related to anthropic pollution. Several studies (Lee-Hyung et al. 2003; Smith et al. 2006; McDaniel et al. 2009) have considered sediments and the pore water contained in them as a more adequate parameter to determine the trophic state, supported by the concept that the trophic state can be substantially influenced by the release, and recirculation of P from the sediments to the water column (Carey and Rydin 2011). Temporetti et al. (2014, 2019) assessed (a) the concentration of soluble reactive phosphorus (SRP) in pore water, (b) the concentration of the P labile fraction in sediments, and (c) the depth distribution pattern of TP in sediments, in nine lakes and reservoirs of Patagonia, including Lake Caviahue. The authors found that Lake Caviahue could be classified as an oligotrophic, mesotrophic, or eutrophic environment, depending on the parameter considered (see Sect. 4). When analysing the sediment parameters mentioned above, the lake is classified as oligotrophic. The results indicated that to determine the trophic state of this environment both sediments and pore water could be considered (Temporetti et al. 2014, 2019) more suitable than using the typical parameters of the water column.

## 7 Acidity Gradient and Algal Epilithic Development in the Lower Agrio River (LA)

The LA presents more stable chemical characteristics at its source in Lake Caviahue (Baffico 2010) than those observed in the UA because the lake produces a buffer effect on the daily variation. However, its flow is controlled by the hydrological cycle: lower in dry seasons (summer-autumn), higher in wet seasons (Fig. 7.9). At LA km 24, pH (approx. 3, Fig. 7.10a), conductivity (Fig. 7.10b), and ion concentrations are similar to those recorded in Lake Caviahue (Pedrozo et al. 2001 and 2008) so the conditions for algae are still extreme although relatively stable throughout the year. This stability is what enables important epilithic developments, which is reflected in biomass values with little seasonal variation (seasonal averages between 20 and 30 mg Chl *a* $m^{-2}$, Fig. 7.10c). At LA km 24, the dominant algae in the epilithic community are *Ulothrix* sp. and/or *Klebsormidium* sp. (Baffico 2010). Both species have been reported in acidified environments in different parts of the world (Novis 2006; Aguilera et al. 2007).

From the confluence with River Trolope (pH = 7, low conductivity), the LA begins to be neutralised (Parker et al. 2008), so the pH value at LA km 29 (pH 3–5) depends on the flow ratio between the LA and its tributary, showing certain seasonality. As a consequence of the increase in pH, the precipitation of dissolved Fe occurs, which stains the riverbed a characteristic orange colour (Parker et al. 2008). Fe precipitates have a direct influence on algal developments due to the fact that

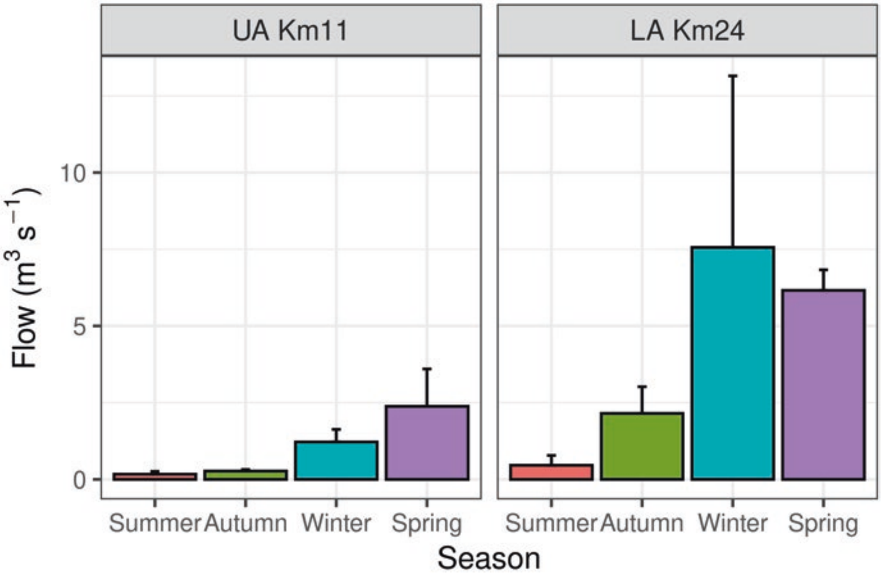

**Fig. 7.9** Seasonal average River Agrio flows at UA Km 11 and LA Km 24

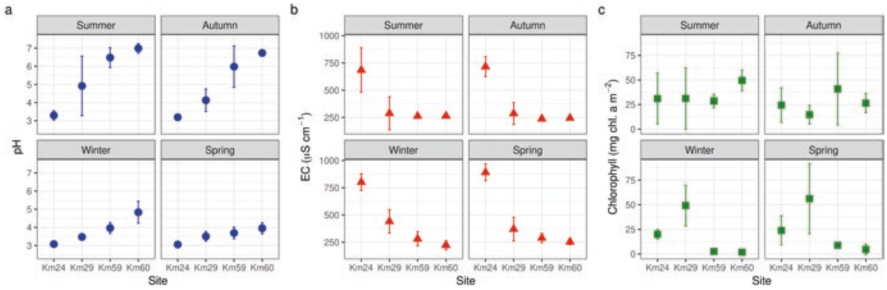

**Fig. 7.10** Seasonal average and standard deviation of (**a**) pH, (**b**) conductivity, and (**c**) epilithic biomass, at sites along the LRA

they cause physical and chemical impacts (Niyogi et al. 1999; Smucker and Vis 2011). The effect of the former is the shading they produce on algae, which, depending on the algal growth rate, may be compensated or will lead to a decrease in algal biomass. On the other hand, the chemical impact is manifested in P concentration since Fe precipitates capture part of the dissolved P (Temporetti et al. 2019) reducing its availability for algal growth. However, the precipitates are subject to photochemical reactions, so Fe is re-solubilised releasing P. This precipitation/dissolution cycle of Fe depends on light, pH, and temperature; therefore, it presents a daily (Parker et al. 2008) as well as seasonal variation. The epilithic community that develops in this area (LA km 29) shows important biomass developments with marked seasonal variation (seasonal averages between 15 and 56 mg Chl *a* m$^{-2}$,

Fig. 7.10c), and a greater diversity of species, although filamentous green algae continue to be dominant (Baffico 2010).

For the next 30 km downstream from Salto del Agrio (LA km 29), the LA riverbed has a deep orange colour, which indicates that Fe continues to precipitate. However, due to the lack of important tributaries (only temporary courses), the chemical characteristics of the LA remain with little variation (Parker et al. 2008; Baffico 2010) until the confluence with the next tributary, River Ñorquín. This tributary of alkaline waters has a lower flow than the LA and produces different effects depending on the time of year. In summer, when the LA flow is at its lowest values, the entry of River Ñorquín waters produces the practical neutralisation of the LA, registering pH values close to 7 or slightly higher (LA km 60, Fig. 7.10a). In winter, the LA flow is at its peak so the inflow of the affluent does not substantially modify pH values (approx. 4–5, Fig. 7.10a; Cabrera et al. 2020b). Aluminium precipitation occurs around pH 5.5, consequently, the Fe precipitates that appear in the LA channel are superimposed by those of Al (Parker et al. 2008; Temporetti et al. 2019). The latter precipitates are not subject to re-dissolution. Therefore, it has been noted that their effects on algae tend to be more damaging than those produced by Fe precipitates (Niyogi et al. 1999). The important seasonal pH variation at LA km 60, from neutrality to acidity, strongly conditions algae developments, registering large algae developments (50 mg Chl *a* $m^{-2}$, Fig. 7.10c) with high species diversity in summer (neutrality), while in winter (acidity), very low biomass values are registered (2 mg Chl *a* $m^{-2}$, Fig. 7.10c) and with low diversity. Filamentous green algae continue to be an important component of the epilithic community, accompanied by some species of diatoms (*Achnanthes* sp., *Epithemia* sp., *Navicula* spp., *Nitzschia* spp.) and filamentous cyanobacteria (*Lyngbya* sp.) during periods of neutrality (Baffico 2010). In this way, it is possible to observe a marked chemical gradient in River Agrio, from the extremely acidic condition to a neutral condition comparable to similar rivers in the area, which influences and enables different algal developments not observed in other water bodies in the Patagonian region.

## 8 Applied Aspects

### *8.1 Behaviour of a Naturally Acidic Lake as a Fenton Reactor*

There are many physical, chemical, and biological processes (with varying degrees of success, cost, and technology) which are used to treat wastewater to an appropriate standard before discharge. In general, these methods include screening to remove large particles, sedimentation to remove heavy gritty or inorganic matter, and most of the solid impurities from the liquid fraction. The resulting clearer liquid, which contains dissolved and colloidal matter, is then subjected to a biological stage during which most of the organic matter and impurities are oxidised to carbon dioxide and water or converted into biomass which is removed by sedimentation before discharge into water bodies (Rae 1998).

In Lake Caviahue, an important source of nutrients is the wastewater effluents from Caviahue town, which used to be discharged into the lake after passing through sediment traps or oxidation ponds. Recently, a new wastewater treatment plant has been built to serve the current permanent and tourist population (975 and 1100, respectively). The effluent from this facility is disinfected and has low concentrations of organic matter. The plant complies with the discharge standards established in the Argentinean Water Code, suitable for irrigation. Nevertheless, some nutrients (especially P) and OM reach the lake by surface runoff. In terms of P load (Pedrozo et al. 2008), this contribution (3.8 tons yr$^{-1}$) would be low compared to that of natural P entering by the UA and estimated at approximately 40 tons P yr$^{-1}$. Even when the plant is efficient in OM removal, some refractory organic compounds may still be discharged into the host water body. In this sense, considering the natural characteristics of Lake Caviahue, we propose a complementary method that would ensure higher degradation of pollutants before discharge.

Wastewater treatment by means of advanced oxidation processes (AOPs) is of major interest in modern environmental chemistry. These processes are based on the production of oxidising species such as hydroxyl radicals ($^{\bullet}$OH), which react with most organic compounds due to their high reactivity and low selectivity (von Sonntag 2008). Among AOPs, Fenton and Fenton-like processes involve a series of reactions during which $H_2O_2$ decomposition is triggered by the presence of a catalyst that leads to the production of hydroxyl radicals ($^{\bullet}$OH). In classical Fenton reactions, $Fe^{2+}$ is the catalytic agent while Fenton-like processes are catalysed by transition metals (frequently ferric ions, hereafter represented as $Fe^{3+}$). The advantages of Fenton and Fenton-like processes include the low cost of the reagent (Fe salts and $H_2O_2$), the absence of Fe toxicity, and the environmentally benign character of $H_2O_2$ (whose reaction by-products are $H_2O$ and $O_2$). The main limitation is the pH range since the catalyst activity drastically decreases at pH values below 2 and above 4 (Pignatello 1992). For this reason, the use of Fenton and Fenton-like systems as a contaminant removal method in natural environments is rather limited, since pH values of natural water bodies are usually neutral or slightly alkaline, quite far from the acidic conditions required for achieving efficient processes.

Natural conditions of Lake Caviahue, which include pH values within the optimal range for the catalyst activity and significant dissolved Fe concentrations (Cabrera et al. 2016), suggest that it could be a suitable environment for natural progress of Fenton and Fenton-like processes. It is noteworthy that, although some authors have studied the possibility of occurrence of these processes at near neutral pH in typical natural water environments (Qin et al. 2013; Vermilyea and Voelker 2009), to the best of our knowledge, natural Fenton and Fenton-like reactions in acidic lakes have not been assessed before.

Given the lack of information regarding the behaviour of the system as a possible contaminant degradation reactor, preliminary studies using a well-known model compound (i.e. phenol) were performed in order to depict the main features of the lake as a natural matrix in degradation processes. The objective of these experiences was to characterise the natural waterbody and the processes that take place in it under simple conditions. The results obtained in the laboratory suggest that Fenton processes are feasible under Lake Caviahue conditions and can lay the foundation for the development of a treatment of more complex mixtures (e.g. sewage effluents) in the natural conditions of the lake.

In this context, Nichela et al. (2019) have carried out a set of phenol degradation experiments: (1) to evaluate whether dark Fenton and/or Fenton-like processes are likely to occur in Lake Caviahue under natural conditions and (2) to assess the feasibility of exploiting the particular composition of this natural matrix for contaminant oxidation upon external addition of $H_2O_2$ in concentrations typically used for technological applications (considering only the contribution of non-photochemical processes).

To assess aim 1, phenol degradation experiments were performed using two $H_2O_2$ concentrations within the range reported for rainwater. Two different solutions were employed as reaction matrices: natural filtered water samples from Lake Caviahue and acidic solutions prepared in the laboratory (hereafter, artificial solutions) with the same Fe concentrations as those in situ. The results indicate the feasibility of natural Fenton processes in the lake. For each $H_2O_2$ concentration used, the phenol conversion degree was lower for the natural water matrix than for the artificial solution. These results can be explained by considering that some lake matrix components may have a detrimental effect on the availability of Fe species towards peroxide attack and that the presence of natural organic matter (NOM) may exert some scavenging effect on the reactive species produced.

To assess aim 2, oxidation experiments were performed using filtered lake water as the reaction matrix and $H_2O_2$ in excess with respect to phenol concentration. For comparison purposes, phenol degradation experiments were performed in an artificial solution with equivalent initial conditions: [phenol], [$H_2O_2$], [Fe] temperature, and pH. The results proved that, upon addition of $H_2O_2$ concentrations in the range used for technological applications, phenol concentration substantially decreases, conversion degrees being at least 70% within the first 30 min for both matrices. Although both profiles obtained showed autocatalytic behaviour, for the experiment carried out in Lake Caviahue water, the lag phase was much longer than that observed for the artificial matrix. The kinetics results suggested that the main difference between the artificial and the natural matrices was associated with $Fe^{3+}$ availability. Hence, the high complexing ability of NOM present in the lake matrix increased the stability of ferric oxidation state, thus suppressing $Fe^{3+}$ reduction and leading to a much longer lag phase for the natural system. It is noteworthy that, despite the NOM retardation effect, oxidation is not blocked. Moreover, although timescales were different, comparable conversion degrees of the model compound were reached in both matrices.

The knowledge of these basic features of the system lays the foundation for the development of tests that will enable the study of the behaviour of more complex pollutant mixtures, such as urban effluents discharged into the water body. This study will be essential in the development of an effluent treatment plan with the lake playing an active role as a natural decontamination system.

## *8.2 Use of Acidophilic Algae as Indicators*

Polycyclic aromatic hydrocarbons (PAHs) are pollutants widely distributed in the environment and accumulate in the soil (Alegbeleye et al. 2017; Hussain et al. 2018). PAHs are hydrophobic compounds, slightly soluble in water, with a high bioconcentration factor, toxic, mutagenic, and carcinogenic (González-Paredes et al. 2013). The use of peroxydisulphate anion (PS) as a strategy for the remediation of PAH-contaminated soils has been very attractive despite its negative effects on microbial activity due to oxidative stress (Tsitonaki et al. 2008) and decrease in soil pH caused by the same procedure. Treatment with PS produces sulphate ions and $H^+$ as final products (Maurino et al. 1997) leading to a decrease in pH as PS concentration increases (Tsitonaki et al. 2008). pH decrease has a direct effect on microbial growth if the system buffering capacity is not sufficient and persists even after the end of the treatment (Lemaire et al. 2013).

Native acidophilic or acid tolerant microalgae (*K. rhaphidiodes*, *Cyanidium caldarium*, and *Euglena mutabilis*) isolated from River Agrio and Lake Caviahue were chosen to study their usefulness as bioindicators of PAH-contaminated soils (Diaz et al. 2015). They were tested in specific monocultures by using soils with and without artificial contamination with phenanthrene. The toxic capacity of PAH-contaminated soils has different degrees according to the remediation stage and is normally detected (Flotron et al. 2005) through toxicity tests and/or analytical procedures (HPLC, GC/MS, etc.). Two of the three acidophilic species assayed resulted in potential bioindicators (Fig. 7.11). *Cyanidium caldarium* and *Euglena mutabilis* were responsive to PAH contamination in the soils tested. On the other hand, *K. rhaphidioides* had a growth response dependent on soil type and would not be a good bioindicator of PAH contamination.

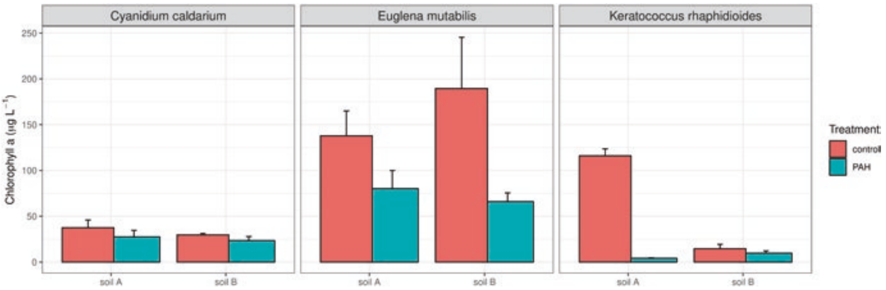

**Fig. 7.11** Average biomass, indicated by the concentration of chlorophyll ***a,*** of the three acidophilic algae growing in soils contaminated with PAH. (Modified from Diaz et al. 2015)

## 9 Conclusion

Lake Caviahue is regulated by the Upper Agrio River, its main tributary, which controls the chemical composition of this environment (pH, conductivity, and nutrient concentration) in relation to the magmatic activity of the Copahue Volcano. A variation in pH was observed, both in the lake and in the river, over the 20-year study period. As a consequence of this variation, strong changes were observed in the composition and substrates for the colonisation of microalgae both in the river and on the lake shore. This natural change has masked the eventual impacts of the contribution of anthropogenic nutrients and organic matter to the lake. However, if the pH continues to increase, the effects of these inputs could become more evident.

Lake Caviahue sediments showed that surface and water-column pH values were similar for both arms but increased with depth and under anoxic conditions. These conditions strongly influenced metal partitioning in lake sediments, particularly of Fe. On the other hand, the sedimentary trace metal content in Lake Caviahue was low, below the toxicity thresholds established by the EPA and CCME, and in non-toxic chemical forms, posing no risk to the aquatic biota. Regarding P in lake sediments, P sequestration did not occur at least while the pH range was acidic (below 3.0). The pH increase observed since 2012 has caused P co-precipitation with iron compounds (schwertmannite, Fe oxohidroxides, or ferricrite deposits) similar to those previously described by Parker et al. (2008), Gammons et al. (2005), and Llano et al. (2020). These red precipitates can be seen along the lake and LA shores, until the point wherein pH increases to approximately 4.5.

The factor that controlled P retention in sediments, in both arms, was the availability of OM, Fe, and Al, which were found in very high concentrations due to low pH in sediments. The low values of C:P and C:N ratios in Lake Caviahue seston indicate that P is not limiting for phytoplankton. Some sediment (TP concentration) and pore water (SRP concentration) parameters are better descriptors of the trophic state of this environment than the parameters of the water column usually used.

The very low diversity and the simple trophic web of Lake Caviahue plankton community are characteristics that make it unique among the rest of the oligotrophic lakes of Patagonia. There is no seasonal succession of phytoplankton species and the almost exclusive presence and growth of *K. rhaphidioides* is linked to dissolved organic carbon as well as to low nitrogen and inorganic carbon concentrations. Low algal diversity is recorded along the Upper and Lower Agrio River, and as in the lake, it is a consequence of the low pH and the high concentration of dissolved metals that reduce the number of species able to cope with the extreme conditions.

The River Agrio-Lake Caviahue system has distinctive characteristics that make it unique in South America. It is of the utmost importance to preserve this particular ecosystem in its natural state enabling us to learn more about it and do research work on processes (e.g. the natural Fenton reactor) that are impossible to observe in other water bodies.

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# Chapter 8
# Mercury in Aquatic Systems of North Patagonia (Argentina): Sources, Processes, and Trophic Transfer

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## 1 Overview of Mercury in the Environment

Mercury (Hg) is a global pollutant of serious concern due to its toxicity, persistence, and mobility in the environment. Currently, regions of the world that are remote and devoid of natural and/or anthropogenic sources of Hg show the impact of this toxic metal due to its long-range atmospheric transport and deposition (Driscoll et al. 2013; Obrist et al. 2018). As a consequence of the global presence of Hg, different Hg compounds have become ubiquitous in the environment and in food (Fernándes et al. 2020). The organic species monomethyl- and dimethylmercury ($CH_3Hg$ and $(CH_3)_2Hg$, respectively) are potent neurotoxins that affect primarily the central nervous system. Elemental Hg ($Hg^0$) and inorganic mercury species (i.e., Hg salts) are also dangerous for human health, and their absorption through inhalation, contact, and ingestion can damage the gastrointestinal tract, the lungs and kidneys, the skin,

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as well as the nervous and immune systems (Lehnherr 2014; WHO 2017). Due to its lipophilic nature, $CH_3Hg$ bioaccumulates and biomagnifies in aquatic food webs more readily than other Hg species. Methylmercury causes long-term consequences at the ecosystem level and affects wildlife and human populations with fish-based diets (Driscoll et al. 2013; Eagles-Smith et al. 2016; Evers 2018; Whitney and Cristol 2018; Chételat et al. 2020). Several factors are determinant of the effects of Hg on wildlife and humans, including the type of Hg, the dose, the route and duration of exposure, and the developmental stage. The presence of $CH_3Hg$ in wildlife is the result of the combination of ecological processes influencing dietary exposure and physiological processes regulating assimilation, transformation, and elimination (Ackermann et al. 2016; Chételat et al. 2020).

Mercury has a complex biogeochemical cycle involving its circulation among the lithosphere, the atmosphere, and the hydrosphere (Driscoll et al. 2013; Obrist et al. 2018). This element is naturally mobilized from lithospheric reservoirs to the atmosphere through volcanic and geological activity, rock weathering, and also volatilizes from surface waters, soils, and vegetation (Selin 2009; Driscoll et al. 2013). Humans have mobilized Hg from reservoirs to the atmosphere through mining, coal combustion, and industrial processes, altering its biogeochemical cycling by increasing drastically the atmospheric concentrations (~450%) and further deposition in ecosystems (Pirrone et al. 2010; Driscoll et al. 2013; UNEP 2018). Recent estimates indicate that anthropogenic Hg emissions (~2000 Mg $yr^{-1}$) exceed by far natural emissions (76–300 Mg $yr^{-1}$) (Streets et al. 2017, 2019). In addition to emissions, anthropogenic sources of Hg include direct releases to the aquatic environment, particularly from point sources, and the remobilization of historical deposition of Hg, and from legacy Hg deposits in contaminated sites (AMAP/UNEP 2013; Kocman et al. 2013). Archives of Hg deposition such as peat and lake sediments from remote regions indicate up to fivefold enrichment of Hg due to atmospheric deposition from the beginning of the industrial era (Streets et al. 2017; Gustin et al. 2020).

The behaviors of the different chemical forms of Hg play critical roles in its biogeochemical cycling. Gaseous elemental Hg ($Hg^0$), the predominantly emitted species and the one with the highest concentration in the atmosphere, has high chemical inertness facilitating its long-range transport before deposition (Sprovieri et al. 2010; Driscoll et al. 2013; Obrist et al. 2018; Lyman et al. 2020). Reactive gaseous Hg (mostly gaseous chloride and oxide forms of ionic Hg) and particle-bound Hg are more soluble in water and reactive than $Hg^0$, having a shorter residence time in the atmosphere (0.5 to 2 days and 0.5 to 3 days, respectively), depositing in local and regional ecosystems (Driscoll et al. 2007, 2013; Sprovieri et al. 2016; Obrist et al. 2018). Although there is uncertainty regarding the atmospheric redox chemistry of Hg, the halogen atoms are potentially important $Hg^0$ oxidants in the atmosphere with binding energies of the Hg compounds produced in the order $HgCl > HgBr > HgI$ (Selin 2009; Driscoll et al. 2013). Therefore, depending on the speciation of the emissions and on the residence time in the atmosphere, Hg deposition may impact locally, regionally, or globally (Dastoor and Larocque 2004; Driscoll et al. 2007, 2013). Wet deposition consists mostly of reactive $Hg^{2+}$, gaseous oxidized Hg (GOM), and particulate-bound Hg (PBM), scavenged by water droplets in the air, whereas $Hg^0$ is a major component of the total Hg in dry deposition along with $Hg^{2+}$ in gaseous or particulate phase. Dry deposition of $Hg^0$ via plant uptake (stomatal gas exchange) is dominant in terrestrial vegetated ecosystems (Gustin et al. 2020). Thus, in forested ecosystems, litterfall is considered a major component of atmospheric deposition and its decomposition results in large inputs of Hg and organic matter (OM) to the soil (Grigal 2002). Also, gaseous and particulate $Hg^{2+}$ depositing directly to the canopy are washed off by throughfall contributing to soil deposits (Graydon et al. 2008; Bishop et al. 2020). Conversion of deposited $Hg^{2+}$ to the organic species monomethyl ($CH_3Hg$) and, to a limited extent, to dimethylmercury ($(CH_3)_2Hg$) occurs through biotic and abiotic processes in anoxic and/or oxygen-deficient sites of soils, wetlands, and aquatic systems. Sulfur, iron, and OM affect $Hg^{2+}$ chemical speciation (Ravichandran 2004; Bravo and Cosio 2019).

Lakes receive atmospheric Hg inputs (mostly $Hg^{2+}$) directly through the surface, and, due to their lower position in the landscape, concentrate the Hg deposited in their catchments which is mobilized by runoff (Fig. 8.1). In particular, remote lakes at high elevation are currently subject to remarkable biochemical changes due to deposition of different elements, including C, N, S, and Hg, among others (Driscoll et al. 2007, 2013; Mladenov et al. 2011, 2012). The relative importance of atmospheric and watershed Hg sources varies depending on land use, hydrology, and the content and composition of dissolved organic matter (DOM) (Hsu-Kim et al. 2018; Obrist et al. 2018; Braunfireun et al. 2020). The C pool of the catchment plays a crucial role in the transport and cycling of Hg from terrestrial and aquatic systems.

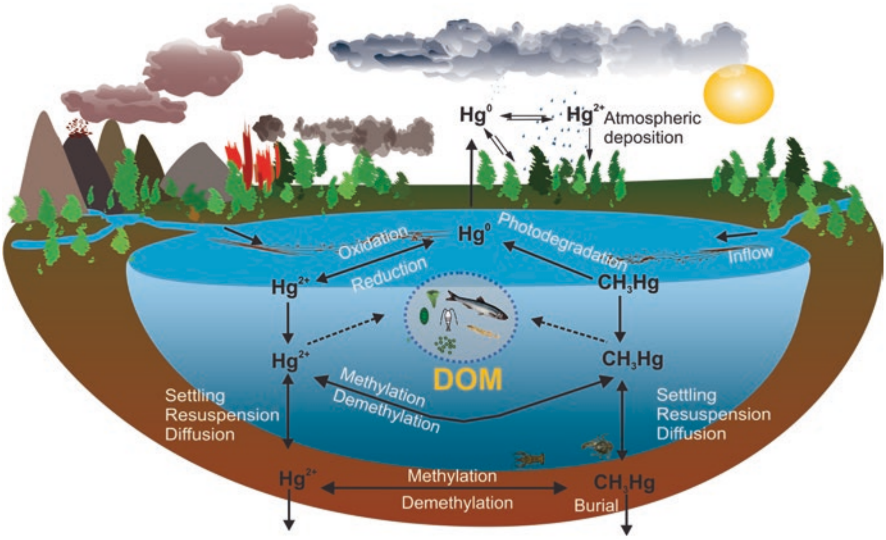

**Fig. 8.1** Mercury cycling in catchments. References: $Hg^0$, elemental mercury; $Hg^{2+}$, ionic mercury; $CH_3Hg$, methylmercury; DOM, dissolved organic matter

Natural OM contains different amounts of thiols, which sulfhydryl group has a high capacity to complex $Hg^{2+}$ and $CH_3Hg$. The concentration and molecular composition of the DOM pool determine Hg fractionation, availability, and biotic uptake, as well as the net $Hg^{2+}$ methylation in aquatic systems (Ravichandran 2004; Bravo and Cosio 2019; Lavoie et al. 2019; Branfireun et al. 2020). In these ecosystems, $Hg^{2+}$ can be either (i) reduced to $Hg^0$ and reemitted to the atmosphere, (ii) methylated to the organic form $CH_3Hg$, or (iii) bound to OM and inorganic particles, depositing on bottom sediments. $CH_3Hg$ formed in aquatic ecosystems can also deposit on the sediments, be methylated, and/or form volatile $(CH_3)_2Hg$, although the latter pathway is still a matter of debate (Fig. 8.1) (Paranjape and Hall 2017; Zhu et al. 2018; Braunfireun et al. 2020). Part of $(CH_3)_2Hg$ can be reemitted to the atmosphere or degraded to $CH_3Hg$, which can also be biotically and/or abiotically demethylated (Marvin DiPasquale et al. 2000; Barkay et al. 2003; Schaefer et al. 2004; Fernández-Gómez et al. 2013). Most of the $CH_3Hg$ present in headwaters is formed in situ or in the surrounding catchment and is subsequently transported into rivers, lakes, and oceans, concentrating in aquatic food chains (Chételat et al. 2020 and references therein).

## 2 Freshwaters in the Patagonian Landscape

Patagonia is a vast territory characterized by wide environmental gradients that reflect in the different landscapes from the Pacific to the Atlantic coast, from northern to southern locations, and from lowlands to high altitudes in the Andes. The headwaters of the region comprise complex fluvial networks of glacial origin born in the Andes, including mountain streams and lakes, deep and shallow piedmont lakes draining through large rivers toward the Atlantic and Pacific oceans (Chap. 9). In Andean North Patagonia (Argentina), aquatic systems occur along a bioclimatic gradient characterized by a west-to-east sharp decrease in the precipitation caused by the rain shadow effect of the Andes on the westerlies, which generates a transition from temperate forests near the Andes to a semidesert in less than 100 km to the east (Fig. 8.2a,b).

The Andean sector of Patagonia is located within the Southern Volcanic Zone, under the influence of active volcanoes (Andean volcanic belt) with several eruptions during the Holocene (Singer et al. 2008; Stern 2008). The Andean volcanic belt extends from 33° S to 46° S, and the emissions of gaseous elements and

**Fig. 8.2** (continued) DH, Dina Huapi; PC, Puerto Cisne; BRC, San Carlos de Bariloche City; PCCVC, Puyehue Cordón Caulle Volcanic Complex. Red triangles indicate volcanoes. Numbers indicate sites in NHNP in which point measurements of atmospheric mercury have been performed (Horvat and Kotnik 2007). Sites 1, 2, 5, and 7 to 10: ~1 to ~3 ng $m^{-3}$; Point 3: ~15 ng $m^{-3}$; Point 4: ~15 and ~18 ng $m^{-3}$; Point 6: ~10 ng $m^{-3}$

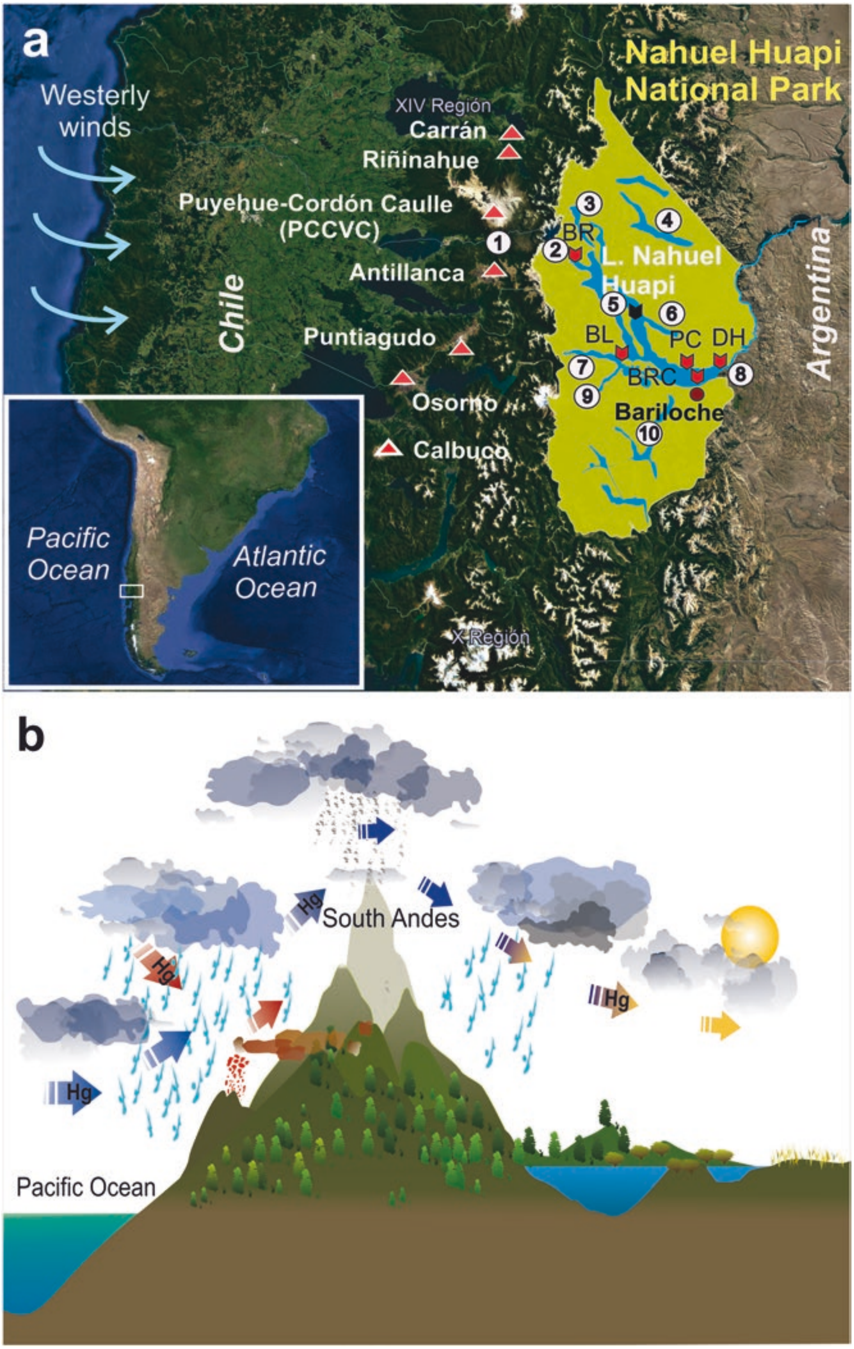

**Fig. 8.2** (**a**) Map of Nahuel Huapi National Park, Patagonia, Argentina (green area) (Aerial Image from Google Earth). (**b**) Longitudinal section of the bioclimatic gradient within Nahuel Huapi National Park (North Patagonia, Argentina). References: BR, Brazo Rincón; BL, Bahía López;

pyroclastic material from active volcanoes impact the surrounding landscape, particularly at the eastern side of the Andes due to the action of the westerlies that promote transport and deposition. Consequently, in Andean Patagonian catchments, soils and lake sediment profiles show the historical accumulation of volcanic ash resulting from the intense volcanic activity of the Patagonian Andes (Ribeiro et al. 2005, 2010; Daga et al. 2008; Pereyra and Bouza 2019). Volcanic eruptions and forest fires are frequent disturbances in Andean Patagonia, shaping the structure and function of terrestrial and aquatic ecosystems and ultimately reflecting in biogeochemical cycling (Veblen and Kitzberger 2002; Diaz et al. 2013; Modenutti et al. 2013, 2016; Berenstecher et al. 2017; Holz et al. 2017; Beigt et al. 2019; Du Preez et al. 2020; see also Chaps. 3 and 7).

In North Patagonia, headwater catchments are included in protected areas since they are unique biomes and reservoirs of biodiversity. The Nahuel Huapi National Park (NHNP, 40° 08'–41° 35’ S; 71° 01’–71 57’ W; 7,173 km$^2$ surface) is the largest natural reserve of North Patagonia and comprises the Nahuel Huapi lake catchment, which includes the headwaters of the major fluvial network of North Patagonia (Fig. 8.2a). The topography of NHNP is characterized by Andean mountains and valleys at the western stretch, grading in altitude toward formations of sierra and meseta at the east (Fig. 8.2b). Glacial and volcanic processes shape the landscape. The climate is cold temperate and precipitation ranges from ~3500 mm yr$^{-1}$ at the west to ~500 mm yr$^{-1}$ at the east of the park. The precipitation pattern is highly seasonal with ~60% of the annual precipitation, concentrated between May and September (austral autumn and winter). Along this environmental gradient, three bioclimatic units can be distinguished from west to east: the High Andean district above the tree line, forested areas including hyperhumid, humid, and subhumid forests of *Nothofagus* spp., and the steppe (Fig. 8.2b) (Ferreyra et al. 1998; Mermoz et al. 2009; Queimaliños et al. 2019).

Topography, vegetation, climate characteristics (i.e., seasonality and precipitation), and lake morphometry are chief drivers of airshed-watershed interactions, determining the circulation patterns of materials within the landscape. Linkages between terrestrial and aquatic environmental matrices depend on ecosystems properties and climate, and, thereby, they are influenced by regional and global trends. In NHNP, the marked bioclimatic gradient reflects in the biogeochemical patterns of lakes. In fact, climate-vegetation co-effects have been shown to drive the timing, quantity, and quality of the terrestrial inputs to deep and shallow lakes and running waters, as revealed through changes in their DOM pools (Queimaliños et al. 2012, 2019; García et al. 2015a, b; Soto Cárdenas et al. 2017; Zagarese et al. 2017). DOM is a main transporter of terrestrial C, nutrients, and Hg toward aquatic systems. In NHNP, lakes at the western end of the gradient display comparatively higher terrestrial signatures (particularly during the wet season), due to the enhanced hydrological connectivity and the contribution of native forests. Despite differences in the amount and quality of terrestrial inputs, deep Andean lakes show similar properties in nutrient and dissolved organic carbon (DOC) concentrations corresponding to oligo- and ultraoligotrophic conditions, whereas shallow lakes are a more heterogeneous group, encompassing ultraoligo- to mesotrophic systems (Díaz et al. 2007;

Queimaliños et al. 2012, 2019; Soto Cárdenas et al. 2017, 2018a). Deep lakes are warm monomictic, showing thermal stratification in late spring and summer, and developing the thermocline between 30 and 40 m (Queimaliños et al. 1999; Pérez et al. 2002). In contrast, shallow lakes are usually polymictic, freezing occasionally in harsh winters and showing thermal stratification in late spring or early summer (Soto Cárdenas et al. 2017).

The waters have low specific conductivity (<50 μS $cm^{-1}$) and circumneutral pH. Low DOC (~0.25–4 mg $L^{-1}$) and chlorophyll *a* (~0.1–1.5 μg $L^{-1}$) concentrations determine optically thin water columns, resulting in high underwater irradiance and extended euphotic zones (ca. 40 m) in deep lakes, and in water columns illuminated up to the bottom in the case of shallow lakes (Morris et al. 1995; Queimaliños et al. 2012, 2019; Soto Cárdenas et al. 2017). Chlorophyll *a* concentration shows remarkable variation in the vertical profile, especially in deep lakes, which characterize by the occurrence of metalimnetic peaks in summer due to the patchy vertical distribution of algae-bearing mixotrophic ciliates and flagellates (Queimaliños et al. 1999, 2002, 2012, 2019; see also Chap. 3).

## 3 Mercury in Andean Patagonian Catchments

Understanding the fate of Hg in aquatic systems requires a comprehensive approach due to the nature and ubiquity of this pollutant and to the fact that aquatic systems concentrate materials from their catchments, reflecting atmospheric as well as terrestrial processes. For two decades and in the light of the pieces of evidence gathered, the focus of Hg research in Andean Patagonian environments moved progressively toward a catchment perspective. Ecosystems of the region still have many areas to be explored and processes to be unveiled. Studies of Patagonian ecosystems usually face the need to step back from a goal to resolve fragmentary environmental information essential to move forward. The following sections aim at providing the reader with critical elements to understand the occurrence and cycling pathways of Hg in a large and pristine headwater catchment of North Patagonia, involving studies on different environmental compartments of NHNP (Fig. 8.2a).

### *3.1 First Records of Mercury in Andean Patagonian Catchments: Evidence from the Analysis of Lake Sediment Sequences (Legacy Mercury/Historical Mercury)*

Most Hg research in the Argentinean Patagonia has been carried out in pristine headwaters including high-altitude lakes (mountain lakes) and deep and shallow piedmont lakes of Northwestern Patagonia. The most extensive body of work has been gathered in NHNP. Early research focusing on the occurrence of metals in

Lake Nahuel Huapi (764 m a.s.l; 557 $km^2$ surface; 464 m maximum depth) detected the presence of Hg above background levels in sedimentary sequences and in the suspended load in the water column at different lake sites (Ribeiro Guevara et al. 2002). Later, a screening of shallow and deep lakes belonging to the Nahuel Huapi catchment (Nahuel Huapi, Moreno West, Morenito, Espejo Chico, and Escondido) and Lake Traful evaluated Hg concentrations in sediment profiles, revealing two background Hg levels. Lower total Hg (THg) levels, ranging from ~0.08 to 0.21 $\mu g\ g^{-1}$, were associated with preindustrial times, while higher levels, ~0.17 to 0.32 $\mu g\ g^{-1}$, were associated with modern times (Ribeiro Guevara et al. 2005). In the upper layers of sediment cores from most of the studied lakes, dated to the second half of the twentieth century, the THg concentrations were even higher (from 0.5 to 3 $\mu g\ g^{-1}$) indicating moderate Hg contamination. In deeper layers, Hg concentrations three- to fivefold above background levels were observed, suggesting natural Hg inputs during the past millennium. Other studies based on sediment cores from the mountain lake Tonček and the piedmont lake Moreno West also detected high preindustrial Hg levels. In the Hg concentration profiles of both lakes, two sections were identified in the core with values up to tenfold the background level (0.05 $\mu g\ g^{-1}$), corresponding to the thirteenth century, and to the eighteenth and nineteenth centuries, with values compatible with contamination (0.40 to 0.65 $\mu g\ g^{-1}$), suggesting the impact of regional events (Ribeiro Guevara et al. 2010). Increased Hg concentrations were detected immediately above some tephra layers, pointing to a link with volcanic events. Besides, deep Hg peaks were found coinciding with charcoal peaks, both matching with evidence arising from tree-ring data and historical records of extended forest fires (Ribeiro Guevara et al. 2010). Thus, in the studied catchment (Fig. 8.2a), lake sediment archives reflect the frequent disturbances caused by volcanic eruptions and forest fires and the departure of Hg concentrations from background levels attributable to the global cycling of Hg in modern times.

At a wider regional scale, the analysis of sediment sequences from other lakes in southern Patagonia supported the connection between increased THg levels in the sediments, volcanic events, and widespread fires (Daga et al. 2016), as well as with environmental and climate changes (Hermanns and Biester 2011, 2013a,b; Hermanns et al. 2013; Biester et al. 2018).

## 3.2 *Mercury in the Nahuel Huapi Catchment*

### Atmospheric Mercury: Assessments Through Bioindication and Ambient Concentration Monitoring

Lichens have been extensively used as bioindicators of atmospheric pollution since they can store nonessential elements from the surrounding environment with little to negligible effects on their biological functions (Garty 2001; Bargagli et al. 2016). Atmospheric Hg bioindication studies in NHNP using native lichen species have reported THg levels in thalli ranging from 0.06 to 1.38 μg $g^{-1}$ dry weight (DW), reflecting both natural and human-related Hg sources (Ribeiro Guevara et al. 1995, 2004; Bubach et al. 2012, 2014). Translocation experiments initiated with thalli of native lichen species from pristine areas of the park and transplanted to urban settings in San Carlos de Bariloche city showed a substantial increase in their natural (background) THg levels (from ≤0.107 μg $g^{-1}$ DW to ≤0.280 μg $g^{-1}$ DW), reflecting the influence of Hg emissions due to local human activity (Bubach et al. 2001).

The first instrumental assessment of atmospheric mercury levels in North Patagonia was performed in 2007, during an international survey that included measurements of gaseous elemental mercury (GEM) in different sites of South America. During this campaign, GEM concentrations were measured on different transects within NHNP, using a portable cold vapor atomic absorbance system (Lumex RA-915M). GEM concentrations ranged from 1 to 18 ng $m^{-3}$, displaying a large spatial variability with high levels (~15 to ~18 ng $m^{-3}$) close to the Traful area and to the volcanic complex Puyehue-Cordón Caulle (PCCVC), and also in the adjacencies of the Huemul branch of Nahuel Huapi lake (~10 ng $m^{-3}$). Other sectors of the park showed much lower GEM levels, ranging between ~1 and ~3 ng $m^{-3}$ (Fig. 8.2a) (Horvat and Kotnik 2007; Higueras et al. 2014).

The continuous monitoring of atmospheric mercury in NHNP started in 2011, when the Global Mercury Observation System (GMOS: http://www.gmos.eu/) established the EMMA Station (41° 07’ S, 71° 25’ W; 800 m a.s.l) in the suburbs of San Carlos de Bariloche City. Currently, this station integrates the GOS4M global network (http://www.gos4m.org), a flagship of the Group of Earth Observation (GEO). Gaseous elemental Hg ($Hg^0$, GEM), oxidized mercury compounds (GOM comprising mostly Hg halides and HgO), and particle-bound mercury (PBM)] are measured using an automated Hg cold vapor atomic fluorescence spectrometer coupled to speciation modules (Tekran Instrument Corp., Canada). High-resolution data of atmospheric Hg obtained from 2012 to 2019 indicated low mean GEM concentrations (0.86 ± 0.16 ng $m^{-3}$) with a seasonal pattern characterized by the highest level in spring (0.95 ± 0.13 ng $m^{-3}$) and the lowest in autumn (0.80 ± 0.15 ng $m^{-3}$). GOM concentration averaged 4.61 ± 4.00 pg $m^{-3}$, fluctuating seasonally with the highest levels in autumn and the lowest in winter. PBM averaged 3.74 ± 3.41 pg $m^{-3}$, with the highest mean level recorded in autumn and the lowest in spring. THg and also its fractions (GEM, GOM, and PBM) displayed overall higher concentrations during daytime hours. Total Hg showed a minimum concentration in the early morning, high values from midday toward the afternoon, and overall lower levels during nighttime. The dynamics of the different atmospheric Hg species have been found to be influenced by the direction and speed of the winds as well as by the temperature and humidity (Diéguez et al. 2019). In order to study the influence of atmospheric transport from local and regional sources in the sector of NHNP covered by the EMMA station, the hybrid single-particle Lagrangian Integrated Trajectory model (HYSPLIT) was applied to calculate air mass backward trajectories (BWT). This analysis indicated that atmospheric Hg concentrations in NHNP are simultaneously affected by local and regional sources (forest fires and volcanoes). Noteworthy, lower concentrations were recorded with the inflow of clean oceanic air masses corresponding to long-range transport. A Potential Source Contribution Function analysis (PSCF) showed that emissions in the marine boundary layer from remote areas in the Pacific Ocean are also sources of GEM and GOM. Thus, the results obtained through the monitoring of atmospheric Hg inside NHNP confirmed the influence of volcanic sources and forest fires previously reported in studies based on lake sediment sequences. Low levels of atmospheric Hg, such as those recorded coinciding with the influx of clean air masses from the Pacific Ocean and corresponding to long-range transport, indicate the influence of circulating Hg in the global atmosphere (Diéguez et al. 2019).

## Mercury in Vegetation and Soils

The distribution of Hg in the landscape has been found to correlate to latitude, annual precipitation, soil organic matter, leaf area, and vegetation greenness. Studies at landscape scale have shown that forested watersheds and their drainage network (including riverine and lacustrine sediments) show comparatively higher concentrations of $Hg^{2+}$ compared to other ecosystems. Forest vegetation and soils store huge quantities of Hg and, thus, largely determine the Hg levels at the watershed scale (Fleck et al. 2016; Obrist et al. 2018). In particular, soil compartments have been extensively studied to assess historical atmospheric deposition because soil is the most important terrestrial repository of contaminants. In mountain regions, which are prone to atmospheric deposition, soil Hg concentrations are the result of the natural background due to local mineral composition and from natural and anthropogenic atmospheric inputs (Zhang et al. 2013). Vegetation patterns largely determine Hg levels in soils due to the importance of the plant-funneled $Hg^0$ deposition process, being atmospheric deposition to forests up to four times higher than wet deposition in open sites (Grigal 2002). Consequently, forest soils show much higher Hg levels than shrublands and deserts in which the smaller contribution of the vegetation and higher reemission of $Hg^0$ lead to lower Hg accumulation. Nevertheless, accumulation and retention of Hg in soils are determined by several intrinsic variables such as morphology and genesis, texture, pH, organic matter (OM) content, and stability (reviewed in Obrist et al. 2018).

Soils of Andean Patagonia display original features due to the accumulation of volcanic ash resulting from the intense volcanic activity of the region and the presence of temperate-cold humid forests (Pereyra and Bouza 2019). Andisols have scarce differentiation of their horizons and a high content of allophanes (amorphous clays derived from the weathering of volcanic glass), showing high capacity for water retention, OM stabilization, P retention, and pH buffering (Mazzarino et al. 1998; Pereyra and Bouza 2019). Little is known about the occurrence and dynamics of Hg in Patagonian soils. In this sense, a first evaluation of Hg levels in connected environmental matrices (including soils) within NHNP focused in the subcatchment of Brazo Rincón (BR, 40° 44’ S, 71° 46’ W), a western branch of Nahuel Huapi lake situated at ~30 km southeast of the active PCCVC (Fig. 8.2a). This lake branch (surface: ~11 $km^2$, maximum depth: 100 m) drains a catchment of ~227 $km^2$ and displays a high watershed to lake area ratio (20/6). The landscape is characterized by steep slopes covered (>70%) by a dense forest of deciduous and evergreen *Nothofagus* spp. Previous studies showed the presence of high THg levels in environmental compartments and aquatic biota of this subcatchment, reaching concentrations found in contaminated sites worldwide, and contrasting with much lower levels recorded in other locations of the park (Rizzo et al. 2011, 2014). These conditions indicate that the subcatchment of BR can be considered as a hotspot of Hg within NHNP, encouraging further studies focused on soils, vegetation, and water bodies draining to BR. Total Hg concentrations measured in rainfall samples collected in the BR catchment varied between 21 and 27 ng $L^{-1}$, suggesting a minor contribution of wet atmospheric deposition. Soil profiles under the canopies of *Nothofagus* spp. showed scarce development below the upper volcanic ash layer deposited during the 2011–2012 PCCVC eruption and characterized by the occurrence of O, A, and 2A horizons. The C horizon which separates the A and 2A horizons corresponded to the weathered parent material with tephra characteristics that correlated with the PCCVC eruption of 1960 and the Calbuco volcano eruption of 1961. OM content in the O horizons ranged from 14.2 to 25.3 %, and THg concentrations varied from 19 to 106 ng $g^{-1}$, with THg:OM ratios ranging from 1.3 to 4.6 (Rizzo et al. submitted). A direct relationship between THg and OM concentrations in the O horizon and with the altitude suggests that altitudinal vegetation patterns could drive THg levels in soils. The native forest is characterized by the dominance of the evergreen *Nothofagus dombeyi* at the piedmont (from ~670 to ~900 m.s.s.l.) and the prevalence of the deciduous *N. pumilio* above 900 m a.s.l. These tree species have distinct litterfall and throughfall contributions of Hg to soils due to species-specific Hg storage and timing of leaf fall. In fact, the foliage of the evergreen species *N. dombeyi* has been found to contain higher THg concentrations (19 to 190 ng $g^{-1}$) compared to the deciduous *N. antarctica* (17 to 45 ng $g^{-1}$) (Juárez et al. 2016). Evergreen foliage incorporates Hg yearlong, while deciduous foliage incorporates it during the growing season, releasing differential amounts of Hg to soils through litterfall (leaf turnover and/or leaf fall) and throughfall, showing also seasonal differences in the contribution (Bushey et al. 2008).

Future studies of Hg dynamics in the area should evaluate key processes such as the uptake of atmospheric Hg by the forest vegetation, its storage, and the factors involved in its transference to soils and aquatic ecosystems, pathways underrepresented in the research carried out in the region up to the moment.

## Mercury in Freshwaters

Mercury is naturally present in waters at very low levels. As it has a high tendency to adsorb on surfaces, a large proportion in the water phase is attached to suspended particles, which play an important role in the transport of Hg in aquatic systems. Inorganic Hg tends to bind strongly to mineral particles and detrital OM, whereas $CH_3Hg$ tends to be more strongly associated with biogenic particles, including organisms such as bacteria, algae, and phytoplankton. Settling of this particulate matter is considered a major delivery mechanism to the sediments which are thus considered as a sink for Hg (Ullrich et al. 2001; Ravichandran 2004; Gallorini and Loizeau 2021). An important part of the biogeochemical cycle of Hg in aquatic systems involves its reemission from the sediments to the water column, which may be substantial in shallow lakes and running waters (Yang et al. 2020).

Between 2011 and 2020, Hg levels in streams and lakes of different areas of NHNP have been surveyed along with physical and chemical water variables, including the concentration and quality of DOC and suspended solids, in order to evaluate Hg availability and fractionation taking into account the THg:DOC ratios. DOC concentration affects the supply and bioavailability of Hg in aquatic systems, and the THg:DOC ratio determines the sorption of Hg to particles and dissolved compounds, which in turn influence the kinetics of Hg reduction, the transference of Hg to the sediments, and its entry to food webs (Ravichandran 2004; Bravo and Cosio 2019; Branfireun et al. 2020; Gèntes et al. 2021). Until now, total mercury (THg) and speciated Hg levels ($Hg^{2+}$, $Hg^0$, and $CH_3Hg$) have been assessed in different water bodies of NHNP alternatively through cold vapor atomic absorption and fluorescence spectroscopy (CVAAS and CVAF, respectively) (Soto Cárdenas et al. 2018a and references therein). In a survey of aquatic systems performed at the western sector of the BR, during the eruption of the PCCVC in 2011, high THg levels were detected, ranging from 41 to 363 ng $L^{-1}$ in streams and from 16 to 268 ng $L^{-1}$ in lakes. High Hg availability in these systems was indicated by the large THg:DOC ratios (from 60 to 1205 ng $mg^{-1}$ in streams and from 52 to 785 ng $mg^{-1}$ in lakes) (Table 8.1, Soto Cárdenas et al. 2018a). Noteworthy, much lower THg concentrations (0.7–9.4 ng $L^{-1}$) were recorded in the water column of BR a few years afterward (2018 to 2020) suggesting that the large inputs of Hg deposited in the catchment during the last eruption of PCCVC may have been stabilized in the terrestrial environment and buried in lake sediments. In fact, the low THg levels detected in porewater (<0.5–22.6 ng $L^{-1}$; Pérez Catán et al. 2016) could be taken as an indication of Hg immobilization in the sediments. In the eastern section of the park, levels of THg were much lower (0.48–52 ng $L^{-1}$ in streams and 0.02–56 ng $L^{-1}$ in lake water), resulting in THg:DOC ratios from 0.8 to 105 ng $mg^{-1}$ and from 0.005 to 55 ng $mg^{-1}$, respectively (Soto Cárdenas et al. 2018a; Arcagni et al. 2019).

High THg concentrations have been detected also in other sites of Nahuel Huapi lake influenced by human-associated pollution. THg concentrations as high as 1750 (±160) ng $L^{-1}$ and 640 (±20) ng $L^{-1}$ were recorded in surface waters of Nahuel Huapi lake in front of the outlet of the sewage treatment plant of San Carlos de Bariloche City and in Villa la Angostura City port, respectively (Pérez Catán et al. 2003).

In natural waters from the BR catchment, THg concentrations measured after the last eruption of the PCCVC in 2011 (Table 8.1) are among the highest concentrations reported in the literature for pristine sites and comparable to those found in impacted environments (Shanley et al. 2008; Kocman et al. 2011). $Hg^{2+}$ was the

**Table 8.1** Water parameters (mean ± SD) of aquatic systems of Brazo Rincón catchment (Lake Nahuel Huapi)

| | Streams | | Lake Pire | | Brazo Rincón (Lake Nahuel Huapi) | | | |
|---|---|---|---|---|---|---|---|---|
| | Mean ± SD | Range | Mean ± SD | Range | Mean ± SD | Water column | Upper layers | Deeper layers |
| DOC (mg $L^{-1}$) | 0.60 ± 0.23 | 0.21–1.02 | 0.48 ± 0.01 | 0.41–0.55 | 0.36 ± 0.06 | 0.31–0.51 | 0.32–0.51 | 0.31–0.33 |
| TSS (mg $L^{-1}$) | 7.07 ± 6.72 | 0.027–22.067 | 3.7 ± 0.7 | 3.22–4.22 | 1.11 ± 0.71 | 0.44–2.79 | 0.50–2.79 | 0.44–1.46 |
| THg (ng $L^{-1}$) | 146.08 ± 109.70 | 40.7–363.0 | 78.4 ± 20.9 | 63.6–93.1 | 126.4 ± 79.5 | 16.8–268.0 | 114.0–268.0 | 16.8–35.6 |
| PTHg (ng $L^{-1}$) | 29.1 ± 19.5 | 7.1–67.3 | 43.2 ± 6.6 | 38.5–47.8 | 40.1 ± 28.7 | 0.3–76.6 | 46.9–76.6 | 0.3–0.9 |
| FTHg (ng $L^{-1}$) | 117.0 ± 95.0 | 14.6–303.8 | 35.2 ± 14.3 | 25.1–45.3 | 86.4 ± 55.1 | 16.5–202.1 | 62.5–202.1 | 16.5–35.3 |
| log $K_d$ | 4.96 ± 0.61 | 3.94–6.03 | 5.6 ± 0.04 | 5.5–5.6 | 5.34 ± 0.67 | 4.03–5.93 | 5.54–5.93 | 4.03–4.70 |
| $Hg^{2+}$ (ng $L^{-1}$) | 145.2 ± 109.5 | 40.4–362.1 | 77.9 ± 20.9 | 63.2–92.8 | 123.1 ± 77.4 | 16.7–262.8 | 110.7–262.9 | 16.7–35.5 |
| $CH_3Hg$ (ng $L^{-1}$) | 0.126 ± 0.09 | 0.04–0.30 | 0.12 ± 0.04 | 0.09–0.15 | 0.06 ± 0.05 | 0.01–0.16 | 0.05–0.16 | 0.01–0.02 |
| THg:DOC (ng $mg^{-1}$) | 324.52 ± 349.58 | 60.3–1204.4 | 172.3 ± 79.0 | 116.5–228.2 | 348.1 ± 226.7 | 51.9–784.8 | 279.6–784.8 | 51.9–106.8 |
| $CH_3Hg$:DOC (ng $mg^{-1}$) | 0.26 ± 0.26 | 0.06–0.83 | 0.27 ± 0.14 | 0.17–0.37 | 0.16 ± 0.10 | 0.03–0.31 | 0.12–0.31 | 0.03–0.06 |
| $Hg^0$ (ng $L^{-1}$) | 0.80 ± 0.37 | 0.22–1.33 | 0.25 ± 0.86 | 0.19–0.32 | 3.33 ± 2.46 | 0.12–6.63 | 3.14–6.63 | 0.12–0.14 |
| %$Hg^{2+}$ | 99.20 ± 0.37 | 98.65–99.76 | 99.50 ± 0.19 | 99.36–99.63 | 97.79 ± 1.31 | 96.14–99.59 | 96.14–98.09 | 99.14–99.59 |
| %$Hg^0$ | 0.71 ± 0.37 | 0.18–1.31 | 0.35 ± 0.20 | 0.20–0.50 | 2.16 ± 1.32 | 0.38–3.81 | 1.82–3.81 | 0.38–0.74 |
| %$CH_3Hg$ | 0.10 ± 0.05 | 0.04–0.21 | 0.15 ± 0.01 | 0.14–0.16 | 0.05 ± 0.03 | 0.03–0.12 | 0.03–0.09 | 0.03–0.12 |

DOC (dissolved organic carbon), TSS (total suspended solids). Mercury species: THg (total Hg), PTHg (total particulate Hg), FTHg (filtered total Hg), $CH_3Hg$ (methyl Hg), $Hg^{2+}$ (ionic Hg), $Hg^0$ (dissolved gaseous elemental Hg)], log Kd (partitioning coefficient between the particulate PTHg and aqueous phase FTHg (Kd = PTHg/FTHg), THg:DOC and $CH_3Hg$:DOC ratios (THg and $CH_3Hg$ relative to DOC, respectively). Upper lake layers (0–60 m), deeper lake layers (70–90 m) prevailing species in all waters, accounting for up to 99.8 % of THg. $CH_3Hg$ levels were down to 0.12% of THg. This probably relates to the combination of high precipitation, low temperature, and steep slopes that promote rapid runoff preventing Hg methylation in soil and streambeds (Soto Cárdenas et al. 2018a). DOC levels were low, fluctuating between 0.31 and 1.02 mg $L^{-1}$. THg:DOC ratios ranged from 50 to 1204, indicating high $Hg^{2+}$ availability, with high partitioning coefficients (log Kd: Hg availability for binding particles). Overall, low DOC levels and high $Hg^{2+}$ concentrations in these systems promote a high adsorption of $Hg^{2+}$ to abiotic and biotic particles. In fact, particulate THg (PTHg) took values up to 76.6 ng $L^{-1}$ and was associated with inorganic particles in the streams and with phytoplankton in the lakes (Soto Cárdenas et al. 2018a).

In streams, Hg fractionation and speciation related directly with DOM terrestrial signatures (high molecular weight and aromatic DOM), indicating coupled Hg-DOM inputs from the catchment. In BR, DOM quality and photochemical and biological processing were found to determine Hg fractionation, speciation, and vertical levels. Dissolved gaseous Hg (DGM) reached higher values in BR (up to 3.8%), particularly in upper layers likely due to photochemical weathering resulting in the photolysis of Hg-DOM complexes and reduction of $Hg^{2+}$ (Fig. 8.1; Soto Cárdenas et al. 2018a).

**Box 8.1 Information**

*Trends arising from different Hg studies in aquatic systems of Nahuel Huapi National Park*

(i) $Hg^{2+}$ comprises up to 98% of the THg concentrations the water phase, while those of $CH_3Hg$ are below 3%

(ii) THg levels in waters directly relate with terrestrial DOC signatures, particularly in streams, indicating lateral transport of tied DOC-Hg inputs from the terrestrial environment

(iii) THg:DOC ratios in waters are high in a global context, indicating remarkable high Hg availability (mostly as $Hg^{2+}$)

(iv) High THg levels coincide with chlorophyll *a* peaks in the water column of lakes, indicating high Hg levels stored in phytoplankton, whereas in streams Hg is adsorbed to inorganic particles

(v) $CH_3Hg$ levels in streams relate with allochthonous DOM inputs indicating a terrestrial source, while in lakes, the higher levels in deep layers can be attributed to diffusion from and resuspension of sediments

(vi) In the depth profile of lakes, internal processing (photo- and biodegradation) tracked through optical DOC proxies, was found to favor the production of dissolved gaseous mercury ($Hg^0$). Particularly, the optically thin water column of deep ultraoligotrophic lakes promotes high light penetration and photolytic transformations of Hg

## Mercury in Lacustrine Sediments

Mercury deposited directly on the lake surface is rapidly scavenged by settling particles and sequestered to the sediments, and Hg deposited on the watershed is transported to the lakes mainly bound to OM and fine-grained mineral matter (Kainz and Lucotte 2006; Cooke et al. 2020 and references therein). Its cycling and distribution between the sediment and the water phase can be physically, chemically, and/or biologically mediated and, therefore, may be influenced by several factors such as pH, dissolved oxygen concentration, temperature, redox conditions, nutrients, and complexing agents (Ullrich et al. 2001; Ravichandran 2004; Bravo and Cosio 2019; Branfireun et al. 2020).

The prevalence of inorganic and organic $Hg^{2+}$ complexes, as well as redox processes involving $Hg^{2+}$ and $Hg^0$, and the partitioning of Hg between particulate, aqueous, and gaseous phases are important in the sediments (Beckers and Rinklebe 2017). Methylation of $Hg^{2+}$ can occur in the sediments and in the water column of stratified lakes, with higher net methylation rates occurring mainly at the oxic/anoxic interface (reviewed in Dranguet et al. 2017; Bravo and Cosio 2019; Gallorini and Loizeau 2021). Moreover, $CH_3Hg$ from terrestrial sources may contribute to the pool of Hg present in riverine and lacustrine sediments. In this sense, lake sediments dominated by OM derived from the catchment show higher $CH_3Hg$ concentrations but lower rates of in situ methylation, in contrast with sediments dominated by autochthonous OM (Branfireun et al. 2020 and references therein).

In NHNP, differences in Hg background levels in sediment cores from different lakes were associated with lake productivity. Background mercury levels in sediment cores obtained in Bahía López (BL; Nahuel Huapi lake), Moreno West, and Traful lakes ranged from <0.08 to 0.210 (± 0.044) μg $g^{-1}$ in preindustrial times, and twice as much in modern times. Higher Hg concentrations were detected in sediment cores of lakes El Trébol (7.02 ± 0.84 μg $g^{-1}$), Escondido (2.80 ± 0.36 μg $g^{-1}$), and Morenito (1.52 ± 0.32 μg $g^{-1}$). These lakes are small, shallow (maximum depth <12 m), and closed systems and display higher productivity than deep lakes of the area (OM between 22.5 and 26.5%). In the deep, ultraoligotrophic Nahuel Huapi lake branch BR, the highest THg concentration was recorded in the uppermost layer (1.10 ± 0.27 mg $g^{-1}$) corresponding to the period 1987–1994, displaying a positive correlation with OM (Ribeiro Guevara et al. 2002, 2003, 2005).

Studies focused on the THg content in suspended particles in the water column of different sites in Nahuel Huapi lake collected along with sediment cores indicated that both OM and Hg inputs from the catchment are highly seasonal and variable depending on the precipitation pattern. In this lake, the concentration of Hg in the suspended load was higher in the wettest sector (BR) compared to the dryer sites Puerto Cisne (PC) and the coast of San Carlos de Bariloche City (BRC) (Ribeiro Guevara et al. 2002), suggesting that the steep precipitation gradient observed in the area reflects in higher deposition and mobilization of materials from the landscape at the wetter BR catchment.

The production of $CH_3Hg$ from $Hg^{2+}$ in lake sediments was evaluated through laboratory experiments with the short-lived radioisotope 197Hg ($t^{1/2}$ = 64.14 h) using surface sediments and biofilms from different lakes of NHNP (Pérez Catán et al. 2004, 2007; Ribeiro Guevara et al. 2004, 2009). Mercury methylation potentials showed a seasonal trend due to variations in biotic and abiotic contributions. An increased abiotic contribution was observed during winter, at a moment of low biological productivity. Slightly higher methylation potentials were recorded in autumn, possibly associated with fresh terrestrial inputs due to increased seasonal runoff (Pérez Catán et al. 2009). Such inputs may include $CH_3Hg$ produced in the terrestrial environment (soil) and could promote changes in the concentration and quality of the OM and nutrient pools, influencing Hg methylation processes in lakes (Bravo et al. 2017). Experiments on mercury methylation using radiolabeled $Hg^{2+}$ showed that top sediments and rocks yielded lower methylation rates than autotrophic and heterotrophic biofilms, stressing the importance of biotic production of $CH_3Hg$ (Pérez Catán et al. 2011).

## 4 Mercury Trophodynamics in Lakes

Mercury entry, speciation, cycling, and accumulation in food webs are influenced by a multiplicity of variables, such as levels of $Hg^{2+}$ in the dissolved phase, DOM concentration and quality, levels of $Hg^{2+}$and $CH_3Hg$ at the base of the food web, activity of microbial methylators, among others. In addition, several ecological features such as primary productivity, habitat use, bioenergetics, and food web structure affect the efficiency of Hg uptake and its biomagnification (Arcagni et al. 2018; Eagles-Smith et al. 2018; Braaten et al. 2020; Chételat et al. 2020). In lakes, the habitat where fish feed can affect the amount of Hg that they accumulate. In large, deep lakes with extended pelagic zones, fish belonging to the same trophic position feeding in the pelagic habitat usually present higher Hg than fish foraging in the littoral (Chételat et al. 2011; Gèntes et al. 2021, and references therein). In contrast, in some lakes, fish feeding in the littoral habitat show higher Hg concentrations than pelagic-feeding fish (e.g., Chumchal et al. 2008; Finley et al. 2016; Hanna et al. 2016). Differential bioaccumulation of Hg between pelagic and benthic habitats may be related to Hg concentrations in prey and/or with differences in bioenergetic processes at the base of the food web (Kidd et al. 2003; Karimi et al. 2016).

The structure of pelagic food webs of deep Andean Patagonian lakes is simple and characterized by high endemicity and low diversity and abundance of organisms (Modenutti et al. 1998, 2010; Chap. 3). Limiting nutrient and C concentrations as well as high solar radiation and low temperature are the main challenges faced by pelagic communities. The picoplanktonic fraction is dominated by heterotrophic bacteria and picocyanobacteria (Callieri et al. 2007; Izaguirre et al. 2014; Soto Cárdenas et al. 2014, 2019; Gerea et al. 2019). Phytoplankton is mainly composed by nanoplanktonic flagellates (*Chrysochromulina parva*, *Rhodomonas lacustris*, etc.), dinoflagellates (*Gymnodinium* spp., *Dynobryon* spp., *Peridinium* spp.), and diatoms (i.e., *Aulacoseira granulata*, *Rhizosolenia eriensis*, and *Cyclotella stelligera*) (Diaz et al. 1998; Modenutti et al. 1998; Queimaliños et al. 1999, 2002; Soto

Cárdenas et al. 2014). Mixotrophy (photosynthesis + phagotrophy) is a common trait to different species of flagellates and ciliates (Hansen et al. 2019), and, in Andean lakes, this nutrition mode allows compensating the C and nutrient limitation through bacteria consumption and/or through the photosynthesis of endosymbiotic algae (Queimaliños et al. 1999; 2002; Modenutti et al. 2010; Gerea et al. 2017). The zooplankton community is composed of ciliates, rotifers, and crustaceans including cladocerans (*Bosmina longirostris*, *Ceriodaphnia dubia*, and the less abundant *Daphnia* sp.), calanoid (*Boeckella gracilipes*), and cyclopoid copepods (Modenutti et al. 1998; Arcagni et al. 2015).

Benthic food webs are also simple and in littoral areas characterized by the presence of the submerged macrophytes *Myriophyllum* sp. and *Nitella* sp., the emergent sedge *Schoenoplectus californicus*, and periphyton communities dominated by diatoms. The littoral zoobenthos assemblages are usually composed of insect larvae and nymphs of Trichoptera, Ephemeroptera, Plecoptera, Odonata, and Diptera (fam. Chironomidae, midges); *Chilina* sp. (snail) and *Diplodon chilensis* (mussel) mollusks; *Aegla* spp. (crab), *Samastacus spinifrons* (crayfish) and *Hyalella* sp. (amphipod) crustaceans, and annelids (hirudineans and oligochaetes) (Modenutti et al. 1998; Arcagni et al. 2013a, 2015). Fish assemblages include five native species, *Percichthys trucha* (creole perch), *Galaxias maculatus* (small puyen), *G. platei* (big puyen), *Olivaichthys viedmensis* (velvet catfish), and *Odontesthes hatcheri* (Patagonian silverside), and three introduced species of salmonids, *Salmo trutta* (brown trout), *Oncorhynchus mykiss* (rainbow trout), and *Salvelinus fontinalis* (brook trout) (Arcagni et al. 2015, and references therein).

Pelagic microbial assemblages of Andean lakes are included in the plankton fractions with higher THg levels (Table 8.2). Heterotrophic and autotrophic bacteria, flagellates (*Gymnodinium* spp.), and mixotrophic ciliates (*Stentor araucanus* and *Ophrydium naumanni*) incorporate substantial amounts of dissolved $Hg^{2+}$ passively (adsorption) and actively (bacteria consumption or attachment), differing in Hg internalization and thus, in their potential for Hg transfer (Soto Cárdenas et al. 2014, 2018b, 2019). Morphological features of protists such as surface area and surface to volume ratio, as well as the concentration and quality of DOC control their passive uptake (Diéguez et al. 2013; Soto Cárdenas et al. 2014). Active incorporation of $Hg^{2+}$ depends on bacteria consumption in the mixotrophic ciliates, or on bacteria association to surface. Therefore, $Hg^{2+}$ accumulated by pelagic protists can transfer to higher trophic levels through their consumption by plankton and fish, can regenerate to the dissolved phase, and/or be transferred within the water column and to the sediments by vertical movements (diel vertical migration), excretes and particle sinking (debris, resting stages, etc.) (Chiaia-Hernandez et al. 2013; Aydin et al. 2015, Soto Cárdenas et al. 2018b, 2019). In aquatic food webs, microbes have a chief influence on Hg pathways and speciation, since they can scavenge different Hg species, reduce $Hg^{2+}$, oxidize $Hg^0$ and produce and degrade $CH_3Hg$ (Gregoire and Poulain 2014, 2018; Bravo and Cosio 2019). The conversion of inorganic $Hg^{2+}$ to $CH_3Hg$ in the environment is mostly carried out by microbial assemblages present in anaerobic sediments, saturated soils, anoxic bottom waters, and also in oxygen-deficient microenvironments (biofilms and microbial flocks) in oxygenated

**Table 8.2** Range, mean, and standard deviation of total mercury (THg) and methyl Hg ($CH_3Hg$) concentrations in biota from Lake Nahuel Huapi and surrounding sites (BL, Bahía Lopez; BR, Brazo Rincón; DH, Dina Huapi; BRC, San Carlos de Bariloche City)

| | | THg ($\mu$g g$^{-1}$) | | $CH_3Hg$ ($\mu$g g$^{-1}$) | | |
|---|---|---|---|---|---|---|
| Taxa | Site | Mean (± SD) | Range | Mean (± SD) | Range | Reference |
| *FISH* | | | | | | |
| *Galaxias maculatus* (small puyen) | BL | 0.169 ± 0.077 (11, 30) | 0.031–0.286[a] | 0.139 ± 0.041 (5, 19) | 0.089–0.188 | 1, 2, 3 |
| | BR | 0.278 ± 0.118 (23, 73) | 0.077–0.650[a] | 0.184 ± 0.079 (7, 48) | 0.027–0.272 | 1, 2, 3 |
| | DH | 0.257 ± 0.264 (17, 48) | 0.040–1.00[a] | 0.051 ± 0.016 (6, 20) | 0.024–0.071 | 1, 2, 3 |
| *Galaxias platei* (big puyen) | BR | 0.673 ± 0.454 (8, 8) | 0.298–1.72[b] | 0.745 ± 0.444 (6, 6) | 0.354–1.76 | 1, 2 |
| | DH | 1.43 (1, 1) | | – | – | 1 |
| *Olivaichthys viedmensis* (velvet catfish) | BL | 0.499 (1, 1) | | 0.447 (1, 1) | – | 1, 2 |
| | DH | 0.719 (1, 1) | | 0.748 (1, 1) | – | 1, 2 |
| *Oncorhynchus mykiss* (rainbow trout) | BL | 0.087 ± 0.047 (7, 7) | 0.039–0.180[b] | 0.082 ± 0.041 (7, 7) | 0.033–0.157 | 1, 2 |
| | BR | 0.166 ± 0.063 (5, 5) | 0.105–0.264[b] | 0.147 ± 0.073 (4, 4) | 0.106–0.257 | 1, 2 |
| | DH | 0.090 ± 0.063 (10, 10) | 0.027–0.252[b] | 0.049 ± 0.020 (8, 8) | 0.019–0.082 | 1, 2 |
| *O. mykiss*, juveniles | BL | 0.083 ± 0.016 (6, 6) | 0.066–0.102[b] | 0.066 ± 0.005 (4, 4) | 0.060–0.072 | 1, 2 |
| | DH | 0.146 ± 0.177 (3, 3) | 0.041–0.350[a] | 0.022 (1, 1) | – | 1, 2 |
| *Percichthys trucha* (creole perch) | BL | 0.706 ± 0.296 (9, 9) | 0.137–1.09[b] | 0.714 ± 0.390 (8, 8) | 0.145–1.27 | 1, 2 |
| | BR | 1.13 ± 0.56 (13, 13) | 0.573–2.33[b] | 1.08 ± 0.41 (11, 11) | 0.559–1.83 | 1, 2 |
| | DH | 0.430 ± 0.317 (5, 5) | 0.090–0.761[b] | 0.070 ± 0.01 (2, 2) | 0.063–0.076 | 1, 2 |
| *Salmo trutta* (brown trout) | BL | 0.256 ± 0.174 (2, 2) | 0.133–0.379[b] | 0.235 ± 0.186 (2, 2) | 0.105–0.364 | 1, 2 |
| | BR | 0.216 ± 0.110 (23, 23) | 0.053–0.467[b] | 0.213 ± 0.144 (13, 13) | 0.054–0.536 | 1, 2 |
| | DH | 0.241 (1, 1) | | – | – | 1 |
| *S. trutta*, juveniles | BR | 0.259 ± 0.051 (5, 13) | 0.206–0.335[a] | 0.243 ± 0.033 (3, 7) | 0.220–0.280 | 1, 2 |
| *Macroinvertebrates* | | | | | | |
| *Aegla* sp. (crabs) | BL | 0.246 ± 0.172 (13, 84) | 0.110–0.786[b] | 0.096 ± 0.014 (2, 9) | 0.086–0.106 | 1, 2 |
| | BR | 0.592 ± 0.722 (12, 63) | 0.168–2.72[b] | 0.223 ± 0.026 (3, 17) | 0.198–0.249 | 1, 2 |

(continued)

**Table 8.2** (continued)

|  |  | THg (μg g$^{-1}$) |  | $CH_3Hg$ (μg g$^{-1}$) |  |  |
|---|---|---|---|---|---|---|
| Taxa | Site | Mean (± SD) | Range | Mean (± SD) | Range | Reference |
| *Chilina* sp. (snails) | BL | 0.564 ± 0.401 (12, 66) | 0.115–1.04[b] | 0.018 ± 0.003 (2, 20) | 0.016–0.019 | 1, 2 |
|  | BR | 0.166 ± 0.064 (38, 188) | 0.069–0.296[b] | – | – | 1 |
| *Diplodon chilensis* (mussel) | BL | 0.160 ± 0.118 (15, 77) | 0.063–0.560[b] | 0.018 ± 0.01 (5, 28) | 0.013–0.031 | 1, 2 |
| *Hyalella* sp. (amphipods) | BL | 0.424 ± 0.033 (2, 22) | 0.400–0.447[b] | – | – | 1 |
|  | BR | 0.346 ± 0.163 (4, 123) | 0.234–0.584[c] | – | – | 1 |
| Insect larvae (various spp.) | BL | 0.374 ± 0.496 (26, 195) | 0.042–1.15[c] | 0.007 ± 0.001 (2, 24) | 0.0062–0.0081 | 1, 2 |
|  | BR | 0.326 ± 0.192 (29, 335) | 0.062–0.806[c] | 0.034 ± 0.004 (3, 86) | 0.031–0.039 | 1, 2 |
| *Samastacus spinifrons* (crayfish) | BL | 0.113 ± 0.040 (15, 30) | 0.058–0.191[b] | 0.083 ± 0.020 (5, 14) | 0.061–0.109 | 1, 2 |
|  | BR | 0.480 ± 0.251 (18, 33) | 0.185–1.10[b] | 0.561 ± 0.448 (6, 9) | 0.165–1.43 | 1, 2 |
|  | DH | 0.123 ± 0.086 (6, 8) | 0.038–0.249[b] | 0.080 ± 0.058 (4, 6) | 0.029–0.136 | 1, 2 |
| *Aquatic Communities* |  |  |  |  |  |  |
| Biofilm | BL | 0.4454 (1) | – | – | – | 1 |
|  | BR | 0.128 ± 0.064 (2) | 0.083–0.173[d] | – | – | 1 |
| Plankton, fraction 10–53 μm | BL | 18.4 ± 14.6 (2) | 8.07–28.7[d] | – | – | 1, 2 |
|  | BR | 81 ± 123 (4) | 0.930–260[d] | – | – | 1, 2 |
|  | DH | 26.4 ± 1.6 (2) | 25.2–27.1[d] | – | – | 1, 2 |
| Plankton, fraction 53–200 μm | BL | 4.76 ± 4.44 (5) | 0.372–9.81[d] | 0.0057 (1) | – | 1, 2, 3 |
|  | BR | 7.14 ± 10.92 (6) | 0.340–31.2[d] | 0.004 ± 0.001 (3) | 0.003–0.006 | 1, 2, 3 |
|  | DH | 8.37 ± 7.77 (4) | 1.47–19.1[d] | 0.0039 (1) | – | 1, 2, 3 |
| Plankton, fraction > 200 μm | BL | 3.82 ± 6.71 (5) | 0.344–15.8[d] | 0.012 ± 0.02 (2) | 0.011–0.013 | 1, 2, 3 |
|  | BR | 1.88 ± 2.71 (10) | 0.131–9.56[d] | 0.016 ± 0.009 (4) | 0.009–0.029 | 1, 2, 3 |
|  | DH | 10.1 ± 16.3 (6) | 1.00–43.1[d] | 0.130 ± 0.174 (2) | 0.007–0.253 | 1, 2, 3 |

(continued)

**Table 8.2** (continued)

| Taxa | Site | THg ($\mu g\ g^{-1}$) Mean (± SD) | THg ($\mu g\ g^{-1}$) Range | $CH_3Hg$ ($\mu g\ g^{-1}$) Mean (± SD) | $CH_3Hg$ ($\mu g\ g^{-1}$) Range | Reference |
|---|---|---|---|---|---|---|
| *Tree leaves* | | | | | | |
| Submerged tree leaves debris (various spp.) | BL | 0.085 ± 0.045 (6) | 0.038–0.144[e] | – | – | 4 |
| Tree leaves from the canopy (various spp.) | BL | 0.056 ± 0.026 (9) | 0.0300–0.119[e] | – | – | 4 |
| | BR | 0.052 ± 0.043 (15) | 0.016–0.174[e] | – | – | 4 |
| *Lichens* | | | | | | |
| *Candelaria vitellina* (crustose areolate lichen) | BRC | – | 1.65–2.67[f] | – | – | 5, 6 |
| | BRC | – | 0.28–0.92[f] | – | – | 5, 6 |
| *Hypotrachyna brevirhiza* (foliose lichen) | BRC | – | 1.02–1.51[f] | – | – | 5, 6 |
| | BRC | – | 0.30–0.85[f] | – | – | 5, 6 |
| *Physcia adscendens* (hooded rosette lichen) | BRC | – | 0.18–2.81[f] | – | – | 5, 6 |
| *Protousnea magellanica* (beard lichen) | BRC | – | 0.096–0.280[f] | – | – | 5, 7 |
| *Usnea* sp. (old man's beard) | BR | 0.197 ± 0.020 | 0.09–0.23[f] | – | – | 8, 9 |
| | BR | 0.669 ± 0.089[f] | – | – | – | 5 |
| | DH | 0.203 ± 0.015[f] | – | – | – | 8, 9 |

Total Hg and $CH_3Hg$ concentrations are presented on a dry weight basis. Numbers in parentheses indicate the number of samples and organisms analyzed; in the case of plankton, biofilm, and tree leaves, only the number of samples is indicated

References: 1-Arcagni et al. (2017); 2-Arcagni et al. (2018); 3-Rizzo et al. (2014); 4-Juárez et al. (2016); 5- Ribeiro Guevara et al. (2004); 6-Ribeiro Guevara et al. (1995); 7- Bubach et al. (2001); 8- Bubach et al. (2012); 9- Bubach et al. (2014)

[a] no head and guts

[b] muscle

[c] whole

[d] bulk sample

[e] leaves

[f] thalli

surface waters (Lehnherr et al. 2011; Lamborg et al. 2014; Gionfriddo et al. 2016; Bravo and Cosio 2019; Gallorini and Loizeau 2021).

The transference of Hg between abiotic and biotic compartments is a critical aspect to understand and delineate the concentration dynamics in the pelagic (water column) and benthic (bottom) habitats. Soto Cárdenas et al. (2018a) showed that in different aquatic systems from the BR catchment (Nahuel Huapi lake), THg concentrations ranged from moderate to high (16.8–268 ng $L^{-1}$), with $Hg^{2+}$ as the predominant Hg species (from 96.1 to 99.6 %), whereas $CH_3Hg$ concentrations were low (from 0.03 to 0.12%) indicating reduced methylation in the catchment and in the aquatic systems despite high Hg availability (Soto Cárdenas et al. 2018a). In BR, most of the THg measured in the plankton was $Hg^{2+}$, and its concentration decreased abruptly with trophic level, from phytoplankton to zooplankton and small puyen. In turn, the concentration of $CH_3Hg$ was comparatively very low but increased with trophic level (Rizzo et al. 2014; Arcagni et al. 2017). This is due to the fact that at the base of the trophic chain, $Hg^{2+}$ adsorbs to the membrane of algal cells, while $CH_3Hg$ is sequestered in their cytoplasm. Zooplankton digest algal cells readily incorporating their cytoplasmic content and excreting the membranes. In this way, $CH_3Hg$ is more efficiently transferred than $Hg^{2+}$ from phytoplankton to zooplankton grazers (Mason et al. 1995, 1996; Diéguez et al. 2013).

In Nahuel Huapi lake, THg and $CH_3Hg$ concentrations in fish species vary by foraging habitat, increasing with a greater proportion of benthic diet over pelagic diet (Arcagni et al. 2017, 2018). Native creole perch, a predominantly benthic feeder, shows higher THg and $CH_3Hg$ levels in muscle than introduced salmonids such as the more pelagic rainbow trout and the benthopelagic brown trout, although the three species are positioned at the highest trophic level of the food web. The native galaxiid known as big puyen is a benthic feeder positioned at a lower trophic

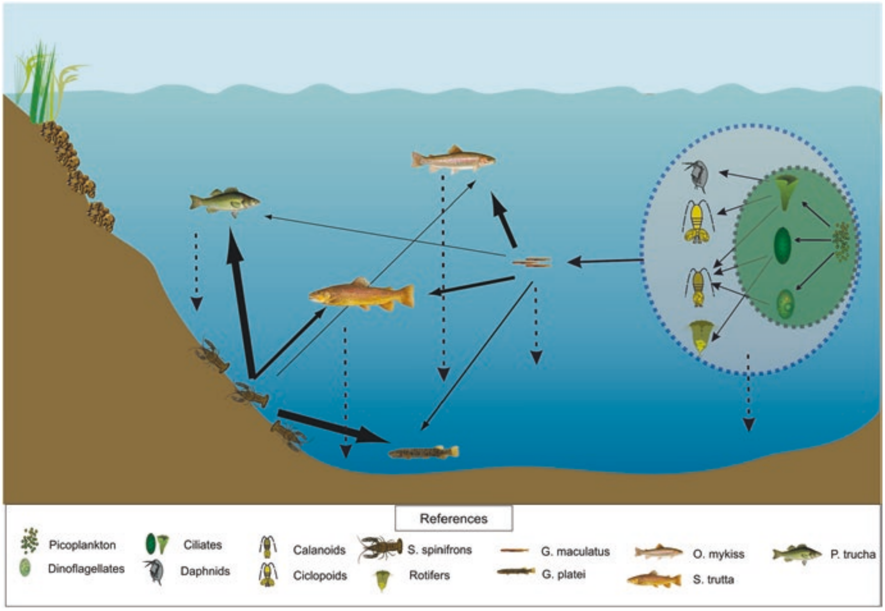

**Fig. 8.3** Mercury transference in a typical food web of deep ultraoligotrophic Andean Patagonian lakes. Solid line arrows: $CH_3Hg$ pathway. Dashed line arrows: THg pathway (arrow thickness indicates magnification). Species references: *Percichthys trucha* (creole perch), *Galaxias maculatus* (small puyen), *Galaxias platei* (big puyen), *Salmo trutta* (brown trout), *Oncorhynchus mykiss* (rainbow trout), *Samastacus spinifrons* (crayfish)

level than the creole perch and exotic trout and usually displays higher THg and $CH_3Hg$ concentrations than the salmonids (Table 8.2, Fig. 8.3; Arcagni et al. 2015, 2017, 2018; Juncos et al. 2015).

The distinctive patterns in THg and $CH_3Hg$ bioaccumulation in different fish species could be explained by the niche segregation among native and introduced fish (Juncos et al. 2015) leading to different pathways of Hg transference: a benthic pathway (from sediments to benthic macroinvertebrates and benthivorous fish) and a pelagic pathway (from water, plankton, foraging fish, and piscivorous fish) (Fig. 8.3). At first sight, in deep Andean lakes, the pelagic habitat appears to be a more important source of Hg to the food web than the benthic habitat. THg levels in size-fractionated plankton samples typically decrease from the smaller fraction (10–53 μm) comprising mixotrophic ciliates and dinoflagellates, toward the fraction between 50 and 200 μm composed of large mixotrophic ciliates and rotifers, and the fraction >200 μm made up of copepods and cladocerans (Table 8.2; Arribére et al. 2010; Rizzo et al. 2011, 2014; Arcagni et al. 2013b, 2015, 2017), whereas macroinvertebrates belonging to benthic food webs usually present lower THg concentrations than the different plankton fractions (Table 8.2, Fig. 8.3; Arcagni et al. 2015, 2017). Noteworthy, regardless of the high THg concentrations present in different plankton size fractions, their $CH_3Hg$ concentrations are low, ranging from 0.021 to 6.8% of THg, while those in benthic macroinvertebrates are higher, between 1.5 and 100 %. These levels are reflected in the different Hg pathways mentioned before. In the pelagic trophic chain, *O. mykiss*, which feeds mostly on the planktivorous forage fish *G. maculatus*, shows lower THg and $CH_3Hg$ concentrations than the benthivorous native fishes *P. trucha*, *G. platei*, and *O. viedmensis*; however, over 62% of THg is in the form of $CH_3Hg$ (mean concentration: 81 ± 13%). In the benthic trophic chain, *P. trucha* and *G. platei* feed mostly on crayfish and other macroinvertebrates, resulting in the highest THg and $CH_3Hg$ concentrations in the fish community, with $CH_3Hg$ over 64% (89 ± 12%) of THg. *Salmo trutta*, which relies on *G. maculatus* at smaller sizes and shifts to a diet composed mainly of crayfish at larger sizes, shows intermediate THg concentrations between native fishes and *O. mykiss* and also a high proportion of $CH_3Hg$ (88 ± 11%%) (Fig. 8.3; Juncos et al. 2015; Arcagni et al. 2017, 2018). The trend of increasing $CH_3Hg$ and decreasing $Hg^{2+}$ with trophic level in the pelagic habitat ends in the *G. maculatus–O. mykiss* feeding link, as the latter species displays lower to similar concentrations of $Hg^{2+}$ and similar $CH_3Hg$ level compared to its prey. On the contrary, in the benthic habitat, both THg and $CH_3Hg$ increase with increasing trophic level, from insect larvae and crayfish to *P. trucha* and *S. trutta*, with intermediate $CH_3Hg$ and THg concentrations between *O. mykiss* and *P. trucha*, which has lower $Hg^{2+}$ concentrations than its benthic diet (crayfish) and similar concentrations than its pelagic diet consisting of *G. maculatus* (Fig. 8.3; Arcagni et al. 2017, 2018).

In the sediments, $Hg^{2+}$ can be methylated and taken up by benthic organisms by direct absorption from the porewater and/or from the water passing through the gills during respiration, or indirectly through deposit-feeding, thus reaching benthic predatory fish (Chen et al. 2014 and references therein). Benthic biofilms may also contribute to the $CH_3Hg$ present in benthic feeders, since they are chemically particular microsites where microbial assemblages include microorganisms with physiological capability for Hg methylation and demethylation (Bravo and Cosio 2019). In fact, biofilms from different Andean Patagonian lakes have been shown to yield high methylation rates, as was discussed in Sect. 4. For example, crayfish, which are polytrophic, may acquire Hg remobilized from the benthic habitat by feeding on small benthic macroinvertebrates and on plant and animal detritus (Rudolph 2002) and then transfer the accumulated Hg to the benthivorous fish, *P. trucha*, or *G. platei* (Fig. 8.3).

## 5 Perspectives on Mercury Cycling in Andean Patagonian Catchments in a Context of Climate Change

In the northern and central sections of Patagonia, long-term drying and warming trends are driving major hydrological changes affecting headwaters (Masiokas et al. 2008; Garreaud et al. 2013; Barros et al. 2015). Annual and seasonal temperature and precipitation records indicate significant warming and decreasing precipitation since 1912. These climate alterations have been implied as drivers of large-scale changes in Andean Patagonian catchments including the drastic reduction of glacial fields (Chap. 4), the decrease in streams and rivers discharge (Masiokas et al. 2008, 2009; Wilson et al. 2018; Chap. 9), as well as the increase in the frequency of extended wildfires (Mundo et al. 2017). Such climate-related changes may interact with those arising from population and urban expansion at a regional scale and also be influenced by global changes (Chaps. 9 and 11). Hydroclimatic changes stress internal processes of terrestrial, wetland (Chap. 10), and aquatic ecosystems and affect the fluxes of materials from the catchment to the fluvial network. Reduced precipitations and changes in the precipitation pattern lead to changes in the strength of the linkages within the catchment that reflect on the mobilization of materials from the terrestrial environment toward the fluvial network, thus affecting biogeochemical cycling (Battin et al. 2008). Terrestrial inputs drive physicochemical factors controlling the dynamics and function of aquatic communities. Freshwaters of Andean Patagonia, particularly the oligo-/ultraoligotrophic systems, rely tightly and respond rapidly to seasonal terrestrial inputs of C and nutrients (Queimaliños et al. 2012, 2019). Due to the affinity of Hg for organic ligands present in natural OM, the C and Hg biogeochemical cycles are tied (Ravichandran 2004; Lavoie et al. 2019). C and Hg pools are washed from the land by precipitation and runoff, draining through the aquatic network. In fact, changes in the volume, type, and timing of precipitation reflect rapidly in the OM, nutrient, and Hg pools of aquatic systems (Queimaliños et al. 2012, 2019; Rizzo et al. 2014). The magnitude of the impact would depend on the severity of the climate changes and also on the hydrogeomorphic features of the catchment (topography, vegetation cover, soil development, lake morphometry, etc.). If warming and drought patterns sustain over time, aquatic systems will undergo profound changes due to reduced inputs of water and materials from the catchment affecting several different aspects. Their morphometry may suffer changes due to reduction in depth and surface, implying the loss of littoral zones and related communities. Water retention time in lakes may increase following the reduction of water inputs and water turnover. This would impact water clarity due to DOM photobleaching (Queimaliños et al. 2019; Chap. 3), ultimately enhancing light penetration, particularly the UV wavelengths. In turn, the higher impact of underwater irradiance will increase photochemical processing. Autotrophic and heterotrophic production will likely react to changes in the terrestrial C and nutrients supplies, causing community changes. Moreover, the potential loss of cold-adapted organisms would imply modification of the trophic pathways in which they intervene. All these changes are likely to synergize, affecting the diversity and function of aquatic communities which in turn impact biogeochemical cycling.

Progress in glacial melting due to warming is expected to release large quantities of Hg locked in ice into the atmosphere and downstream ecosystems. However, it is possible that the opposite process will occur and that the glacier-to-vegetation succession already going on will increase the capture of atmospheric Hg, as it has been observed in areas experiencing glacial retreat (Wang et al. 2020). Additionally, the increase of wildfires may also contribute to unlocking Hg stored in natural forests; however, this would depend on concomitant changes in land cover and use due to population growth and urban expansion. In this regard, predictive models of future Hg emissions forecast significant perturbations due to increased wildfire Hg emissions, driven by the changes in climate, land use, and Hg anthropogenic emissions. Modeled scenarios for the 2000–2050 period indicate an increase of Hg emissions by 14% globally and by 18% in South America. The potential increase of Hg in terrestrial ecosystems in response to changes in global Hg anthropogenic emissions and deposition could enhance global Hg emissions due to wildfires. However, this scenario may depend on changes in land use by 2050, since agricultural land expansion in detriment of natural vegetation could decrease global Hg emissions from wildfires (Kumar et al. 2017).

The changes in Hg cycling are expected to be manyfold because Hg stored in ecosystems, processing rates, the lateral transport to aquatic end points, and methylation are climate-sensitive (Obrist et al. 2018). Evidence arising from sediment cores of a remote lake in southern Patagonia (Lake Hambre, Chile) suggests that Hg accumulation over the past 4000 years has been influenced by cyclic changes in climate, total solar irradiance, and lake productivity. The accumulation of Hg was higher in dry periods coinciding with high solar irradiance and lake productivity. In such environmental conditions and assuming low Hg fluxes from the atmosphere to the catchment and to the lake due to drier conditions, Hg burial in the sediments may have reduced the concentration of Hg in the water column as well as the evasion of $Hg^0$ from the system (Biester et al. 2018).

Given the global reach of Hg pollution, climate-related changes in Hg cycling in the Andean Patagonian region would have to be foreseen in the context of the evolution of global anthropogenic emissions, and, above all, having in mind the stochastic impact of volcanism which is the main source of Hg to regional landscapes. Nevertheless, it can be inferred that the forecasted warming and drying trends would increase Hg reemission due to enhanced photoreduction processes in terrestrial and aquatic systems, will decrease the lateral transport due to the reduction in the terrestrial-aquatic linkages, and will trigger changes in terrestrial and aquatic methylation due to the reduction of allochthonous C and Hg inputs and changes in microbial communities. Extreme events such as intense rainfall, floods, and droughts, among others, affect South America regardless of the seasons (Marengo et al. 2009). Exceptional floods, which have been forecasted as a component of climate change in the region, could provide unusual pulses of terrestrial materials to aquatic systems, whereas drought would reduce the land-water linkage, shrinking the flow of terrestrial materials within catchments.

These changes would impact the amounts of terrestrial C, nutrients, and Hg reaching the aquatic environment, which in turn drive the uptake, accumulation, and transfer of Hg in aquatic food webs. Moreover, changes in terrestrial subsidies are expected to affect the structure and dynamics of aquatic communities, with potentially profound consequences on the trophic pathways of the biogeochemical cycling of Hg.

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# Part II

# Socioecological Aspects of Freshwater Ecosystems in Patagonia

# Chapter 9
# Hydrologic Systems, Water Uses, and Emerging Conflicts Around Freshwater Availability in Patagonia

**Adriana B. Urciuolo and Rodolfo J. Iturraspe**

## 1 Introduction

Patagonia occupies a vast territory in southern South America, between 36° S and 56° S (Fig. 9.1), in which the Andes Mountain range is the main geographical unit. Hydrologic feedbacks in mountainous regions of the world control the availability of water, influence the distribution of vegetation, dominate biogeochemical fluxes, and contribute to global and regional climate variability (Bales et al. 2006); the Andean region not being an exception.

The westerlies are the prevailing winds in the middle latitudes, between the high-pressure areas of the subtropics and the low-pressure areas over the poles (Toggweiler 2009). Patagonia is the only continental terrain in the Southern Hemisphere offering resistance to this humid wind corridor. The Southern Andes mountain range, as a natural barrier magnifying the disturbance of westerlies, determines an asymmetric water distribution, rendering Patagonia one of the regions of the world with the greatest disparity in water availability. The hyperoceanic environment of the Chilean Pacific archipelago shows the highest precipitation records at sea level, especially between latitudes 46° S and 52° S, reaching 7220 mm $year^{-1}$ in Guarello Island at 50° 22′ S (Aravena and Luckman 2009), while in some sectors of the extra-Andean plateau, precipitation drops to only 150 mm $year^{-1}$.

The Wet Andes is a climatic-glaciological-hydrological subregion of the Andes that extends south of the Dry Andes between ~36° and 56° S (Lliboutry 1998). It matches approximately with the Patagonian forest ecoregion of Argentina described by Morello et al. (2012), and part of the temperate rainforest of Chile (Veblen 2007).

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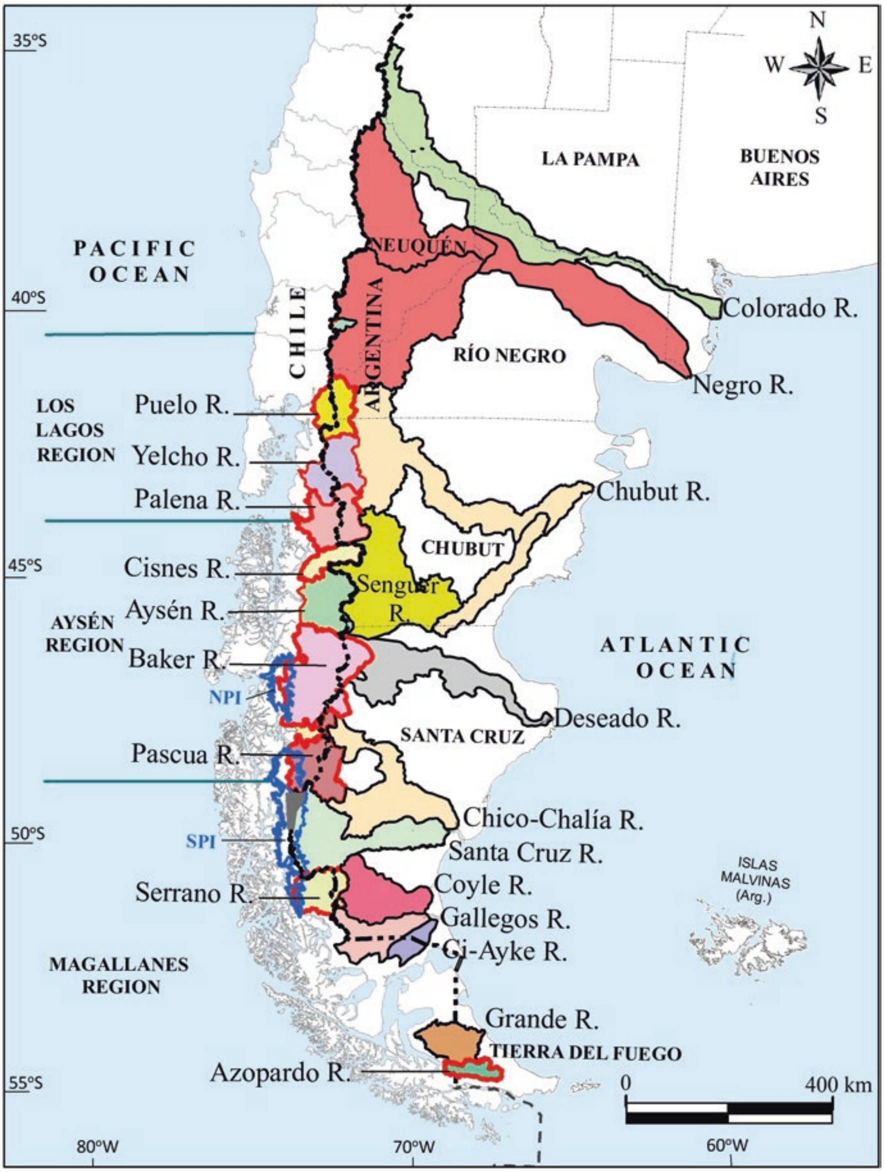

**Fig. 9.1** Study Area: Main river basins of Patagonia

The northern mountainous area of the province of Neuquén corresponds to the transition from the Dry to the Wet Andes. On the other hand, extra-Andean Patagonia involves mostly the steppe ecoregion, also including part of the Austral subregion of Monte of plains and plateaus ecoregion (Morello et al. 2012) that extend in eastern areas of the provinces of Neuquén, part of Río Negro, and NE of Chubut (Fig. 9.1).

Chilean Patagonia is among the richest regions in the world in freshwater availability, in terms of both runoff and storage. In turn, rivers running from the mountain range to the Atlantic Ocean provide water for populations of dry extra-Andean Patagonia, as well as the sustenance of primary production, and hydroelectric energy for the National Electric Interconnection System of Argentina.

The colonization of the Patagonian territory, as from the last decades of the nineteenth century, was a process based on the forced displacement of its original inhabitants. Notably, the first settlements began from the south. Punta Arenas, founded in 1848, was a pioneer in this regard, achieving great prosperity due to its strategic location on the Magellan Strait. In 1865, settler-colonists arrived at the lower valley of the Chubut River to develop farming activities. Moreover, they explored this province and the North of Santa Cruz in search of mineral resources. In 1884, Ushuaia city was established in Tierra del Fuego Island.

Freshwater scarcity has been a major disadvantage for the population and development of broad swaths of extra-Andean Patagonia. In Chile, water scarcity was not an obstacle for colonization, but in addition to the rugged topography, different water manifestations, such as ice fields, glacier-shaped fjords, wetlands, and large rivers operated as geographical barriers conserving extensive wild zones.

This chapter analyzes the hydrological features of the great Patagonian region, from the Colorado River in Argentina and Puelo River in Chile to the south where Andean and extra-Andean zones differ completely, as well as water uses and the conflicts that have emerged around this essential element for life and for the development of the societies that populate Patagonia.

## 2 General Hydrological Features

### 2.1 Andean Hydrology

The Andes determine contrasting drainage patterns on each oceanic slope. At present, eastern Patagonian rivers transport less water and sediment than in the past (Isla et al. 2015). Mighty rivers fed by large ice systems in the late glacial period that flowed toward the Atlantic Ocean drastically reduced their runoff after glaciers receded, opening up new waterways to the Pacific Ocean (Thorndycraft et al. 2019; Coronato et al. 2004, 2009).

The Andes mountain range is the source of freshwater for Patagonia, by generating water surpluses that sustain runoff. The entire western slope fits this productive area since it is entirely humid, favored by the dominant western wind. The moderate altitude of the Andes in Patagonia partially allows the passage of moisture to the eastern slope. The seasonal migration of the Pacific anticyclone modulates the westerlies circulation. In winter, northward anticyclonic movement allows humid and cool air masses from the Pacific Ocean to enter, inducing the formation of typical cold fronts that produce persistent rain and snowfall from April to October (Ostertag and Cuello 2005). The intensity of this seasonal pattern diminishes toward the south and the Atlantic coast, since in the upper basins of Neuquén and Limay Rivers, winter precipitation (October-March) represents around 80% of the total annual (González et al. 2021) while in Evangelistas Island (52° S) and southward, no manifest rain seasonality is observed (Aravena and Luckman 2009). In turn, the west-east circulation of cyclonic depression affects the climate in Southern Patagonia, causing regular precipitation, notably in Tierra del Fuego.

Short rivers and rich drainage networks characterize the hydrography in the western Andean slope. River pathways are highly controlled by tectonics, which frequently determines rectangular drainage patterns. Major rivers are those having a transboundary basin that collect water from both sides of the mountain range, such as Puelo, Yelcho, Palena, Baker, and Pascua. Many small coastal streams, which are not discussed in this chapter, discharge into the Pacific Ocean from western massifs. The rugged topography restricts terrestrial communication, which often is only available through river valleys (see Chap. 16).

The Andean runoff has a pluvio-nival flow regime, with snow dominating on the eastern slope and rain prevailing on the west. Glaciers are major drivers of the behavior of rivers that drain the Patagonian Icefields. Mean annual flows increase progressively downstream with increasing drainage area because the entire Andean region is hydrologically active.

Even though there are very different environments in Patagonia, the water is characterized by generally being of good quality for different uses. Ice-snow-fed Andean rivers and lakes have very low contents of total dissolved solids (TDS) in waters and slightly alkaline conditions. The Andean Lakes often show oligotrophic to ultra-oligotrophic conditions, and high transparency, though glacier dynamics and torrential regimes tend to increase the river sediment load and turbidity (Diaz 1994, Chap. 3). Water chemical composition expressed in equivalents follows in general the relationships $Ca^{++} > Mg^{++} > Na^{+} > K^{+}$ and $HCO_3^{-} > SO_4^{=} > Cl^{-}$ (Diaz 1994; but see also Chap. 7).

Ashfalls from intense volcanic eruptions might produce disturbances in freshwaters, with increasing TDS values, sediment load, pH changes, and Hg deposition in lacustrine sediments (Iturraspe 2022; Chaps. 3 and 8). Andean basins present natural hydrological regulation systems combining different components that control seasonal runoff variability, floods, and sediment transport. The extension and the hydrological importance of these reservoirs vary for each basin.

## Hydrological Regulation Systems in the Andes

Snowpack, glaciers, lakes, wetlands, and soil are the main interactive components conforming regulation systems of the Andean basin hydrology, inducing peak flow reduction, as well as runoff persistence, even during prolonged periods of low precipitation.

Snowmelt from mountains is the major source of many of the regional water supplies, involving downstream processes, such as groundwater recharge and interactions with ecosystems, controlled by processes at higher elevations (Bales et al. 2006). It is a major feature in the Andean hydroclimatic system and is crucial for the distribution and preservation of other components of the cryosphere like glaciers and rock glaciers (Masiokas et al. 2020). Snowpack is a substantial water reservoir along the Andes, notably in Northern Patagonia, where winters are wet and summers are dry. The Patagonian Andes is the region with the largest seasonal snow accumulation in the Southern Hemisphere outside Antarctica. Snow accumulation depends on land elevation, but in addition, the wind drives both initial deposition and later redistribution, making up notable spatial variability, linked to land topography and vegetation. Even though the Patagonian Andes lose height southward, snow accumulation does not decrease since temperature and radiation both drop, and lower cols allow humid air from the west to pass through, lessening snowpack asymmetry between both sides of the ranges. Snow melting is a slow and gradual process that begins in late winter in lower valleys and concludes in early summer at higher elevations. Persistent snowmelt promotes infiltration and groundwater recharge. Warm weather intensifies melting, resulting in moderate floods. Extraordinary floods are caused by rain and notably by rain-on-snow events. The importance of glacier contributions to the runoff is related to the glacier area rate in the water basin, as well as on the rain regime, and the capacity of other existing regulation systems.

At lower latitudes than 45° S, glaciers exhibit small to middle size, occupying cirques in higher massifs and covering volcanic cones. In the Atlantic slope of the Andes, glaciers diminish in extension, as well as their incidence on the annual runoff. However, glacier flow can be significant at a seasonal level, during dry and warm periods, after depletion of the main water reservoirs. In contrast with snowmelt, the glacial flow does not depend on the precedent precipitation events.

Lakes fulfill hydrological functions in river basins as water reservoirs. The most significant Patagonian lakes have a glacial origin in the Pleistocene. Glacier terminals have molded current lake beds some hundreds of meters below sea level (e.g., Buenos Aires-Carrera, San Martín-O'Higgins, Viedma, Argentino, and Fagnano lakes). The largest Patagonian lakes located in the oriental Andean slope were shaped by Pleistocene glacier lobes. Instead, the western glacier terminus sculpted current Pacific fjords. Great lakes have a strong influence on downstream fluvial regimes, moderating peak floods, seasonal flow variability, and sediment transport, as the long residence time of water in big lakes favors silt and clay sedimentation.

During the twentieth century, the general ice retreating gave rise to new lakes of small and middle sizes. Between 1984 and 2016 and south of 36° S latitude, the number of glacial lakes[1] increased by 46%, from 2663 to 3889 units, and their area increased from 4458 to 4783 km$^2$ (Wilson et al. 2018). Most of the total area corresponds to the three biggest glacial lakes: Argentino, Viedma, and San Martín-O'Higgins, which comprise 3612 km$^2$.

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[1] Glacial lakes are those with a free surface in, under, beside, and/or in front of a glacier and originated from glacier dynamic and/or retreating processes of a glacier. The biggest lake of Patagonia (Buenos Aires-General Carrera) is not reached by this definition.

Wetlands are abundant in Chilean Patagonia, with peatlands and swamp forests predominating. Peatlands provide hydrological regulation and sediment control, as well as other benefits such as carbon sink and carbon storage (Iturraspe and Urciuolo 2021; Chap. 10). *Mallines* (wet meadows), small lakes, and peatlands (*Sphagnum* bogs, graminoid fens) are also present in the Eastern Andean slope (Chap. 10).

## *2.2 Extra-Andean Hydrology*

Semiarid to arid conditions prevail behind the oriental Andean foothills, wherein the steppe replaces the forest. Annual precipitation decreases fast eastward of the Andes to less than 200 mm. The water deficit at the central plateau determines large arheic and endorheic areas. Hence, no significant streams join the Andean rivers on their way to the Atlantic, and beyond them, surface water occurs as sporadic and endorheic streams or emerging springs that feed wet meadows and lakes. Extra-Andean lakes are shallower and smaller than those of glacial origin. Only 15 extra-Andean lakes extend over 15 km$^2$, and just 4 of them exceed 100 km$^2$. Most of the extra-Andean lakes unlinked to Andean outfalls are endorheic, and many of them dry up in summer due to the high water deficit.

From the south of the province of Mendoza to the south of the province of Santa Cruz, large basaltic plateaus formed by lava flows can be seen along the east side of the Andes (Mazzoni and Rabassa 2013). Many small lakes scattered on basaltic tablelands were generated by differential erosion processes that resulted in relief inversion. Wind and water have eroded old upper areas not covered by basalts, whereas basalt tablelands have resisted erosion (Ardolino et al. 2008). These water bodies are frequently found beneath the basalt bottom level, some tens of meters below the plateau's top, and their location allows groundwater feeding. Basalt's secondary porosity allows rainwater percolation to recharge these groundwater sources. In a similar process, on the sloping margins of the basaltic plateau, springs feed *mallines*, which concentrate available water for flora and fauna, therefore being of great production value for livestock farming (Chap. 10). In quaternary sediment deposits, lakes occupy depressions generated by deflation, receiving water supply from small basins that are hydrologically active only in winter.

Snow accumulation is normally low and unstable, having better development and persistence in the highlands. However, exceptional snowfalls have blocked roads and killed cattle, as happened in Santa Cruz, Tierra del Fuego, and southern Magallanes in July–August, 1995 (Endliger and Santana 1997).

Eastern Patagonian rivers follow long routes to the Atlantic Ocean, flowing through W-E-oriented valleys about 100 meters below the plateau level, which already existed in the Miocene, before the general Andean ascent (Groeber 1927; Ghiglione 2017).

Total dissolved solids (TDS) and nutrient concentrations in rivers on the Atlantic slope tend to grow downstream, such as the Senguer River, wherein TDS increases in one order of magnitude (Scordo et al. 2020). Minor variations can be seen along rivers having a large supply from the upper basin and poor downstream contributions; in the Santa Cruz River, for example, TDS levels are regularly about 30 mg $L^{-1}$ along its course.

Endorheic lakes on the plateau evidenced higher saline concentrations than Andean lakes (Quirós 1988) and very contrasting chemical features (Chap. 14). Water chemical composition in equivalents follows in general the relationship $Na^+ > Mg^{++} > Ca^{++} > K^+$and $Cl^- > > SO_4^{=} > > HCO^-$ (Diaz 1994).

Extra-Andean basin boundaries are often defined by hydrologically active areas rather than topography. However, meteorological anomalies can result in exceptional rainfalls on the arid plateau. Hence, some catchments are not constant in size and shape, but they change according to extreme event intensities (Serra 2006). The catastrophic flooding that occurred in Comodoro Rivadavia City (Ocampo et al. 2018) is an example. On March 31, 2017, after an exceptionally rainy period, a 232-mm rainfall recorded over 24 hours (245 mm being the local annual average) triggered flash floods on usually dry gorges, which descended from the plateau and converged in the city, resulting in floods, gullies, and landslides that devastated the urban area.

## 3 Main Economic Activities and Land Use in Patagonian Basins

The economy of Patagonia is defined by the exploitation of natural resources. Extensive sheep livestock has been the dominant identity in the Patagonian space, although it has lost its position in several regions surpassed by other activities such as hydrocarbon exploitation or tourism (Coronato et al. 2015).

Water is the necessary resource for the development of a productive economy, and livestock, agriculture, oil exploitation, mining, and tourism are the major economic sectors strongly connected to freshwater availability.

In northern Argentinean Patagonia, cultivating pastures and forages is mainly linked to cattle rearing; however, in the main river valleys, the intensive production of fruit and other crops has developed notably, such as in the Colorado River basin, wherein irrigated agriculture occupies more than 160,000 ha. Agriculture coexists with dense urbanization in sections of Neuquén and Limay River valleys. The fruit cultivating activity, in particular related to pome, fine fruits, and vineyards, is relevant within the so-called area of the *Alto Valle de Río Negro y Neuquén* (Gallina et al. 2012). This area, which includes the lower valleys of the Neuquén and Limay rivers, as well as the upper Negro River, is recognized for its productive development.

To the south, extensive sheep farming on natural pastures is the most extended rural production in Chubut, Santa Cruz, and north of Tierra del Fuego (TDF), now combined with cattle breeding. Agriculture is complementary to livestock, mainly in the irrigable valleys. The lower Chubut River valley includes the most populous communities in the basin. Fertile soils and water availability allow horticulture and livestock production. The Bajo de Sarmiento, in the alluvial fan of the Senguer River, is a center of intensive production, focusing on livestock, forage, fruit, and vegetables (Raguileo 2020).

In the Province of Santa Cruz, farms cultivate fruit trees, forage, and vegetables on the southern shore of Lake Buenos Aires where the town of Los Antiguos, the cherry national capital, is located (Cepparo 2013). Intensive agricultural and grazing operations have been focused in the Chico River valley (Vázquez and Mazzoni 2004), as well as around the cities of Perito Moreno and Gobernador Gregores. The Gallegos River basin is one of the areas with the highest livestock productivity in Southern Patagonia due to its physical characteristics (Vázquez and Mazzoni 2004).

The conditions of some reservoirs are appropriate to salmonid farming, such as in Alicurá, on the Limay River, where the expansion of this activity is planned (Chap. 13). Nevertheless, this could damage water quality and would be incompatible with tourism.

The oil industry is developed in different oil fields from Neuquén to TDF. The Colorado River basin includes one of the most productive areas in Argentina, covering about 18,000 km$^2$ (SPTIP 2013). Oil activity is also significant northwest of the Río Negro province, and it is one of the pillars of the regional economy of the Neuquén River basin, followed by agriculture (Forni et al. 2021). Close to the confluence of the Limay-Neuquén rivers, in addition to conventional oil fields, significant shale and tight deposits are being exploited (Villalba 2018).

The oil industry has a long history in the Chubut province, particularly along the San Jorge Gulf coast and, more recently, in the Senguer River basin. In Santa Cruz, it has also become the main economic activity. Main oil fields under exploitation are located in the San Jorge Gulf oil basin and the area near the town of Perito Moreno, as well as in the south of Santa Cruz and north of the province of Tierra del Fuego, where companies extract gas and oil from the Magallanes-Austral oil basin.

Mining takes place in several river basins of north Patagonia, as in the north and central zone of the Colorado River and the Alto Valle of Río Negro. The Province of Chubut has enacted legislation prohibiting open-pit metalliferous mining and the use of cyanide. Currently, several firms are exploring uranium.

In Santa Cruz, metalliferous mining for gold and silver is mainly concentrated in the Deseado Massif, in the center-north of the Province, as well as to the east of the town of Gregores, where companies use groundwater for their processes. According to the zoning endorsed by Provincial decree 1327, mining has become an activity of importance to the east of the mountain range. In the western Gallegos River basin, the firm "*Yacimiento Carbonífero Río Turbio*" is in charge of coal mining and electric generation through a thermal power plant.

North Patagonia involves valuable natural sites such as Lake Nahuel Huapi, and the cities of Bariloche and San Martín de los Andes. Recreational fishing is carried out along the Limay River, which has unique characteristics for this sport. Tourism has notably expanded in South Patagonia due to the scenic beauty of its aquatic environments and glaciers. Coastal areas have also gained importance during the last decades. In the province of Chubut, tourism is one of the main activities in mountain river basins as Puelo and Futaleufú. There are remarkable sites for nature/

cultural tourism in the province of Santa Cruz, such as the upper Pinturas River, a tributary of Deseado River, where the archaeological site "Cueva de las Manos" offers a rising tourist attraction. Los Glaciares National Park, declared a World Heritage Site by UNESCO, is located at the head of the Santa Cruz River basin. It preserves an extensive area including continental ice, lakes, rivers, and Andean-Patagonian forest. The Perito Moreno glacier acquires special relevance during the breaking period of its forward front.

Ushuaia, the capital city of TDF, is currently an international tourist destination based on its natural beauties and its status as a gateway to Antarctica. The river valleys near the city and the Lake Fagnano basin comprise natural landscape reserves. Likewise, the Tierra del Fuego National Park is a destination visited by thousands of tourists, where much of its attraction is based on its rivers, lakes, and wetlands.

Fisheries in the Santa Cruz, Serrano, and Grande River of TDF offer specimens that reach world-class sizes, favored by the good quality of the waters (Casalinuovo et al. 2019). Many cattle ranches have diversified activities toward sport fishing and agrotourism.

Chilean Patagonia is widely recognized for the diversity of sites of natural beauty and pristine state. Tourism is a key economic activity developed in all river basins. The Puelo basin has been declared an "Area of Tourist Interest" by the "Servicio Nacional Turístico" in 2007. The town of Futaleufú, next to the Futaleufú National Reserve, is a special destination for kayaking in Futaleufú's rapids. The Lake Yelcho area, with exceptional landscapes and fishing aptitude, presents transformations such as the arrival of new inhabitants, global investors, and tourism companies (Sáenz Passeron 2015).

The Austral road to the south, in the Aysén region, enabled tourism development and land trade with the rest of the country (DGA 2004b). Due to the impressive landscape of the Baker River, the activity in the basin is strongly oriented toward nature tourism. The number of international class fishing lodges has increased in later years.

The Pascua basin is one of the least inhabited and most difficult to access areas in the region, but nature tourism has also gained relevance. The Serrano basin has many sites of high scenic value (CENMA 2010). Among them are Torres del Paine and O'Higgins National Parks, which include a great part of the Southern Patagonian Icefield. Both have been declared as Areas Under Official Protection.

Agricultural-livestock activities take place near main towns, such as in the Coyhaique commune, where cattle production is very significant as well as in Puerto Aysén (Gobierno de Aysén 2009).

In the Baker basin, agriculture is developed in several places such as Chile Chico and Puerto Ibáñez as well as cattle rising in most of the coastal area of Lake General Carrera. A lower economic activity takes place in the wild area of the Pascua basin, corresponding to subsistence livestock production. In the town of O'Higgins, local organizations are concerned with maintaining a clean and pristine environment to promote three main activities: aquaculture fishing, agriculture, and tourism.

In the upper Serrano basin, extensive livestock production is developed in *mallines* of the Cerro Castillo area (CENMA 2010). The Magallanes Region accounts for 56% of the sheep production in Chile. This activity has historical relevance in Punta Arenas as well as in northern TDF, wherein 15,000 ha of pastures are irrigated (ODEPA 2019).

Mining projects in the Aysén region are located near Lake General Carrera, where the activity was developed for a long time. Oil and gas extraction is localized in the Magallanes Region, on both sides of the Magellan Strait. In TDF, this industry led to the establishment of oil towns, such as Cerro Sombrero and Cullen.

## 4 Main River Basins of Patagonia

Each hydrographic basin presents both biophysical and social particularities. For this reason, these aspects are described below for the most important basins in the region, which range in terms of the complexity of physical characteristics and hydrological processes, as well as in the water uses, social development, and related conflicts. As a result, each instance had to be examined on a different degree of analysis.

Published information has been compiled, and new data has been generated such as flow series updates, which are necessary for recent flow changes identification. Free access data, available on official web pages, have been processed to this end. The main data source in Chile has been the Dirección General de Aguas (DGA), and in Argentina, the Sistema Nacional de Información Hídrica (SNHI).

In concordance with other authors (e.g., Rivera et al. 2018), the streamflow yearly cycle was defined from April to March; then the period 2010–2019, used to compare flow series, starts on April 1, 2010, ending on March 31, 2020. For the Baker, Pascua, and Santa Cruz rivers, the annual cycle was defined from September to August because of the strong ice melting and lake regulations influence over the drainage regime.

### *4.1 Colorado River Basin*

This interjurisdictional river basin includes areas in the Argentine provinces of Neuquén, Rio Negro, Mendoza, La Pampa, and Buenos Aires, adding up to an extension of 46,500 km$^2$. The coordinated water management is carried out through the Colorado River Interjurisdictional Committee (COIRCO), the first basin committee in Argentina (Perl et al. 2005).

The Colorado River, at the northern limit of the Argentinean Patagonia (Fig. 9.2), results from the confluence of the Grande and Barrancas rivers, which collect the runoff from the Andes of southern Mendoza and northern Neuquén. This is a transitional sector between the Dry and Wet Andes, wherein precipitation increases southward, as mountains altitude decreases (Lauro et al. 2016). The precipitation is

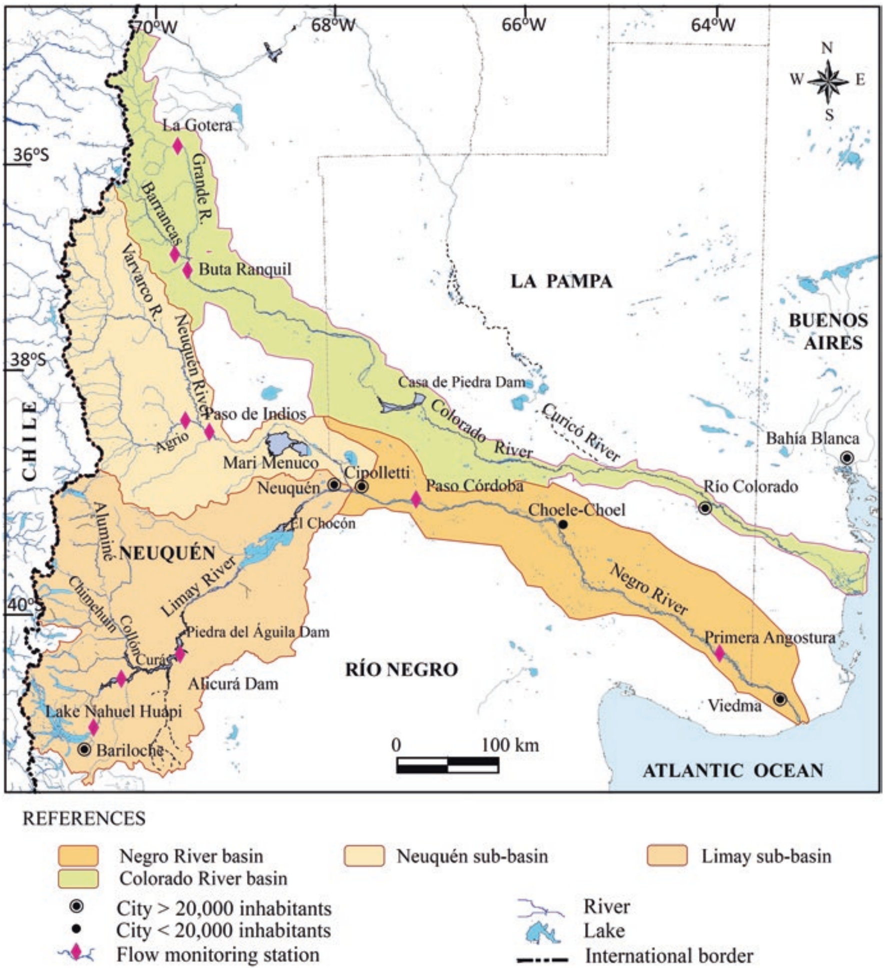

**Fig. 9.2** Colorado and Negro river basins

characterized by a wet season that extends from late fall to late winter and by a marked dry summer. After the wet season, snowmelt drives the flow regime.

From April 2010 to March 2019, the average flow of the Colorado River at Buta Ranquil, 20 km downstream of the Grande-Barrancas confluence, was determined to be 84 $m^3s^{-1}$, much lower than the historical flow average of 140 $m^3s^{-1}$. This decadal reduction is the result of a strong precipitation decrease as from 2010 in the Andes both in Argentina and Chile between 30° S and 37° S, which Garreaud et al. (2020) qualified as “mega-drought” because of its exceptional persistence.

Annual precipitation in Buta Ranquil averaged 200 mm in 2015–2019, indicating the semiarid to arid environmental character of this settlement. Eastward, the climate becomes dryer, and flow contributions are null or sporadic. From Buta

Ranquil, the Colorado River flows southeastward through a desolated and arid landscape, which is interrupted only by a few small towns such as Rincón de los Sauces and Catriel.

Casa de Piedra dam regulates the flow regime rendering water available during the low-water period (January–March). Downstream of the dam, farm production depends on the water accumulated in winter, but no irrigation channels are connected to the dam. A minimum 15 $m^3 s^{-1}$ flow is guaranteed for granting water supply to populations and the environmental flow.

Eastward of the dam, the riverbed meanders on the floodplain, which averages a 0.05% slope. By the northern margin, the waterless riverbed of Curicó stream joins the main valley. Until the early twentieth century, it was the terminal section of Desaguadero River, a significant flow collector of streams running from the Andes. Dams, which were made upstream for intensive irrigation, mostly in the province of Mendoza, disrupted this drainage leading to interprovincial disputes and claims by the province of La Pampa, which has been seriously affected.

Climate improves toward the Atlantic, as annual precipitation rises to 480 mm in the town of Hilario Ascasubi, 55 km from the shore. However, there is a 280 mm $year^{-1}$ water deficit, and annual rainfall varies widely, ranging from 240 to 730 mm (Aliaga et al. 2014).

The Colorado River has formed a constructive-destructive delta with destructive ascendency, dominated by waves on mesotidal coasts (Spalletti and Isla 2003). Holocene sea-level fluctuations caused salinity in the delta plain both in soil and in groundwater. For this reason, the water table has been depressed through drainage channels, to improve rural production by irrigation. In addition, dredging and rectifications modified the mouth sector, and three intakes on riversides derive water through a system of five main irrigation channels.

Apart from being a complex ecosystem wherein aquatic life develops in its various expressions, the Colorado River and its tributaries provide water for different uses, such as human consumption, irrigation of crops, raising livestock, oil exploitation, recreation, and hydropower (Jovanovich et al. 2014). Water usage for irrigation accounts for almost 98% of total consumption, while human supply, mining, and oil use each represent 1% (Aumassanne 2019).

The main environmental threat to this basin in northern Neuquén is related to oil spills due to numerous wells and related premises in this zone. Although oil activity is not currently a major user of water in the basin, it involves environmental problems due to spills and soil erosion that increased sedimentation levels (FAO 2015), contributing to the silting of channels and reservoirs. Another problem affecting the basin is soil salinization of irrigated areas, which is frequently linked to irrigation infrastructure deficiencies and/or land abandonment. In addition, except for the irrigated valleys and the mountain range, many zones exhibit varying degrees of desertification due to overgrazing and soil degradation caused by oil activities. COIRCO conducts frequent monitoring, and studies show that the quality of water for different uses is acceptable (Alcalde 2018).

## 4.2 Negro River Basin

The Negro River collects eastern Andean flows between 36° 30′ S and 41° 30′ S latitude through two large flow systems: the Neuquén River in the North and the Limay River in the South (Fig. 9.2). These rivers' confluence, close to the city of Neuquén, forms the Negro River, which is the most important fluvial system in the Argentinean Patagonia, and one of the main ones of the country. Its total basin reaches 138,000 km$^2$. The significance of this river owes to both its flow magnitude and the comprehensive use of its water. They make up a geographic space, covering almost the entire Province of Neuquén and part of the Río Negro and Buenos Aires Provinces, which is managed in an integrated manner through the Interjurisdictional Watershed Authority (AIC).

The surface water is used for human consumption by major population centers in the provinces of Neuquén and Río Negro. Water is extensively used for irrigation throughout the basin. Approximately 17,950 ha of the Neuquén River basin and 8600 ha of the Limay River basin are irrigated in the Province of Neuquén. The "Alto Valle" irrigation system begins at the Ballester dam on the Neuquén River. It irrigates around 60,000 ha of land, the majority of which is used for fruit cultivation (Lozeco 2014). Water is also employed for secondary and tertiary recovery in the oil industry.

The dams built on the Limay and Neuquén rivers to take advantage of their flow and control floods produce 37.8% of the country's hydroelectric energy, making up the largest hydropower generating area in Argentina, with 4647 MW production capacity (IAEA 2019).

### Neuquén River

The Neuquén River extends south of the Colorado River basin, draining an area of 40,000 km$^2$. The upper runoff begins in the northwest of the basin, at the Cajón de Los Chenques where steep hills flank deep and narrow valleys and torrential streams.

Small glaciers remain over 3000 m altitude in the Domuyo volcano, as well as some ice patches on the top of the Copahue volcano. The Neuquén River continues southward for about 100 km, increasing its flow by tributaries from the west. The Varvarco River supplies the main contribution from the east, with sources in Lake Varvarco, the largest in the basin (23 km$^2$), and on the western side of the Domuyo volcano. Before changing its trajectory southward, the Neuquén River meets the Agrio River, the last important affluent. The Agrio has a basin area of 9800 km$^2$ with headwaters in the Copahue volcano (3000 m a.s.l.). Magmatic effusions of the volcano confer notable acidity to waters in the upper course, with pH < 4 (Pedrozo et al. 2010; Chap. 7). These water features neutralize before the river reaches the Neuquén River. The Agrio's average flow was 75 m$^3$ s$^{-1}$ for 1990–2019 at Bajada del Agrio; the highest daily flow was 1250 m$^3$ s$^{-1}$, and the lowest was only 7.5 m$^3$ s$^{-1}$over this period, suggesting a considerable flow variability.

Downstream of the Agrio River confluence, the Covunco Stream is the last tributary. At this section, the Neuquén River reaches its full flow and heads eastward through the steppe.

The pluvio-nival drainage regime presents a winter flood period triggered by seasonal rain and another, in November–December, when snowmelt produces greater monthly flows. The fast river's response to heavy rainfalls yields the major peak floods in winter. Following the melting phase, low flows in February and March are the result of sparse rainfalls and weak basin regulation.

In the years 2010–2019, the Neuquén average flow of 179 $m^3\ s^{-1}$ in the Paso de Indios section accounted only for 57.5% of that recorded during the 1903–2009 period. This runoff decrease indicates that the regional drought affecting the Colorado upper basin had also reached the Neuquén River sources.

The "Cerros Colorados" (CC) hydropower complex is located on the low Neuquén River. It was built to supply water and mitigate floods. The complex consists of the Portezuelo Grande dam, Los Barreales Reservoir (which serves as a flood attenuator in winter and a water reserve in summer), Loma de la Lata and Mari Menuco dams (fed by the former reservoir), and Planicie Banderita dam including its associated hydroelectric plant with a capacity of 450 MW (AIC 2021). In 2006, the Neuquén River suffered the highest recorded historical flood, which put the populations of the lower valley and the Upper Río Negro Valley at risk. The CC Complex played a central role in regulating that extreme event.

Water is used for human consumption in the basin's major populations, as well as for intensive agriculture, the oil industry, and other industrial operations. In the Province of Neuquén, irrigation consumes 55.7% of the Neuquén River's yearly flow, industry 4.5%, and human use 2.1% (MERN 2021). This basin is rich in shale deposits, such as the Vaca Muerta formation, the richest in unconventional hydrocarbon (UHC) in Patagonia. The amount of water utilized in shale production is modest, accounting for less than 1% of overall water use (Forni et al. 2021).

Problems related to water quality are associated with agriculture and urban use. Macchi et al. (2018) analyzed pesticide concentrations in waters from drainage channels discharging to the Neuquén River, in combination with macroinvertebrates biomonitoring in the irrigation channels. Chlorpyrifos was the key responsible for the decrease in taxon richness and abundances of this community, followed by azinphosmethyl and carbaryl. Nevertheless, due to the good dilution capacity of the main course, its water is suitable for human consumption after a normal treatment and can support aquatic and terrestrial life (López Ordierez 2016).

Given the significant urban expansion over the past 30 years, the presence of sewage discharges in the Confluencia area causes increasing bacteriological problems in the rivers (FAO 2015). Following the discovery of the UHC Vaca Muerta deposit, there has been an acceleration in oil development on productive soil (Rodil 2015). Many spill incidents due to poor maintenance of oil and gas pipelines, as well as illegal dumping of waste and sewage from the HC industry, have been reported (Álvarez et al. 2017). The AIC conducts monitoring at different points along the courses of the main rivers, to evaluate the quality of the resource concerning the activities carried out.

## Limay River

The Limay River begins at the outlet of Lake Nahuel Huapi (LNH) in the province of Río Negro and flows along 410 km to meet the Neuquén River. Several dams flood the course along 55% of the way. To the west of the main course, the Patagonian forest and the forest-steppe ecotone predominate, while the steppe extends to the east.

The forest canopy increases southward along the Andes in the province of Neuquén, indicating increasing water availability. In this Andean sector, the transverse structural faults shaped by the glacier dynamics determined at both sides of the Andes wide valleys connected by low cols that present extensive lakes, peatlands, and wet meadows. These morphological patterns are remarkable in the Limay upper basin, wherein more than 40 lakes (Pessacg et al. 2018), among which the LNH is the largest, control the Andean flow.

The LNH covers an area of 557 km$^2$, and its maximum depth is 464 m (Quirós 1988). It is a typical example of the ultra-oligotrophic lakes of the Andean Patagonia, with very low nutrient and salinity content in the water column (EC 34 ± 5 µS cm$^{-1}$) prevailing calcium and bicarbonate, slightly alkaline (pH: 7–7.6) and with high transparency (18 ± 0.6 m Secchi disc) (Pérez Catán et al. 2016). Its basin extends 4000 km$^2$ in area. From Mount Tronador, the highest peak in the basin, the Frías River runs along a valley covered by peatlands to end in Lake Frías, which links to LNH.

Figure 9.3 shows the contrast between the hydrological regime of the upper Limay River, well regulated by its lake system, and the Neuquén River in the Indios passage, which has poor lake regulation. On July 12, 2006, a sudden Neuquén flow peak reached 7845 m$^3$ s$^{-1}$, while the Limay River flow on the same date barely exceeded 400 m$^3$ s$^{-1}$, reaching 462 m$^3$ s$^{-1}$ on July 26, and remaining over 400 m$^3$ s$^{-1}$ for 35 days.

The course of Limay heads northeastward, and just at the Alicurá reservoir tail, it meets the Traful River, which is fed by the homonymous lake. Twenty km downstream of this junction, between Alicurá and Casa de Piedra dams, the Collón Curá

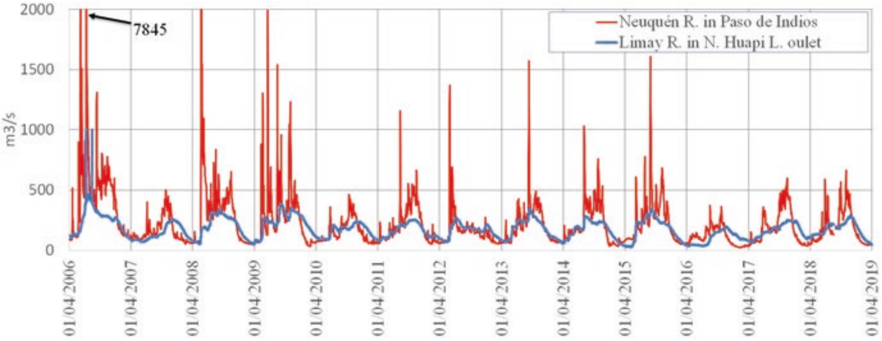

**Fig. 9.3** Daily flow comparison of Neuquén and Limay rivers

River discharges. It is the main Limay's tributary, which has its source in the Lake Aluminé, located north of the basin. This lake's outflow gives origin to the Aluminé River. It flows southward collecting tributaries from the west, such as the Chimehuin River, which drains the Lake Huechulaufquen, located at the foot of the Lanín volcano (3740 m a.s.l.), and flows marginally to the urban area of Junín de los Andes. In the southern river section, the Aluminé River changes its name to Collón Curá. These rivers' average flow is indicated in Table 9.1.

**Table 9.1** Patagonian rivers. Flow average during the historical record and in the last decade

| River Basin | River | Section | Lat S | Long W | Flow $m^3\ s^{-1}$ | Period | Flow 2010–2019 $m^3\ s^{-1}$ | Country |
|---|---|---|---|---|---|---|---|---|
| Colorado$^{E}$ | Grande | La Gotera | 35° 52′ | 69° 54′ | 103 | 1972–2020 | 58 | Arg |
| | Barrancas | Barrancas | 36° 48′ | 69° 54′ | 35 | 1961–2020 | 24 | |
| | Colorado | Buta Ranquil | 37° 06′ | 69° 44′ | 140 | 1940–2020 | 85 | |
| Negro$^{E}$ | Agrio | Bajada Agrio | 38° 25′ | 70° 03′ | 72 | 1953–2020 | 51 | |
| | Neuquén | Paso Indios | 38° 32′ | 69° 24′ | 300 | 1903–2020 | 179 | |
| | Chimehuin | Lake Huechulaufquen | 39° 47′ | 71° 13′ | 63 | 1971–2020 | 55 | |
| | Limay | LNH Outlet | 41° 03′ | 71° 09′ | 215 | 1921–2020 | 160 | |
| | Limay | Paso Flores (BJ C. Curá) | 40° 34′ | 70° 38′ | 265 | 1941–2020 | 211 | |
| | Limay | Paso Limay (AJ C. Curá) | 40° 31′ | 70° 27′ | 734 | 1903–1990 | – | |
| | Negro | P. Córdova | 39° 07′ | 67° 38′ | 925 | 1922–2020 | 569 | |
| | Negro | Primera Angostura | 40° 27′ | 63° 47′ | 734 | 1922–2020 | 543 | |
| Chubut$^{E}$ | Gualjaina | BJ Chubut | 42° 38′ | 69° 57′ | 13 | 1957–2020 | 6,2 | |
| | Chubut | Los Altares | 43° 53′ | 68° 24′ | 47 | 1943–2020 | 32 | |
| | Chubut | Valle Inferior | 43° 17′ | 65° 30′ | 34 | 1993–2020 | 24 | |
| Senguer$^{E}$ | Senguer | Los Molinos | 45° 59′ | 69° 28′ | 48 | 1987–2020 | 37 | |
| Deseado | Deseado | Route 39 | 45° 44′ | 69° 36′ | 1.6 | 2004–2020 | 1.3 | |
| Puelo$^{W}$ | Manso | Confluencia | 41° 35′ | 71° 41′ | 80 | 1965–2020 | 73 | |
| | Azul | L. Puelo | 42° 05′ | 71° 37′ | 80 | 2009–2020 | – | |
| | Epuyen | Angostura | 42° 11′ | 71° 24′ | 14 | 1951–2020 | 10 | |
| | Turbio | L Puelo | 42° 13′ | 71° 41′ | ~110 | Estimated | – | |
| | Puelo | Border | 42° 06′ | 71° 43′ | 161 | 2009–2020 | – | |
| | Manso | BJ Puelo | 41° 44′ | 72° 01′ | 225 | 2001–2020 | 209 | Chile |
| | Puelo | BJ Manso | 41° 46′ | 72° 03′ | 317 | 2003–2020 | 309 | |
| | Puelo | San Basilio | 41° 37′ | 72° 14′ | 625 | 2003–2020 | 599 | |
| Yelcho$^{W}$ | Futaleufú | Futal. Dam | 43° 10′ | 71° 37′ | 274 | 2007–2020 | – | Arg |
| | Futaleufú | BC Malito | 43° 27′ | 72° 07′ | 422 | 2002–2020 | 392 | Chile |
| | Yelcho | River mouth | 42° 59′ | 72° 43′ | ~570 | Estimated | – | |

(continued)

**Table 9.1** (continued)

| River Basin | River | Section | Lat S | Long W | Flow m$^3$ s$^{-1}$ | Period | Flow 2010–2019 m$^3$ s$^{-1}$ | Country |
|---|---|---|---|---|---|---|---|---|
| Palena$^{W}$ | Carrenleufú | Puente de Hierro | 43° 33′ | 71° 29′ | 49 | 1990–2020 | 35 | Arg |
| | Palena | AJ Rosselot | 43° 59′ | 72° 29′ | 821 | 2000–2020 | 771 | Chile |
| Cisnes$^{W}$ | Cisnes | P. Cisnes | 44° 46′ | 72° 37′ | 215 | 2001–2015 | – | |
| Aysén$^{W}$ | Coyhaique | Coyhaique | 45° 34′ | 72° 02′ | 5.4 | 2011–2019 | – | |
| | Simpson | AJ Coyhaiq | 45° 33′ | 72° 04′ | 43 | 1984–2020 | 43 | |
| | Mañihuales | BJ Simpson | 45° 24′ | 72° 30′ | 169 | 1992–2020 | 145 | |
| | Aysén | Puerto Aysén | 45° 24′ | 72° 37′ | 530 | 1985–2020 | 495 | |
| Baker$^{W}$ | Baker | L. Bertran | 47° 03′ | 72° 49′ | 560 | 2003–2020 | 549 | |
| | Nef | AC Baker | 47° 08′ | 73° 05′ | 52 | 2010–2016 | – | |
| | Baker | Bajo Ñadis | 47° 30′ | 72° 59′ | 941 | 2003–2020 | 942 | |
| | Baker | River mouth | 47° 47′ | 73° 30′ | ~1000 | Estimated | – | |
| Pascua$^{W}$ | Mayer | V O’Higgins | 48° 25′ | 72° 33′ | 115 | 1986–2020 | 110 | |
| | Pascua | AJ Quetru | 48° 09′ | 73° 05′ | 715 | 2003–2020 | 718 | |
| Serrano$^{W}$ | De las Chinas | Before L. Toro | 51° 15′ | 72° 32′ | 15.8 | 1991–2020 | 17.5 | |
| | Grey | BJ Serrano | 51° 11′ | 73° 01′ | 123 | 1981–2020 | 134 | |
| | Serrano | L. Toro | 51° 12′ | 72° 56′ | 85 | 1986–2020 | 96 | |
| | Serrano | River Mouth | 51° 20′ | 73° 07′ | 408 | 1981–2020 | 426 | |
| Chico$^{E}$ | Chalía | Ruta 40 | 49° 35″ | 71° 24′ | 2.0 | 2010–2020 | – | Arg |
| | Chico | Gregores | 48° 47′ | 73° 15′ | 22 | 2016–2020 | – | |
| Santa Cruz$^{E}$ | Las Vueltas | Chaltén | 49° 20′ | 72° 53′ | 51 | 1992–2020 | 51 | |
| | La Leona | L. Viedma | 49° 48′ | 72° 04′ | 261 | 1956–2020 | 288 | |
| | Santa Cruz | Charles Fuhr | 50° 15′ | 71° 54′ | 724 | 1955–2020 | 831 | |
| Coyle | Coyle | River mouth | | | 4 | Estimated | | |
| Gallegos$^{E}$ | Gallegos | Route 40 | 51° 54′ | 71° 36′ | 29 | 1993–2020 | 23 | |
| | Gallegos | River mouth | 51° 41′ | 69° 39′ | 20 | 2016–2020 | – | |
| Grande$^{E}$ | Grande | M. Behetty | 53° 49′ | 67° 56′ | ~45 | Estimated | – | |
| TDF | Grande | Route Y-85 | 53° 54′ | 68° 53′ | 30 | 1990–2020 | 32 | Chile |
| Fagnano-Azopardo$^{W}$ | Azopardo | Lake outlet | 54° 30′ | 68° 49′ | 49 | 2006–2020 | 47 | |

$^{E}$ Eastern slope drainage, $^{W}$ Western slope drainage, *BJ*/*AJ* before/after junction, *Arg* Argentina

The basin’s water is utilized for human consumption in several towns, tourism, irrigation, hydroelectric generation, and oil operations. Some reservoir conditions are favorable for salmonid farming, and there are proposals to expand the activity on a wider scale. Along the river, an intensive sport and recreational fishing activity is carried out. A considerable portion of the basin is protected by the Nahuel Huapi, Lanín, and Los Arrayanes National Parks, as well as the Limay River Protected Natural Area, which is a provincial reserve located in the southwest of the Río Negro province.

On the Limay River, five dams were constructed with the main objective of generating energy and controlling floods. Alicurá, Piedra del Águila, Pichi Picún Leufú, El Chocón, and Arroyito are mentioned following their position, from upper to lower basin (AIC 2021). Dams caused important changes in the regime and natural conditions of the Limay River by fragmentation of the fluvial ecosystem, turning hundreds of kilometers of the river into lake habitats (Pascual et al. 2007). Reservoir regulation, on the other hand, has reduced the frequency with which the waters spread over the floodplains, and this hydrological shift has resulted in significant changes in land use and the expansion of irrigated agriculture in the river valley, as well as other productive uses. This situation induced rapid growth of the local population. At present, the urban occupation of the River valley is advancing over the agricultural farms (FAO 2017).

### Negro River (Downstream of Neuquén-Limay Confluence)

From the confluence of Neuquén-Limay rivers to the Atlantic Ocean, the Negro River flows across the province of Río Negro along 550 km, through an enclosed valley that dissects the plateau, and reaches a maximum width of 17 km in front of Choele-Choel City. The gentle valley slope of 0.05% favors the development of meanders and islands. No tributaries meet on the way to its mouth in the Atlantic Ocean.

The dams on the Neuquén and Limay rivers allow the Negro River flow management. In Primera Angostura, 110 km upstream of the mouth, the Negro flow average is 973 $m^3 s^{-1}$. Data analysis revealed a significant flow decline since 2010, notably in Neuquén River (Table 9.1).

After the confluence of the Neuquén and Limay rivers, the Province of Río Negro has the highest population density. Viedma, the provincial capital, is located near the mouth of the Negro River. Intensive agricultural operations with irrigation are being developed in the “Alto Valle” region. Water is used for fruit-horticultural production in the middle valley, which is interspersed with forages and vineyards, whereas water is used for agriculture and livestock in the lower valley (Martínez 2009). Main issues concerning water quality are related to the agricultural production chain, as well as urban and industrial water demands. There are 110,000 hectares under production and several associated industries in the irrigated valleys of the province, where chemicals are extensively used (LIBIQUIMA-CITAAC 2016). Pollutants from a variety of sources are found in some waterways, primarily in the Alto Valle area, with those related to urban, industrial, and productive land uses being particularly significant (Migueles 2019). Normally, effluents are released into a drain collector, after treatment and authorization to dump. Pesticides have a greater impact in the Alto Valle than in the Medio and Bajo valleys (FAO 2015). The urban advance over the irrigation system in this area implies that sections of canals and drainage collectors are gradually influenced by urban pollution, which directly relates with the existence of legal and illegal urban settlements on the irrigation system’s side. Therefore, although monitoring showed that water in the entire river course was suitable for the uses it received, as well as for the aquatic life more than a decade ago (AIC 2008), this situation is likely to have worsened.

Also, as pointed out before, the inadequacy or lack of drainage, as well as insufficient maintenance, causes irrigation water to accumulate in the soil, resulting in salinization processes in the Valle Medio, particularly in abandoned areas (FAO 2014).

## *4.3 Chubut, Senguer, and Chico River Basins*

The Chubut River basin extends entirely in Argentina, mainly in the province of Chubut, and in small parts of the Río Negro and Santa Cruz Provinces. The Chubut, Senguer, and Chico rivers (Fig. 9.4) compose an extended hydrographic network whose connectivity has undergone modifications in the last century, because of climatic and anthropogenic factors.

Chubut River has typical features of the main Patagonian watercourses of the eastern slope, with the upper basin in the mountain range and a straight outfall toward the Atlantic. In the lower part of the river, it joins the Chico River, originally connected to the Senguer River through Lake Colhué Huapi. The source of the Senguer River lies in southwest Chubut and discharges into two large inland lakes: Musters and Colhué Huapi, the latter fed the Chico River in the past. This has not happened for several years, and thus Senguer River became an endorheic course, disconnected from the Chubut river basin.

### Chubut Main River Basin

The area of the Chubut main river basin (excluding the Chico sub-basin) reaches 46,600 km$^2$ involving contrasting landscapes where semiarid conditions prevail. The upper Chubut River basin extends in the Río Negro and Chubut Provinces, east of the Puelo and Futaleufú upper basins, which include the wettest area of this Andean sector in the Argentinean side of the Andes. The Chubut headwaters' location results in drier conditions and a lower water yield than in the adjacent Andean upper basins. In the western Chubut basin, the precipitation that reaches 1000 mm year$^{-1}$ rapidly decreases eastward, giving place to the forest-steppe ecotone complex, which is part of the northern forest subregion of the Andean forest ecoregion (Morello et al. 2012). Forest patches, *mallines*, bare soils, and steppe vegetation make up the land cover there.

Processes related to snow accumulation and snowmelt, in combination with the pluvial precipitation, are drivers of the flow regime in the upper basin. Glacier contributions are negligible, since the glacier extension in the basin is only 1.2 km$^2$, with rock glaciers accounting for 0.24 km$^2$ (Secretaría de Ambiente y Desarrollo Sustentable 2019). The river presents two flood periods related to winter rainfalls and spring snow melting.

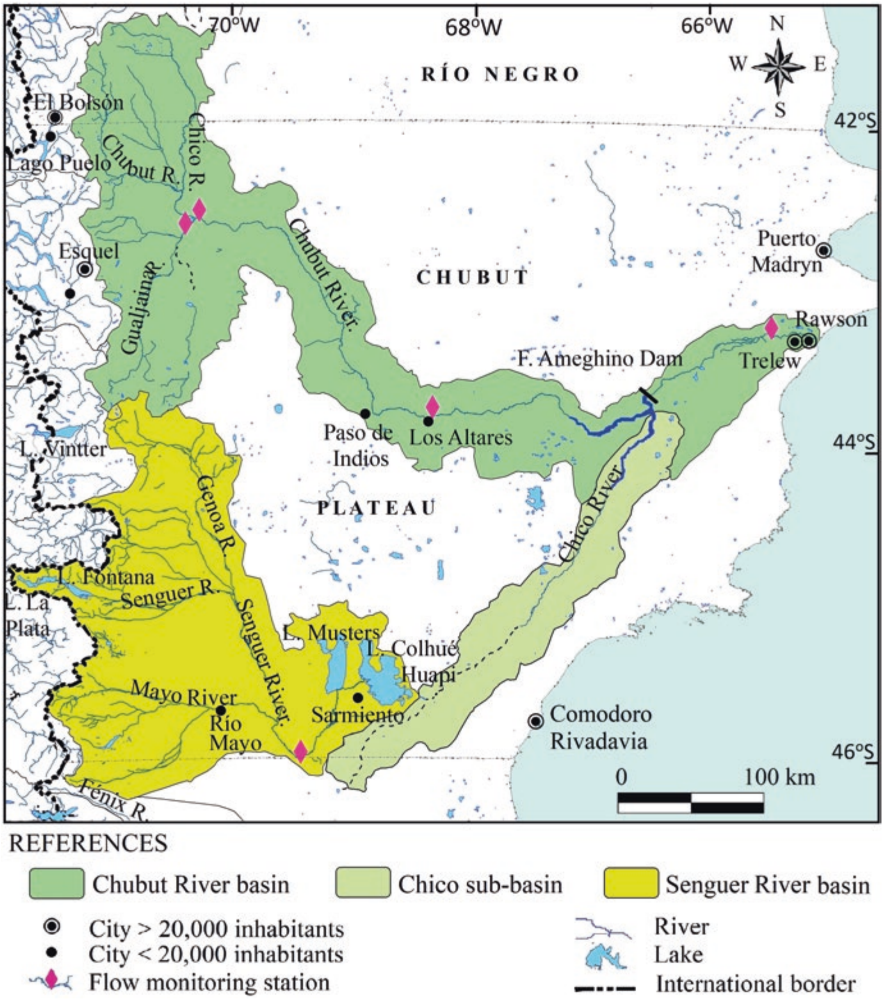

**Fig. 9.4** Chubut, Senguer, and Chico river basins

From the province of Río Negro, the Chubut River runs southward to join the Chico River. Downstream, the next tributary is the Gualjaina River that runs from the south, discharging 13 $m^3 s^{-1}$ on average. Its sub-basin extends over 5800 $km^2$. The Chubut River meanders on the valley, which by sectors is 2 km wide. Sudden floods fully inundate it, such as on July 1, 2004, when the peak flow reached 802 $m^3 s^{-1}$, with Gualjaina River accounting for 155 $m^3 s^{-1}$.

Eastward, the river valley is encased in the plateau. This upland is formed by a volcanic pyroclastic complex consisting of volcaniclastic, intrusive, pyroclastic, and effusive rock sequences, formed in the late Paleocene and early Eocene (Aragón and Mazzoni 1997). Dry creeks that descend the plateau connect the main valley, but they seldom have water.

Between 1889 and 1904, extraordinary floods disturbed the early years of the Welsh immigrants in the lower valley they colonized. The Florentino Ameghino dam reduced the historical flood risk that threatened the lower valley, which gradually was occupied by farms and the urban areas of Dolavon, Gaiman, Trelew, and Rawson cities.

The dam, inaugurated in 1968, is situated 130 km upstream of the river mouth. It inundates the encased Chubut River valley along 55 km, as well the Chico River Valley for 25 km. The dam provides hydropower, flood control, and water to supply the population and the irrigation system. The dam has fully controlled the maximum-recorded daily flow that reached 841 $m^3 s^{-1}$ on July 8, 2004. Downstream of the dam, the uniform flow contrasts with the variable hydrological regime in the rest of the basin, which, due to the lack of natural regulation mechanisms, has a faster response to winter rains, and currently shows a persistent low water period. In Los Altares (middle stream), the flow average declined from 49.3 $m^3s^{-1}$ (historical series before 2010) to 32 $m^3 s^{-1}$ in 2010–2019.

The drought was extreme in the April 2016–March 2017 hydrological year, when daily flows dropped to values never registered before (Table 9.2). Although the dam water reserve mitigated water scarcity, the drought harmed farm productivity in the lower valley due to the temporal suspension of water supply for irrigation as well as a drop in the hydropower generation that affected the urban areas. The dam, on the other hand, significantly reduced sediment transport. Furthermore, flood peak attenuation hindered transport capacity downstream of the dam, resulting in contraction of the riverbed, which shows narrowing and bottom elevation due to sediment contribution from alluvial basins (Kaless 2015).

The river ends in a meso-tidal estuary that was flooded by the last two marine transgressions. The Playa Union coastal barrier grew from north to south, leading to the migration of the Chubut River outlet. Waves and tides dominate the current estuarine dynamics. Tidal effects attenuate in less than 5 km (Isla et al. 2015).

Most of the population of the basin is located in the lower valley. Trelew and Rawson, as well as several smaller towns, utilize water for human consumption. The Chubut River is the only source of water supply for 200,000 people. Although

**Table 9.2** Chubut River flow ($m^3 s^{-1}$) for the extreme drought during the 2016–2017 hydrological year, upstream and downstream of Ameghino dam

| Chubut River Section | Annual mean flow | Max daily flow | Minimum daily flow | SD | Critical flow periods |
|---|---|---|---|---|---|
| Los Altares (upstream of the dam) | 17.1 | 70 | 3.1 (Mar 3, 2017) | 15.7 | 2 months below 8 $m^3 s^{-1}$(Apr–May 2016)<br>3 months below 8 $m^3 s^{-1}$ (Jan–Mar 2017) |
| Valle Inferior (downstream of the dam) | 17.1 | 32 | 8.4 (Oct 31, 2016 | 5.4 | 2 months below 10 $m^3 s^{-1}$during spring 2016 (preventive management) |

*SD* daily flow standard deviation

Puerto Madryn is not part of its hydrographic basin, it gets permanent flows from the Chubut River to supply its inhabitants. Mining and oil companies use underground water in their operations.

Three zones are distinguished in the basin, according to their characteristics and activities: Upper, Middle, and Lower Valley. The Upper Valley of the Chubut River (VARCH) has a rich water potential. In this zone, 99% of the irrigated land in the Chubut River basin corresponds to planted pastures. The Middle Valley of the Chubut River (VAMERCH) contains about 70 cattle companies with river frontage where water is used for gravitational irrigation of pastures. The strength of the wind, along with the existence of loose soils, causes substantial wind erosion and limitations for crops in this region. The Lower Valley of the Chubut River (VIRCH) is the agricultural zone with the highest predominance in the basin, accounting for about half of the province's present agricultural activity on around 20,000 hectares. The agricultural-livestock sector is the largest consumer of water in the basin (Pessacg et al. 2020). The Chubut River provides irrigation to the whole VIRCH through a channel network whose scope extends to multiple towns (Consejo Federal de Inversiones 2013).

The Florentino Ameghino dam allows the irrigation of 22,000 hectares that are destined for agricultural production and has produced energy since 1969. The current low capacity of the Chubut River limits the efficiency of the dam (Sainz-Trapaga 2018). After the dam, there are two main irrigation channels, which have favored the settlement of the population, contributing to the development of the towns in this area. Some drainage problems have been generated by the change of the river's natural conditions. Thus, there are limitations for some types of crops in the valley due to sodium or salinity in soils (Consejo Federal de Inversiones 2013).

The environmental degradation in this basin is shown by advanced desertification in several areas, most likely due to overgrazing induced by extensive livestock rearing, mostly sheep (CIEFAP 2021).

### Senguer River Basin

The Senguer River basin extends 37,800 km$^2$ in southwestern Chubut and northern Santa Cruz. The outlet of La Plata-Fontana lakes is the main source of the river. Both lakes are located westernmost of the basin, in a narrow and humid valley that penetrates sharply into the western forest ecoregion, flanked by snow-covered summits that include some small glaciers. The river runs eastward across the ecotonal landscape to meet the Genoa River that flows from the northwest across the steppe, with headwaters in the eastern side of the front moraines deposited by the Palena-Vintter paleoglacier. The Senguer River flows south, merging with Mayo River, its last tributary, and turns northeast to end in Bajo de Sarmiento, where the Lakes Musters and Colhué Huapi (LCH) are situated. Scordo et al. (2020) report spatial and temporal TDS and nutrients variability. Concentrations increase downstream; higher values match rain and snowmelt periods in winter and spring. The ultraoligotrophic character of Lake Fontana differs from the mesotrophic-oligotrophic state of Lake Musters. TDS increases along the basin from 14 to 278 mg $L^{-1}$, but water quality parameters are within the limits for human consumption after a normal treatment (Scordo et al. 2020).

The town of Sarmiento is located just south of both lakes in Bajo de Sarmiento, a dry sub-basin that covers 5900 $km^2$ but seldom supplies runoff. The lakes are then recharged by the Senguers outflow, which is fed by the basin upstream from Bajo de Sarmiento, which covers 31,900 $km^2$. A vast channel network that distributes the Senguer's flow has transformed Sarmiento into a fruitful oasis, but this area has a long history of unusual hydrological shifts. Until the late Pleistocene, the old Senguer would have been directly connected to the Chico River, but its direction changed to Bajo de Sarmiento, likely due to backward fluvial erosion. Before this significant shift, in the late-glacial period, the water level reached 60 m above the current Lake Musters level, forming a 4300 $km^2$ paleolake that merged both lakes (Gonzalez Díaz and Di Tomaso 2014), reaching four times the current lake area. At that time, the Genoa River would have been the pathway for the Palena-Vintter paleoglacier discharge (Coronato et al. 2004), and then the Senguer flow would have been at less four times larger than it is now, to be able to compensate for evaporation losses from the large lake area. Before the Holocene, the water level dropped, resulting in the separation of these lakes.

Gonzalez Díaz and Di Tomaso (2014) attribute this level drop to the onset of arid conditions; however, we believe it was mainly caused by fluvial erosion over the LCH-Chico spillway, followed by the release of the Lake Vintter drainage's waterway to the Palena basin through the Carrenleufú River after the ice receded.

Satellite images show the alluvial fan formed by the Senguer's discharge into Bajo de Sarmiento that fed the Lake Musters on the left and the LCH on the right. Fan morphology suggests that more than half of the flow was originally discharged into LCH, letting excesses drain into the Chico River, but this natural distribution was altered. The LCH-Chico connection ceased in the 1920s (Tejedo 2004), resulting in the current endorheic state.

In 2010–2019, the Senguer average flow into Bajo de Sarmiento was 37.4 $m^3\ s^{-1}$, which only represents 70% of the historical average (Table 9.1). Although this input seems to be significant, it is the necessary flow to balance the evaporation from the lake's surface, considering the basin's endorheic nature. On the other hand, in recent years, the incoming Senguer's flow to Bajo de Sarmiento evidenced a marked summer decline, showing minimum monthly values in March–April, ranging between 5 and 7 $m^3\ s^{-1}$.

A drugged channel leads the Senguer River directly into Lake Musters. Just before its outlet, it bifurcates to feed the stream called Falso Senguer that drains toward LCH. The Lake Musters recharge is prioritized because it is the water supply for the local population and the aqueduct that supplies other cities (Scordo et al. 2017). These consumptive water uses magnify the LCH water unbalance, resulting in cumulative storage losses, which have a substantial impact on the extension of the shallow Lake Colhué Huapi (LCH).

Recreational salmonid fishing is the main attraction in the upper basin, which is sparsely populated. Along the river valley, water is used for flooding lands and irrigation of *mallines* that serve as cattle feed. In Bajo de Sarmiento, which is an area of intensive livestock and agricultural production, part of the river discharge is diverted to irrigation canals before the mouth in Lake Musters (Tejedo 2004). Water is also extracted in the El Codo del Senguer area for oil companies that are located in the northwestern zone of Santa Cruz (Scordo et al. 2017). Water is extracted from Lake Musters to supply ca. 350,000 inhabitants of the major regional urban centers, as well as residents, and for oil companies located in the southeast of Chubut.

Approximately 80% of the total water extracted from the basin's rivers and lakes is destined for flood irrigation for agricultural-livestock purposes in Bajo de Sarmiento.

Natural fluctuations and consumptive water uses are causing drastic changes in both water bodies (Scordo et al. 2017). LCH fluctuation caused health problems to the local population as well as residents of the coastal cities of Comodoro Rivadavia and Rada Tilly because the material of the lake substrate is transported by the wind (Tejedo 2004). Commercial fishermen from LCH had to relocate to Lake Musters. This complex situation has generated conflicts among the productive sector of Bajo de Sarmiento and those who wish to prevent the disappearance of LCH (Scordo et al. 2017).

### Chico River Basin

The Chico riverbed extends through an arid 300-km-long valley that remains dry most of the time, except for the lower section, flooded by the influence of the Ameghino Dam. The entire basin extends in the steppe ecoregion, presenting 62% of bare soils (Kaless et al. 2019).

Transverse dry ravines have dissected the plateau, ending in the main valley, where minor alluvial fans were formed by sediment deposition from sporadic flow events. However, occasional heavy rains from the Atlantic Ocean can generate floods with a high sediment load. Extraordinary rainfalls amounting to 330 mm were registered in the nearby city of Comodoro Rivadavia between March 29 and April 4, 2017, with a maximum intensity of 232 mm in 24 hours. The storm reached the Chico River basin, causing a 667 $m^3 s^{-1}$ peak flow (Kaless et al. 2019) that raised the dam water level by 12 m and discharged an extraordinary sediment load, interrupting the water supply for the population of the lower Chubut valley for several days.

## *4.4 Puelo River Basin*

The Puelo is a transboundary river, whose upper basin is partially located in Argentina, in the Patagonian forest ecoregion, involving the SW of the province of Río Negro and NW of Chubut. In Chile, the basin encompasses part of Los Lagos Region, wherein rainforest prevails. The total basin area is 9000 $km^2$ (Fig. 9.5).

**Fig. 9.5** Puelo and Yelcho river basins (left). Palena, Cisnes, and Aysén river basins (right)

The mountain altitude, around 1600 m, allows humidity to enter the eastern upper basin, which comprises the sub-basins of Manso River in the North and Lake Puelo in the South. The Manso River originates in the proglacial Lake Ventisquero Negro, which has expanded in the last 30 years due to the retreat of the glacial tongue that descends from Mount Tronador (3478 m a.s.l.). Before emptying into Lake Mascardi, the upper Manso River runs through a 13 km valley, partly covered by peatlands. The drainage system extends west and south, linking with other lakes. The river enters Chile through Paso El León (420 m a.s.l.).

In the southern upstream system, Lake Puelo has three main tributaries. From the north, the Azul River drains the mountains that enclose a humid and inhabited valley to end in Lake Puelo, after crossing the town of Lago Puelo. The Epuyén River begins in the mouth of Lake Epuyén and discharges in the Puelo's eastern shore. From the South, the Turbio River, which originates in the Río Turbio Provincial Park, supplies the lake with the largest flow (Table 9.1), fed by meltwater from Cordón Cholila.

Winter floods occur in the Manso River headwaters, whereas Lake Puelo outlet floods occur in November–December, indicating the considerable influence of the Turbio river basin seasonal snow, as well as the regulation that Lake Puelo supplies.

The Puelo River tributaries provide water to the valleys of El Bolsón, Puelo, Golondrinas, El Hoyo, and Epuyén, wherein 600 hectares of fertile soils cultivated with hops, vegetables, fruit trees, and pastures are irrigated (Bava 2021). There is a growing conflict due to the urbanization and subdivision of land through which the irrigation canals pass, which makes almost unfeasible to continue their use (SPTIP 2017).

More than 70% of the basin total surface is covered by protected areas. The Lago Puelo National Park extends 27,700 ha. There are also provincial protected areas, such as parks and forest reserves. The Manso River sub-basin is partially under the jurisdiction of the Nahuel Huapi National Park, and a World Biosphere Reserve has been established near the international border. Tourism associated with the beauty of the rivers and lakes in the area has grown in importance during the last decade. The Puelo River is well-known for its aptitude to support sport fishing for trout and salmon, as well as other sports such as kayaking and rafting.

Water from the Azul River is used for productive activities around the cities of Lago Puelo and El Bolsón, whose strong growth owes to the booming tourism in this area. The floodplain of the Azul River was occupied by Lago Puelo town, and consequently urbanized and cultivated areas have suffered repeated flooding (Herrera et al. 2007). In the El Bolsón area, the Quemquemtreu River, a tributary of Azul, runs through the city and its banks, and water irrigation areas have been irregularly occupied by precarious homes exposed to recurrent floods (Nini 2016). Although the quality of the Azul water has been preserved so far, there is evidence of alterations due to the great increase in anthropic activities. As a result of these circumstances, as well as extreme events that occurred in the basin, the provinces of Chubut and Río Negro established the Río Azul Basin Authority (ACRA) in 1997, which promotes studies and works related to flood prevention and pollution.

In Chile, the Puelo River begins at the mouth of Lake Puelo, its main upper tributary being the Ventisquero River. The Manso and Puelo Rivers join in the lower basin, 6 km upstream of Lake Tagua-Tagua. In the lower section, the flow regime is driven by winter rains which cause flood peaks in June–July. March is the month with the lowest flow, but flow seasonality is moderate since the supply water regulation. In the Argentine side of the basin, the main lakes are Guillermo, Mascardi, Los Moscos, Fonk, Linco, Roca, Christie, Manzanos, Martín, Steffen, Escondido, Epuyén, and Esperanza, while in Chile, major lakes are Carmen Vidal, Puelo, Interior, Las Rocas, Azul, and Tagua-Tagua.

The Puelo River supplies the main freshwater flow to the Reloncaví Fjord. The Petrohue River discharges 280 $m^3 s^{-1}$ in the northern head of the fjord as well as Cochamó and Blanco, which are minor rivers. Fluvial discharges result in a transitional marine ecosystem that has a 5-m-deep low-salinity upper layer. The moderate intertidal regime results in a tidal front advance speed of 10 cm $s^{-1}$ that does not disturb this stratification (Valle-Levinson et al. 2007).

The population is scarce and represents the largest user of water for drinking, domestic use, and sanitation. Fish farming is another important water use that takes place in coastal areas and some inland lakes, accounting for nearly 25% of the surface consumptive rights delivered. There are nonconsumptive rights granted for hydroelectric generation (95%) and industries. Many tourism activities are developed in aquatic environments (Universidad de Chile 2016). The local community, through different associations, requested the declaration of a water reserve for the conservation and local basin development, considering the high amount of the flow granted for hydropower (Corporación Puelo Patagonia 2019).

The areas under Official Protection belonging to the National System of Wild Areas Protected by the State correspond, in a small proportion, to the Hornopirén National Park. Other areas of interest for conservation are part of the Pumalín Park and the ecosystem associated with the Puelo River, a priority site for conservation (Universidad de Chile 2016).

## 4.5 Futaleufú-Yelcho River Basin

This is a transboundary basin that extends in the Province of Chubut in Argentina as well as in the Los Lagos Region in Chile, over a total area of 11,500 $km^2$ (Fig. 9.5).

In Argentina, the upper Futaleufú River basin is bordered to the north by the Cordón Cholila, which is a water divide with the Puelo River basin. Small glaciers and seasonal snow feed Cholila Lake, the first one of the lake chain that regulates the upper basin runoff. Further south, the Cordillera de las Pirámides stands out, which includes small glaciers. The Futaleufú Hydroelectric Complex completes the upper basin flow control. The Futaleufú dam has unified four preexisting lakes, forming the 50-km-long lake Amutui Quimey that releases a 300 $m^3$ $s^{-1}$ average flow. The drainage regime is pluvio-nival, with glacier contribution and winter floods.

After the dam, the river runs down to reach the NE-SW-oriented Futaleufú valley and meets the Percey River, which runs from the north through the forest-steppe ecotone. The Esquel River is the main tributary of the Percey, whose abundant flow of around 3 $m^3$ $s^{-1}$ has been the basis for the valley's development. Winter rains produce recurrent floods. The urban growth of Esquel City occupied lowlands at the Esquel River edges, and this resulted in the flooding and damaging of homes located in marginal areas of the floodplain in the years 1992 and 1998 (Ferrary 2008).

The basin is located in the Los Alerces National Park, which contains one of the richest formations of Andean-Patagonian forests (CIEFAP 2018). The most populous city is Esquel, followed by Trevelin and Cholila, where water is used for human consumption as well as the main economic activities, which include extensive cattle ranching, tourism, and forestry (CIEFAP 2018). The availability and access to water are decisive for agriculture, which is limited to horticultural and fruit crops. At present, around 800 productive hectares are irrigated in the “Valley 16 de Octubre” system (SPTIP 2017).

There is evidence of environmental degradation of watercourses in the area encompassing the largest localities, caused by fires, cattle grazing, extraction of vegetation cover, and urbanization, through modification of the natural channels of rivers and streams, erosion of banks, high sediment content, and biodiversity loss. Overgrazing has also advanced desertification, and important areas, even virgin forests, have been eroded (CIEFAP 2018). Although the water is suitable for human consumption (Kusch and Pizzolon 2021), mining developments in the vicinity of Esquel city may be considered as a threat to water quality (Yannopoulos 2012).

In Chile, the Futaleufú River joins the rivers Espolón, Malito, and further smaller streams. Before the Malito River confluence, the Futaleufú flow average was 428 $m^3\ s^{-1}$ (2002–2020). The maximum monthly flow is 547 $m^3\ s^{-1}$ in July, and the minimum is 250 $m^3\ s^{-1}$ in March. There is no flow data available downstream of this section, but the final flow can be estimated between 550 and 600 $m^3\ s^{-1}$. Lake Yelcho, whose outlet gives place to the Yelcho River, intersects the path of the Futaleufú River along 33 km.

Lake Yelcho (116 $km^2$) has oligotrophic to mesotrophic characteristics (Soto 2002). At the lake outlet, Yelcho River water presents very low salinity (EC: 41 $\mu S\ cm^{-1}$) and neutral pH (DGA available data).

In the low stream, Yelcho River receives the Michimahuida and Amarillo Rivers. These tributaries rise northward, fed by glaciers and snow from the Michimahuida volcano (2404 m a.s.l.). Landslides are frequent in the Amarillo valley, due to heavy rains that destabilize steep slopes. The Chaitén volcano is located only 17 km westward of the Michimahuida. Although the volcano is situated outside the basin, ashfalls from the 2008 eruption that destroyed the city of Chaitén (Pierson et al. 2013) caused severe impacts in the Amarillo river basin, such as forest destruction and changes in the riverbed morphology and sediment transport. The Yelcho River ends into the Reloncavi Fjord, 5 km southward of Chaitén town, forming a wide delta.

Futaleufú and Puerto Cárdenas are the only localities in the basin. The Futaleufú National Reserve, situated next to the town of Futaleufú, includes a section of this river as well as its tributary, the Espolón River. In the Espolón River, in April 2010, the first bloom in South America of *Didymosphenia geminata* (“didymo”) was observed, which soon replicated in the Argentine section of the Futaleufú River (Sastre et al. 2010; see also Chap. 12).

The main use of water is for population supply, which is mostly rural. In Lake Yelcho, some water rights for fish farming have been granted. The basin’s hydroelectric potential is reflected in the large number of rights granted for this purpose. A run-of-river hydropower plant (no storage facility) produces 1400 kW on the Azul River, a tributary of the Futaleufú. Tourism activities in the area are inextricably linked to its water resources. The Futaleufú River has been named one of the top three destinations in the world for the practice of sports such as fly fishing, rafting, and kayaking. Given the significance of these activities, the regional government determined that it is crucial to establish methodologies and calculations of tourist reserve flows (Universidad de Chile 2016).

## *4.6 Carrenleufú-Palena River Basin*

The Palena River basin (Fig. 9.5) extends in the Chubut province of Argentina and the Chilean regions of Los Lagos and Aysén, covering a total area of 13,300 $km^2$. The Lake Palena-Vintter collects streamflow from a large mountainous massif covered by cirque glaciers drained by the Carrenleufú River. This river runs eastward through the forest-steppe ecotone, drawing a wide curve to head west. Near the town of Corcovado, the average flow is 49 $m^3\ s^{-1}$.

There is little population in the Argentine sector, where the economy is linked to livestock and forestry production, based on native forest and pine plantations and activity concerning fine woods. Lake Vintter and Corcovado areas have good natural potential for tourism; however, it is limited because of their remote location from the main circuits.

In Chile, the river takes the Palena name, running west. The regional tectonic structure determined a rectangular drainage pattern in the basin. Downstream of Palena town, the river joins the Frío River and heads southward by a straight valley to meet the Rosselot River.

The Rosselot River runs shortly between the Lake Rosselot outlet and the Palena River and is fed by a large and elongated basin, with headwaters 70 km eastward from the international border, which the Pico River drains in the Argentine steppe. The Pico's flow rapidly increases westward. Main tributaries, as Las Pampas and Encuentro, join in the 11-km-section previous the border. In Chile, the Pico River is named Figueroa and meets the Turbio River, which runs from the SW through a deep and humid valley 50 km long, fed from ice-snow-covered massifs, where forest covers the hillslopes and peatlands extend at the bottom valley. Lake Verde occupies the last valley section before the Turbio-Figueroa junction. Westward, the Figueroa River ends at Lake Rosselot. This lake also collects a significant flow from snow and glaciers of the southern Queulat Mount.

After the Rosselot confluence, the Palena average flow was 821 $m^3\ s^{-1}$ (2000–2019), showing a yearly declining trend of −8.5 $m^3\ s^{-1}$. The average flow, which was 874 $m^3\ s^{-1}$ from 2000 to 2009, fell to 771 $m^3\ s^{-1}$ in the next decade. Risopatrón, Correntoso, and Dinamarca rivers are the last significant tributaries before the discharge of the Palena River into the Corcovado Gulf.

Landslides are recurrent events in western valleys of the basin when rainfalls destabilize volcanic soil layers on steep hillsides. In December 2017, a 124-mm rainfall triggered a landslide in Burrito Stream's headwater, a tributary of the Frío River, which impacted a glacier terminal, causing a mud-ice flood that incorporated large woody debris. The torrent destroyed 50% of the Santa Lucía village, wherein the thickness of the fan deposit was estimated at 2 m on average over 90 ha (Duhart et al. 2019).

The river and lakes constitute a tourist destination due to their pristine nature and their aptitude for salmonid fishing. Currently, few water use rights have been granted in the basin, but its abundant hydroelectric resources are under intense pressure due to a pending hydroelectric project application (DGA 2015).

## 4.7 Cisnes River Basin

The Cisnes River basin, located between the ones of Palena and Aysén, expands 5400 $km^2$ in the region of Aysén, along a single valley, which is a direct corridor where the Cisnes River runs along 160 km, from the steppe to the Puyuhuapi Fjord. Morainic deposits materialize the border with the Senguer basin, permitting transit to Argentina without exceeding 915 meters in altitude. These conditions allowed ancient settlers to use the Cisnes upper basin as a marginal territory and later as a gateway to the forest, ca. 2800 years BP (Méndez and Reyes 2008).

The upper basin of Cisnes is covered by steppe and ecotone ecosystems. The river flow grows downstream as humidity increases westward, supplied by short and torrential tributaries coming down from the valley's outlying mountains. The evergreen rainforest dominates the central and lower basin, where swamp-forest wetlands are present in poorly drained areas (Ardiles et al. 2020). Precipitation is 2950 mm $year^{-1}$ in Puerto Cisnes, where the river average flow is 215 $m^3\ s^{-1}$. Waters are of very low salinity; EC ranges between 11 and 70 $\mu S\ cm^{-1}$. Cu and Al (of natural origin) show relatively high rates; their respective ranges are 10–40 $\mu g\ L^{-1}$ and 0.1–0.4 mg $L^{-1}$ (DGA 2004a).

The basin's limited population is concentrated in Puerto Cisnes (ca. 5000 inhabitants). The economic activity on the coast is related to salmon farming. Tourism services are growing in the lower basin, while livestock is important in the upper and middle basin. Rivers and lakes have good conditions for recreational fishing. As well as other rivers in the region, the Cisnes has a significant hydropower potential, which has originated numerous demands for water rights from private companies (DGA 2007).

Protected areas in the basin are the Queulat National Park, the Lake Carlota National Reserve, and the Lake Las Torres National Reserve.

## 4.8 Aysén River Basin

The Aysén River basin (Fig. 9.5) extends over 12,000 $km^2$, mainly in Chile. The Simpson River, one of its main tributaries, has sources in northern Santa Cruz, in Argentina, and in the Rojo hill (46° 06′ S, 71° 45′ W). The local tectonics, shaped by intense glacial activity, resulted in a rich drainage network, whose hydrology responds to a rain-snow regime, complemented by glacier flow. The high precipitation results in water abundance in rivers and lakes.

The Aysen, Mañihuales, and Simpson River valleys penetrate deeply at low altitudes, between massifs of 1500 m elevation. Despite the open valleys and the moderate altitude of the mountains, the negative eastward precipitation gradient is significant: 2940 mm in Puerto Aysén, 1385 mm in Coyhaique, and 621 mm in Balmaceda (DGA 2004b).

The Simpson and Coyhaique rivers merge close to the city of Coyhaique, the capital of the Aysén Region, and the largest city in Chile between Puerto Montt and

Punta Arenas. The Simpson average flow is 43.3 $m^3 s^{-1}$, from which 6.0 $m^3 s^{-1}$ is supplied by the Coyhaique River. The maximum monthly flow occurs in October and the minimum in February.

In the lower basin, the Simpson merges the Mañihuales River, which runs from the northern side of the basin, supplying 169 $m^3 s^{-1}$ average flow. This junction forms the Aysén River, which after 25 km, discharges into the Aysén fjord, near Puerto Chacabuco. The river average flow is 530 $m^3 s^{-1}$, showing a declining trend over the last decade (Table 9.1). Aysén floods have caused damage in Puerto Chacabuco, as those occurred in May 1966 and June 2010 wreaked homes and roads. Alluviums are frequent in much of the basin, due to intense rains on the rough relief.

The towns of Coyhaique, Puerto Aysén, and other smaller ones have the highest demand for water supply. Rivers and lakes are used to generate hydroelectric energy for the major cities, and recreational fishing is practiced in the lakes and several rivers. There are water demands for industrial use, mainly for fishery industries, and water rights granted to mining companies (DGA 2004b).

Hydropower is generated on a small scale from tributaries. Lago Atravesado is the largest plant (11 MW) followed by some run-of-river hydroelectric plants such as Monreal, Puerto Aysén, and El Toqui, each one with an installed capacity between 1.5 and 3 MW.

The basin's water quality is generally very good for the current uses. However, certain natural and human-induced phenomena, such as salmon industry effluents, rainfall-induced leaching of mining waste deposits, forest fires, mine water drainage, and mine tailings, have altered the quality of the waters (DGA 2004b).

It should be noted that these areas were affected by the largest forest fires in the history of colonization. The settlers burned forests to obtain land suitable for agricultural activities, but the situation got out of control, and between 1920 and 1940, about two million hectares burned in the Region of Aysén, mainly in this basin, with the consequent modification of the natural environment (Bizama et al. 2011). Currently, many areas are under official protection as part of the National System of Protected Areas by the State (SNAPE), to preserve the region's biodiversity and scenic beauty. The total area occupied by these sectors is equivalent to 12% of the basin.

### 4.9 Baker River Basin

The Baker is the major river in Chile. Its basin has an area of 26,500 $km^2$, part of which is located in the extra-Andean Argentine territory (Fig. 9.6). It is the outlet of two vast systems: the eastern slope of the Northern Patagonian Icefield (NPI) and the Lake Buenos Aires-General Carrera (BA-GC), the second largest lake in South America (1850 $km^2$). Other great lakes in the basin are the Pueyrredón-Cochrane (P-C), Brown, and Tranquilo. The flow regime is nivo-pluvial with great glacial contribution and effective lake regulation.

During the last glaciation, the Patagonian Icefields expanded east, draining toward the Atlantic shore. The warming in the late Pleistocene resulted in ice front retreating and proglacial lakes expansion over ice-released areas. The levels of BA-GC and P-C lakes reached 400 and 500 m a.s.l., respectively, which means 200 and 350 m higher than today.

Ice receding resulted in drainage reversal to the Pacific slope, causing heavy hydrographic shifts and huge environmental changes, in several phases. At least six drainage events occurred since 15 k year BP, with the aperture of the Barrancos spillway (420 m a.s.l.) in the west of P-C paleolake. Later, the Soler Glacier retreated, releasing the Bertrand col, which resulted in the paleolake Chelenko[2] development by merging BA-GC and P-C lakes. Blocked on the west by the NPI, Chelenko lake level increased, causing drainage northward, through the Bayo Valley (Exploradores River), northwest of the basin. Between 12.6 and 11.7 k year BP, further ice receding in the lower Baker valley resulted in strong hydrographical shifts: the current westward waterway opening, Chelenko paleolake lowering, Bayo outflow abandonment, and paleolake fragmentation (Thorndycraft et al. 2019). The impacts of these devastating events transformed landscapes and ecosystems. Morainic sequences at the BA-GC eastern head at upper levels than those of the last glaciation, evidence the alternation in the general drainage toward the Atlantic Ocean in glacial times and toward the Pacific during interglacial periods.

At the basin level, bioclimatic features are extremely heterogeneous, as well as the hydrology of the basin components. BA-GC's northern tributaries like the Murta and Ibañez rivers have springs in highlands wherein precipitation is abundant and glaciers cover mountain tops. The Jeinimeni River discharges from the hills of the National Reserve Jeinimeni into the southern lakeshore, running along the international border in its lower sector. The area of the Jeinimeni River basin is about 1300 km$^2$.

Short east of Jeinimeni River discharges Los Antiguos River, which drains water from the NW sector of the Lake Buenos Aires Plateau and steppe plains, without glacier contribution.

Los Antiguos and Jeinimeni Rivers feed a permanent irrigation network for agricultural activities in Los Antiguos town and Chile Chico, respectively, located in the southern shore of Lake BA-GC (Sánchez et al. 2016). The lake also supports fish farming and recreational trout fishing.

The Lake BA-GC presently discharges into Lake Bertrand (Chile), whose outlet is the origin of the Baker River. Lakes assimilate flood peaks of tributaries, resulting in a much-regulated river regime. The minimum monthly flow average occurs in September and the maximum in March. Baker's tributaries present a nivo-pluvial regime with a rapid response to rainfalls.

The Baker meets the Nef River, which drains the NPI Nef Glacier and heads westward until the Chacabuco River confluence, where it changes its direction southeastward joining the Cochrane River and Salto River, before joining the

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[2] Chelenko is the indigenous name of the BA-GC Lake.

Colonia River from the NPI, and the Ñadis River on the opposite margin. Sixteen km downstream of the Ñadis discharges the Ventisquero River, which is the last tributary from the NPI.

Great anomalies in maximum flows have occurred since 2008 due to the recurrent emptying of Lake Cachet 2 (47° 11′ S, 73° 16′ W), dammed by the Colonia glacier. This ice body was destabilized by climate change, allowing subglacial lake discharge. Glacial lake outburst floods (GLOFs) release sudden and torrential flows, producing severe morphological alterations downstream. Jacquet et al. (2017) identified 21 GLOFs events after 2008, which duplicated normal yearly floods at the Bajo Ñadis Station. GLOFs have changed flood dynamics and fluvial sediment processes in the middle and low streams. Ulloa et al. (2018) recognized 246 islands and 393 bars along the Baker main course. They also identified 170 wetlands of 46.8 ha on average and described them as permanently or seasonally submerged areas characterized by poorly drained soils with hydrophytic vegetation and lack of trees, increasing downstream both in number and extension. Due to the unavailability of high-resolution satellite images before 2008, comparing these results to the pre-GLOFs state is unfeasible; however, these data indicate strong fluvial dynamics with GLOF events acting as the primary driver.

The Baker flow average in Bajo Ñadis is 924 $m^3 s^{-1}$; no trends have been observed in this period. Monthly flows are highest in January–February, as a consequence of glacier melting. There is no available flow data for river sections downstream of Bajo Ñadis, but assuming for Ventisquero River, a similar flow rate to that of the Nef River (52 $m^3 s^{-1}$) and 24 $m^3 s^{-1}$ for tributary streams that join along the 50-km-section downstream of Ventisquero River, the total discharge of Baker River is estimated to be 1000 $m^3 s^{-1}$. The river mouth forms a delta with islands, mostly covered by peatlands. The town of Tortel sits south of the river margin, on the coast of a long and branched Pacific fjord, between the NPI and SPI.

The main Chilean localities are Puerto Ingeniero Ibáñez, Puerto Murta, Cochrane, Caleta Tortel, and Chile Chico, which is connected to Los Antiguos town (Argentina) via a route that borders Lake BA-GC. The basin is a primary source of water for the development of local economic activities such as mining and horticulture in the Chile Chico sector, which use water of the Jeinimeni River for irrigation of cherry plantations and other crops (Álvarez Hernández 2015). Intensive livestock and agriculture are developed in the lake coastal areas.

Water is of great importance for tourism activities in the whole basin, based on the scenic beauty of the different water environments: the Baker River, lakes, glaciers, and wetlands. The preferred uses of the river are for rafting, kayaking, fishing, and, to a lesser extent, bathing. The General Carrera Lake encompasses a Nature Sanctuary known as “Marble Cathedral.” It also has a very good aptitude for recreational fishing and is an important connecting route between cities and towns.

The water quality in almost the entire basin is excellent. There are two protected areas near Chile Chico and Cochrane: San Rafael National Park and Patagonia National Park. The regional Biodiversity Strategy has prioritized conservation in various basin sectors (SEREMI MMA 2018). Furthermore, the basin is part of the temperate rainforest eco-region that extends from the Maule region, which has been designated internationally as one of the 25 most important sites for biodiversity conservation worldwide (Rovira et al. 2008).

## 4.10 Pascua River Basin

The Pascua River basin (Fig. 9.6) extends 14,200 km$^2$, both in the province of Santa Cruz (Argentina) and the Aysén Region (Chile). The Pascua River is the outlet of Lake San Martín-O'Higgins, which is the fifth deepest lake in the world, with 836 m maximum depth. The lake collects most of the flow in the basin coming from the Southern Patagonian Icefield (SPI), notably through the western and southern lake branches, which receive direct contributions from glaciers or through proglacial lakes.

Large glaciers, like Chico and O'Higgins, end at the Southwestern Arm. During the 1945–1980 period, the O'Higgins glacier front experienced an amazing 11.5 km retreat in the lake arm, which changed the local hydrography and triggered

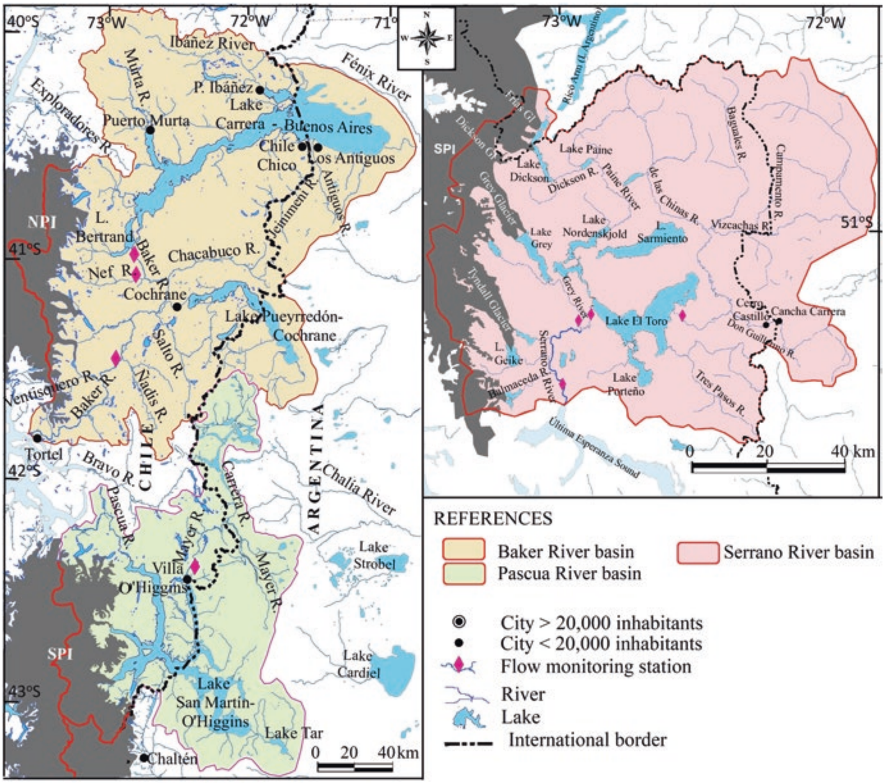

**Fig. 9.6** Baker and Pascua river basins (left). Serrano River basin (right)

geomorphological processes, like slope destabilization and gully's progress in the lateral moraines on the arm margins, as well as the emptying of an adjacent lake, dammed by the Gaea glacier (Iturraspe and Strelin 2002). Similar processes replicate in southern Patagonia as the result of the glacier recession.

The Mayer River is the major eastern tributary of Lake San Martín-O'Higgins. It integrates a complex water system, mainly extended in Santa Cruz, Argentina, which is nourished from the south by streams draining the southern slopes of Sierra de Sangra, and from the north by the lake complex composed by Volcán, Belgrano, Azara, Nansen, and further minor lakes. These water bodies collect the southern slope drainage of San Lorenzo Mount (47° 35′ S, 72° 18′ W, 3706 m a.s.l.). Both systems meet near the international border in a 1500 m wide glaciofluvial plain, forming a net of braided and parallel channels.

In Argentina, the basin is sparsely populated and far from the large urban and productive centers. The Perito Moreno National Park protects large areas of the basin that include glacial lakes and shallow lakes.

In Chile, the Mayer River flows southward, through a narrow encased valley. The lower basin presents open valleys covered by forest and peatlands. A chained lake system drains to the Mayer River before it ends in the Lake San Martín-O'Higgins, just along Villa O'Higgins.

The Pascua River originates at the outlet of Lake O'Higgins, in the Northwestern Arm. In the final river section, Borquez and Bergues rivers join, draining proglacial lakes from Pascua and Lucia SPI glaciers, respectively. In the northern margin, the Pascua collects the outflow of Lake Quetru, fed by glaciers and snowmelt of the mountains that divide water with the Baker Basin. From O'Higgins Lake, the Pascua River follows a curving route along 54 km to discharge into the Calén Fjord, which connects to the Penas Gulf.

The Mayer River flow average is 115 $m^3 s^{-1}$. The richest flow period is from November to March, with 182 $m^3 s^{-1}$ in January as the highest monthly average. The Pascua River has an average flow of 715 $m^3 s^{-1}$, with the highest monthly flow in February–March (1065 $m^3 s^{-1}$) and the lowest in September (415 $m^3 s^{-1}$). No trend has been identified for these flow amounts, which do not include the Berguer River's flow.

This basin is one of the most sparsely inhabited and difficult to access in the region. Part of the basin belongs to the Katalalixar National Reserve, recognized as a "World Biosphere Reserve" by UNESCO (Álvarez Hernández 2015).

Average values (2016–2020) from Pascua river water (DGA data) indicate a slightly alkaline condition (pH 7.7), very low salinity (EC 46 $\mu S\ cm^{-1}$), and significant iron presence (0.7 $mg\ L^{-1}$ on average). The Pascua river is perhaps the most pristine in Chile, and its waters qualify in the category of exceptional class (CONAMA 2011). It has great potential for special interest tourism, due to its geomorphological characteristics that make it one of the most challenging rivers for kayaking in the world (Álvarez Hernández 2015). On the other hand, the significant river flow and the fluvial morphology determine a great hydroelectric potential (Centro de Cambio Global UC 2015). However, the instrument on Regional Strategy states that the environmental vocation collides with initiatives for large hydroelectric projects (Gobierno de Aysén 2009), which have also been refused by social organizations and local communities because of the environmental threats they involve. Finally, the projects for great dams in the Pascua and Baker River basins were rejected.

## *4.11 Serrano River Basin*

The Serrano River basin extends 8850 km$^2$ both in the province of Santa Cruz (Argentina) and the province of Última Esperanza, Magallanes Region of Chile (Fig. 9.6). It involves contrasting landscapes including glaciers, lakes, forested valleys, grasslands, and steppe. Rivers drain southern SPI terminals, alternating their way with lakes of glacial origin.

The main glaciers of the SPI in the basin are from N to S, Frías, Dickson, Gray, Tyndall, and Balmaceda. All of them show recessive behavior, holding proglacial lakes that increase in size due to their ice front retreat.

Until 20 years ago, the Frías and Dickson glacier terminus blocked the transverse valley that connects Dickson and Argentino lakes, forcing northward the Frías glacier drainage to the Lake Argentino basin (Iturraspe and Sottini 2005). A rapid glacier retreat released the valley of ice, allowing to change the Frías flow direction toward Dickson Lake, leading to the most recent flow reversal in Patagonia, from the Atlantic to the Pacific Ocean.

The Dickson River connects Dickson and Paine lakes, which the Paine River links to the Nordenskjold-Pehoé-Toro Lake system. The Lake Sarmiento is endorheic; hence, it is not connected to the drainage network. Lake Toro also collects drainage from the eastern basin, by the Chinas and Baguales Rivers. These courses have sources in the Baguales mountain range, which is the water divide with the Santa Cruz River basin. The Zanja Honda and Vizcacha rivers run from Argentina and discharge in the Baguales River, which in turn ends in the Las Chinas River. Don Guillermo River is the last cross-border tributary of Las Chinas before its discharge of 16 m$^3$ s$^{-1}$ into Lake Toro. From the south, this lake receives the contributions of Lake Porteño and Tres Pasos River.

The Serrano River begins at the outlet of Lake Toro. The important lakes in the basin (Table 9.4) moderate seasonal flow variability. The average flow of the Serrano River near the Toro lake outlet is 85 m$^3$ s$^{-1}$; the highest and lowest monthly flows are 117 m$^3$ s$^{-1}$ in March and 37.5 m$^3$ s$^{-1}$ in September, respectively.

The Grey River is the first Serrano's tributary, which flows from the outlet of the proglacial Lake Grey, discharging 123 m$^3$ s$^{-1}$ on average. Tyndall and Balmaceda Rivers drain the homonymous glaciers, being the subsequent tributaries from the west. The average discharge of the Serrano in the final section is 408 m$^3$ s$^{-1}$, being the highest monthly flow in February (781m$^3$s$^{-1}$) and the lowest in August (155 m$^3$ s$^{-1}$). The yearly flow of the Serrano River and its tributaries increased between 2010 and 2019 (Table 9.1).

Populations are scarce and dispersed in the Argentine sector. Water is used for the irrigation of horticultural crops and human consumption on a small rural scale in some farms and stalls, as well as recreational fishing. There are works to capture water for irrigation along the Vizcachas River. Water is also used for HC operations. All rivers on the Argentine side of Serrano basin present TDS concentrations below 300 mg $L^{-1}$ (Díaz et al. 2020).

In Chile, the small town of Cerro Castillo, the center of an extensive livestock zone, is located at the edge of the Don Guillermo River, which is the water source for human consumption as well as for grassland and crop irrigation (CENMA 2010).

Water is used for recreational activities as well as human consumption in all tourist facilities related to National Parks. Torres del Paine National Park is a set of turquoise blue lakes, lagoons, rivers, and snowdrifts that surround a massif which includes the Torres de Granito and Cuernos del Paine. The Bernardo O'Higgins National Park protects part of the SPI, which contains one of the world's largest freshwater reserves. In certain rivers, recreational fishing is being developed. The Serrano River is particularly well-known for fly fishing.

The water quality is very good in most of the basin's watercourses, and only in a few sectors, it shows characteristics associated with diffuse pollution from livestock activities, making it one of the least disturbed basins in the country (CENMA 2010). Although the population of the basin is scarce, the significant increase in tourism in Torres del Paine Park is generating environmental and social conflicts (Saelzer et al. 2019).

### *4.12 Deseado River Basin*

The Deseado River completely crosses the Argentine province of Santa Cruz, along approximately 500 km through arid and lonely territories, to end in the estuary wherein the city of Puerto Deseado is located (Fig. 9.7). The river basin has an extension of 27,600 $km^2$.

Before the Holocene, the Deseado was a mighty river fed by the BA-GC and Cochrane-Pueyrredón-Salitroso-Gío proglacial paleolakes. As described in Sect. 4.9, this drainage reversed to the Pacific slope through the Baker River, and Deseado's flow regime changed dramatically.

The current river sources lie in the hills of the Ap Iwan Mount (2300 m a.s.l.), north of Lake BA-GC. Small ice bodies and seasonal snow feed the Fénix River, which runs eastward along a mountain valley in the forest-steppe ecotone, to continue through intermorainic corridors to Perito Moreno town. There, it reaches the older Deseado riverbed in a setting wherein the interoceanic water divide is poorly defined.

In the nineteenth century, the Fénix River turned west shortly before the town, thus ending in Lake Buenos Aires, but in the year 1898, it was artificially redirected by the settlers toward the Deseado riverbed (Roberts and Garivati 2008). Some years later, a water intake was built 60 km upstream of Perito Moreno to irrigate

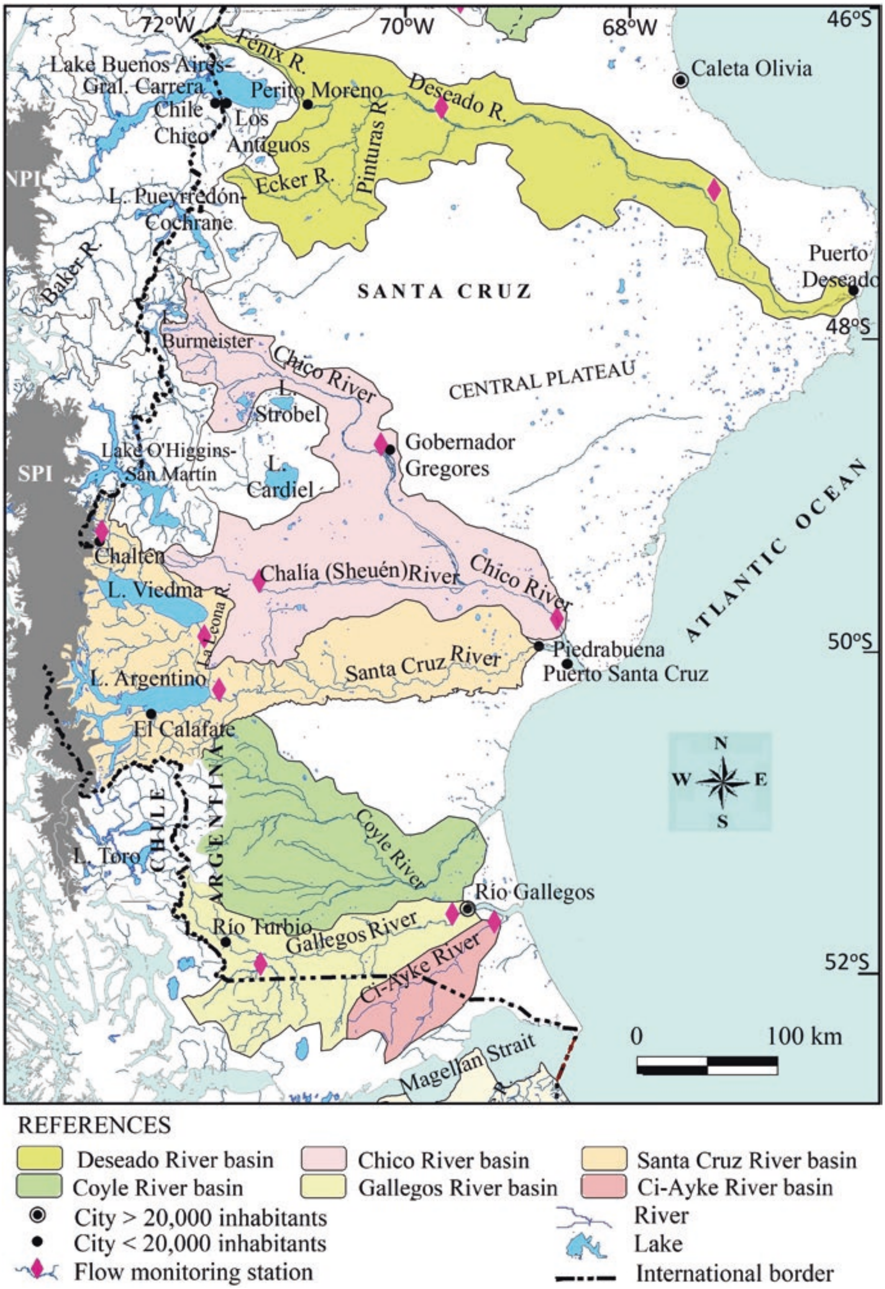

**Fig. 9.7** Deseado, Chico-Chalía, Santa Cruz, Coyle, and Gallegos-Cy-Aike river basins

pastures, draining water excess to Lake Buenos Aires. As these practices continue until present, the Fénix River flows nowadays toward both oceanic slopes. The Fénix River has a seasonal flow pattern, presenting high waters in winter and a very low summer flow.

Between Perito Moreno and the Pinturas River confluence, vertical walls confine the Deseado riverbed. The Ecker River, which collects runoff from small streams and groundwater from the SE portion of the Lago Buenos Aires Plateau, feeds the Pinturas River. The Pinturas goes S-N oriented, through a deep valley between high walls that expose Jurassic volcanic ignimbrite rocks (Geuna and Ecosteguy 2008). The intermediate valley is wide and preserves levels of terraces on both sides of the course. Downstream, the valley narrows when it crosses volcanic rocks, and almost all terraces disappear (Giacosa and Genini 1998).

The river flow diminishes downstream; it discontinues by sections due to infiltration and evaporation losses. Flow data in the Route 39 river section, 120 km downstream of Perito Moreno, indicates frequent null runoff between January and May and 1.3 $m^3 s^{-1}$ average yearly flow. Dunes spread on margins and terraces as well as wet meadows and marginal saline shallow lakes, fed by groundwater. Because of the poor water yield, the salt content in water varies over time in more than one order of magnitude. During a 3-year sampling period, Depetris et al. (2005) observed a TDS range from 140 to 9400 mg $L^{-1}$, with higher values produced by the occasional washing of marginal salines.

The river ends in a macro-tidal estuary, resulting from the flooding of the sea on the river sector which during glaciations had a base level far below the current state (Giacosa and Genini 1998). The estuary is 30 m deep in front of the town of Puerto Deseado and diminishes to 0.30 m 15 km inland, with a noticeable tidal effect up to 35 km from the coast (Isla et al. 2004). It presents a clear intertidal dynamics, with small or null fluvial flow. Depth, morphology, and orientation features define excellent port conditions. The estuary has a Provincial Natural Reserve status due to its high biodiversity.

Water is used for human consumption in the towns of Puerto Deseado, Pico Truncado, Las Heras, and Fitz Roy, among others. Water is also used in sheep farming activities, as well as in the oil and mining industries (Fundación Patagonia Tercer Milenio 2015). The hydrocarbon (HC) activity is concentrated primarily in the east, near the San Jorge Gulf oil basin.

Mining companies located to the north of the Deseado River use groundwater for a variety of processes. To the west, near Perito Moreno town, farms use irrigation to cultivate forage plants, vegetables, and fine fruits. There are some environmental issues associated with HC and mining operations. The intensive oil exploitation has accelerated the soil degradation triggered by livestock activities (Vázquez and Mazzoni 2004), while the mining use of cyanide to separate gold, silver, and copper from the rest of the rock as well as the eventual occurrence of spills have a significant impact on water quality.

## *4.13 Chico-Chalía River Basin*

South of the Deseado River basin, the Chico-Chalía (Shehuén) River basin (Fig. 9.7) extends over a surface of 23,964 km$^2$ (Díaz and Seeber 2015). The upper basins of both rivers are separated by an endorheic area which includes the lakes Strobel, Cardiel, Quiroga, and many smaller units (Chaps. 10 and 14). Ending the lower Pliocene, a tributary of the Chalía River captured the Chico River by backward erosion, near the current location of Gobernador Gregores. This way, the Chico River abandoned its original valley, parallel to the Chalía (Panza et al. 1998)

In the upper Chico River basin, the north-south oriented mountain range that divides waters with the Mayer River basin presents glacial cirques including small ice bodies and nivation niches that supply meltwater. Lake Burmeister, located in the Belgrano River sub-basin, is the largest in the basin. Belgrano stream joins the Chico River that flows from the west. The fluvial action shaped the valley, which presents three well-marked terrace levels (Panza et al. 1998).

The Chico riverbed is meandering, with patches of riparian vegetation and *mallines* in the margins, which contrast with the arid terraces. Annual precipitation in Gobernador Gregores, located at the center of the basin, is 204 mm (Consejo Federal de Inversiones 2013), without significant seasonal patterns. No extended flow series is available. The average flow for the 2017–2020 period was 24 m$^3$ s$^{-1}$ in the final section of Route 3.

The Chalía River runs from the west, through a valley whose morphology indicates a higher flow magnitude in the past. Horta et al. (2019) estimated the biggest expansion of Lake San Martín-O'Higgins in 12 Kyr BP. At that time, the SPI still blocked the lake drainage westward, and the lake level reached around 100 m above the present level, joining the Lake Tar, and draining to the Chalía River, which also collected the Lake Viedma outflow (Ponce et al. 2019). The Pascua River activation and the capture of the Viedma drainage by La Leona River resulted in the Chalía deactivation. The current average flow of this river is only 2.0 m$^3$ s$^{-1}$.

At present, Chalía sources are located in the Pana massif (2000 m a.s.l.), between lakes San Martín and Viedma. The river runs through a 50–100 m deep erosion furrow dug in marine sediments, when the river flow was vastly higher. Chalía and Chico Rivers join 80 km upstream of the Chico river mouth, in the northern arm of the estuary shared by the Chico and Santa Cruz Rivers, where Puerto Santa Cruz is placed.

Water is used for human consumption in the two small localities that exist in the basin: Puerto Santa Cruz, on the estuary that the Chico River shares with the Santa Cruz River, and Gobernador Gregores, which is a rural productive center. The Chico River Valley near the latter city has a network of canals improved by associations of local producers that irrigate thousands of hectares of farms with intensive agricultural activities. Erosion and salinization are visible in Chico River tributaries as a result of the concentration of agricultural operations on the margins of watercourses. In many sectors of the basin, the degree of desertification reaches very serious levels (Vázquez and Mazzoni 2004).

## 4.14 Santa Cruz River Basin

The river basin spans 28,400 km$^2$ in the west-east direction, across the entire Santa Cruz province (Fig. 9.7). This river is the only one between the large eastern outfalls of the Patagonian Icefield keeping its drainage direction after ice retreat. The Santa Cruz River has its origin at the eastern head of Lake Argentino and flows for about 330 km along a straight valley to reach the estuary that it shares with the Chico River.

The SPI covers most of the upper basin surface, and the ice-released valleys have been flooded by the arms of Lake Argentino. Hence, except in the River de las Vueltas sub-basin (unlinked from SPI), forest development is restricted to the free-of-ice steep hillslopes. After a sudden forest-steppe transition, the steppe ecoregion dominates the basin eastward of 72° 50′ W longitude.

Argentino and Viedma lakes are the main water bodies in the basin regulating SPI flows. Lake Viedma (1205 km$^2$) belongs to the Lake Argentino basin. The Viedma Glacier, the largest in Argentina, ends at the western head of the lake. Also, the River de las Vueltas discharges a flow average of 51 m$^3$ s$^{-1}$ into the northwestern shore. This river begins at the mouth of Lake del Desierto (9.2 km$^2$), which is located in the upper valley. La Leona River connects the lakes Viedma and Argentino, discharging 261 m$^3$ s$^{-1}$ in the northern sector of Lake Argentino's eastern head.

Lake Argentino has an extension of 1450 km$^2$ and a basin area of 15,500 km$^2$. Its lake arms extend westward, where the Upsala and Perito Moreno glaciers end. The average lake level is 178.2 m a.s.l., fluctuating usually between 180.5 and 176.9 m (EBISA 2017a); however, a greater variation results from the recurrent Rico arm obstruction by the Perito Moreno glacier. This ice blockage collapses spectacularly every few years, and the sudden water release causes the lake level to increase 1 to 3 m.

In the middle section, at the Cóndor Cliff area, the bottom of the valley presents low terraces with deposits of gravel in a fine sandy matrix. The tributaries in this section are small and more numerous on the north bank. From the south, short and intermittent streams run down from the marginal plateau.

The river width varies between 100 and 250 m, expanding by tidal influence in the last section, close to the town of Piedrabuena. The soils of the entire valley show strong degradation, with erosion pavements and mobile dunes. These effects were already evident in the mid-twentieth century (Castro and Brun 1964). Wind persistence and water deficit were the natural factors that triggered the most severe widespread erosion processes after its destabilization by overgrazing and intensive cattle trampling.

The Santa Cruz mean flow in the 1955–2019 period at Charles Fuhr was 724 m$^3$ s$^{-1}$, slightly higher than that determined by Iturraspe and Sottini (2005) for the 1955–2004 interval, due to the increase of the flow in the last decade to 831 m$^3$ s$^{-1}$. All annual flows from 2010 to 2019 exceed the series average value. The monthly average flow is lowest in September (306 m$^3$ s$^{-1}$) and highest in March (1319 m$^3$ s$^{-1}$). Waters are moderately alkaline (pH E 8) with very low salinity and nutrients along the river. TDS concentrations are around 30 mg L$^{-1}$ (Depetris et al.

2005, EBISA 2017a). Lakes Argentino and Viedma are oligotrophic, with high turbidity due to fine suspended glacier sediments.

Water is used to sustain tourism activities in the basin's mountain area, encompassed in Los Glaciares National Park, the largest one in Argentina. Various activities, such as navigation and ice trekking, are developed due to the beauty of the basin's rivers, lakes, and glaciers. Calafate, near the Park's entrance, and Chaltén, with Mount Chaltén-Fitz Roy in its background, located on the banks of the Las Vueltas River, are the two major tourist destinations in this region.

Fisheries in the river are characterized by specimens that reach world-class sizes, favored by the good quality of the basin's waters (Casalinuovo et al. 2019). In terms of hydroelectric generation, the construction of the Néstor Kirchner and Jorge Cepernic dams in the Santa Cruz River's middle valley is underway. The two plants will produce enough electricity to power 4% of the country (EBISA 2017b).

Water scarcity hampers livestock productivity. The region's overgrazing and the subsequent desertification in various areas of the central Plateau of Santa Cruz result in present sheep raising based on the extensive use of natural grasslands with low productivity and a decrease in the quality of the basin's waters, associated with important erosive processes. The degradation of water, soil, and vegetation associated with desertification alters the ecosystem's capacity to provide ecological services such as water purification among others (Paruelo et al. 2005). In the Andean sector, the strong geographical concentration of tourism poses a risk to the quality of its waters, on which the activity is based (Díaz 2005).

### *4.15 Coyle River Basin*

The Coyle River basin extends 16,300 km$^2$ in southern Santa Cruz, between Santa Cruz and Gallegos rivers (Fig. 9.7). The upper basin is made up of sub-basins of three tributaries: Pelque, Coyle Norte, and Coyle Sur. The Pelque's sources are located northwest in Meseta de las Vizcachas and in the eastern slope of Cerro Pináculo (2160 m a.s.l.). Further south, Coyle Norte stream runs eastward, with sources in the humid Magellan steppe, in rapid transition to the dry steppe (Oliva et al. 2001). Several streams converge to join the Coyle main course, 100 km upstream of its mouth. From the confluence on, the watershed narrows, since it is flanked by the plateaus. However, the abundance of *mallines* (Chap. 10) in the valley and lower river terraces indicates groundwater contribution from the plateau.

There is limited river flow data. Due to the lack of glaciers and minor lakes' presence, the flow regime depends exclusively on rainfalls in the upper basin, but the extended development of this area favors water capture.

Human settlements are scarce and scattered. Water is mainly used for human consumption and agriculture. Fluvial waters are alkaline (pH around 8.4) calcium and bicarbonate prevailing. After the confluence of the tributaries, TDS varies from 375 mg L$^{-1}$ (summer) to 440 mg L$^{-1}$ (winter), and SAR index indicates a low risk of sodium, resulting in general suitable water quality for irrigation in almost all kinds of crops and animal drinking, except for endorheic small lakes, which are usually saline, of very variable composition dominating chlorine and sodium (Casalinuovo et al. 2019).

### *4.16 Gallegos River Basin*

The Gallegos is a transboundary river with sources in the southern Chilean Magallanes Region, ending into the Atlantic shore, at the south of the Santa Cruz province (Fig. 9.7). Successive glacial advances shaped the southern part of the basin; glacial deposits, low volcanic reliefs, and streams dominate the landscape (Oliva et al. 2001). The upper basin corresponds to the forest-steppe ecotone, with forest patches, grasses, and *mallines*.

The Gallegos River takes its name after the Penitentes-Rubens confluence, wherein the average river flow is 29 $m^3 s^{-1}$. The monthly flow is highest in October (59 $m^3 s^{-1}$) and lowest in February (12 $m^3 s^{-1}$), showing a marked decline in the last decade (Table 9.1). Both rivers originate in Cordillera Vidal (Chile), in the boundary of the Patagonian forest. Downstream, the Turbio River joins from the northwest. It rises northwest of the basin, from the junction of the Primavera and San José streams. The Primavera Stream supplies most of the Turbio flow, from which there are no available flow statistics. Marderwald (2016), based on periodic measurements, reported minimum Turbio flow values between February and April, from 0.2 to 0.5 $m^3 s^{-1}$, increasing to near 2 $m^3 s^{-1}$ for the rest of the year with floods produced by winter rains. The San José Stream, before joining Primavera, passes through the city of Río Turbio and the homonymous coal mine. El Zurdo and Gallegos Chico streams drain modest flows from the south, with sources in Chile, and they join the Gallegos River downstream of the Gallegos-Turbio confluence. The flow regime is pluvio-nival, although seasonal snow storage is significant only in the upper basin. Before the estuary, the Gallegos flow averaged only 20 $m^3 s^{-1}$ between 2016 and 2020.

The estuary has a macro-tidal regime, with an 8 m average tidal amplitude, and 12.60 m maximum. Tidal influence reaches 12 km upstream. The Ci-Ayke River, also called Chico Sur, flows from the south 0.34 $m^3 s^{-1}$, sharing the estuary with the Gallegos River. The city of Río Gallegos, the capital of the Santa Cruz province, is located in the river's estuary.

Almost all human activities take place in the Argentinean section of the basin. Just over 40% of the population of the province is concentrated in this region of key economic importance, which is under increasing environmental pressure due to changes in land use (Chap. 10).

The water is used for agricultural irrigation in the surrounding area of Río Gallegos (Cepparo 2013), as well as livestock pasture irrigation in many cattle ranches. Mallines are also subjected to irrigation experiments to protect and preserve them. Surface and underground water are used by mining and hydrocarbon (HC) companies. Recreational fishing is carried out in the waterways of the basin.

The upper basin's streams such as the Rubens and Turbio have very low total dissolved solids (TDS) concentrations (ca. 40 mg $L^{-1}$). Although lower basin's tributaries have moderate salinity (ca. 180 mg $L^{-1}$), the waters of the Gallegos River keep TDS values below 100 mg $L^{-1}$ throughout the entire watercourse as well as a very low sodium absorption ratio (SAR), and then the natural quality of water is excellent for irrigation and other uses (Díaz and Tiberi 2017).

There are no studies that allow evaluating HC effects on surface waters, although localized spills have affected groundwater and some waterways. The watercourse of the Turbio River sub-basin is exposed to various sources of contamination. The impact caused by the industrial mining discharges of coal to the San José stream varies in a range from high to very high (Caballero 2014). This, combined with the domestic wastewaters of Río Turbio city and the stone washing of a quarry, notable degrades its water quality just before it discharges into the Río Turbio, which in turn shows a negative trajectory of its physical and chemical features due to the influence of the mine and the cities of Río Turbio and 28 de Noviembre (Casa et al. 2021). In all, studies carried out recommend taking urgent measures to treat the effluents before overturning them.

## *4.17 River Basins of Isla Grande de Tierra del Fuego*

The Isla Grande de Tierra del Fuego is separated from the American continent by the Magellan Strait. The western sector of the island belongs to Chile and the eastern sector to Argentina. Thus, a large part of the basins share water resources between both countries (Fig. 9.8). The spatial variability of the climate is less contrasting in TDF than in continental Patagonia due to both its insular condition and the W-E Fuegian Andes orientation along the south of the island, which moderates the rain-shadow effect. It results in a rich drainage network, where endorheic systems are reduced to small areas located in the NE of the island.

Iturraspe and Urciuolo (2000) classified Fuegian basins according to distinctive features, related to hydrological regulation systems, climate, morphology, and vegetation, recognizing the following basin types (Fig. 9.8).

- Mountain basins
- Transitional basins
- Steppe basins
- Peatland basins

Mountain basins extend entirely over the Patagonian forest ecoregion, on the Andean area south of TDF. Glaciers, seasonal snow, lakes, and peatlands (*turberas*, see Chap. 10) account for the hydrological regulation systems of basins. The *Nothofagus* forest is the dominant vegetation, while *Sphagnum* bogs are scattered on valley floors. Yearly precipitation in Ushuaia, at sea level, is 530 mm, but it almost doubles in the highlands. The Fagnano-Azopardo is the major basin of this class, occupying the center of the mountain area.

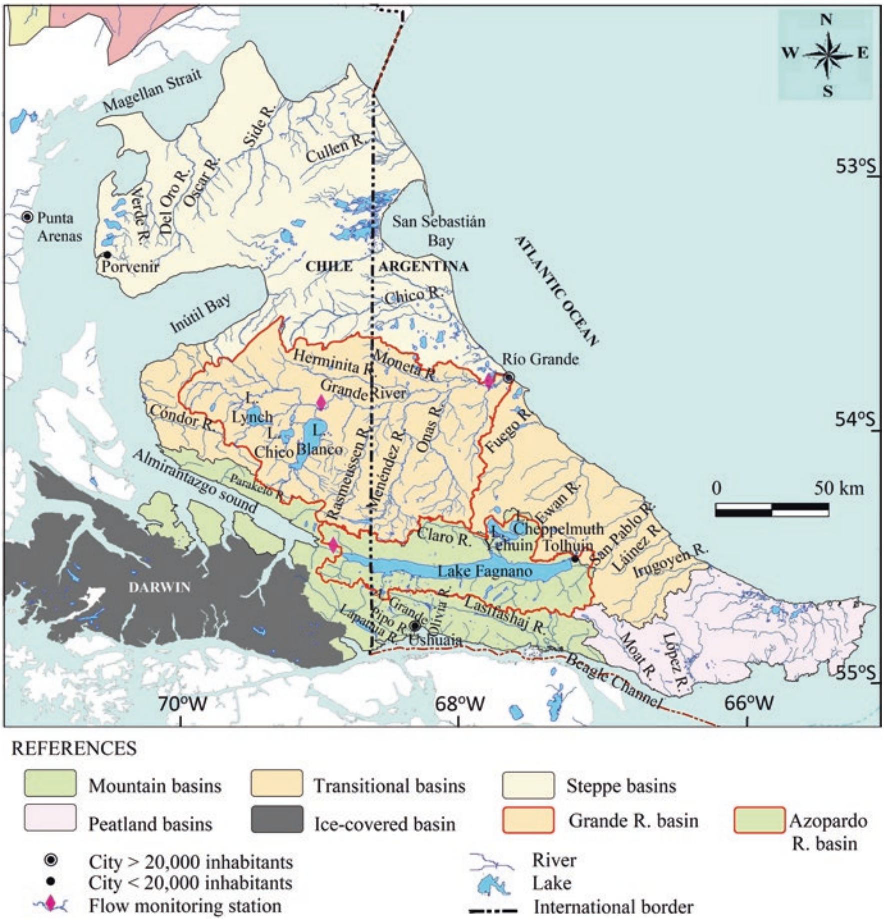

**Fig. 9.8** River basins of Tierra del Fuego

Small rivers drain into the Beagle Channel, showing an NW-SE development defined by tectonics. Among these, Lapataia River, with sources in Chile and discharge into Lapataia Bay (Argentina), has the largest flow, of about 19 $m^3 s^{-1}$ after the Lake Acigami-Errazúriz.

Pipo, Buena Esperanza, Arroyo Grande, and Olivia River basins involve attractive landscapes, with rivers ending on the shore of Ushuaia city. Among these, Olivia presents the highest flow rate, 5 $m^3 s^{-1}$ on average. The Lashifashaj River, flanked by the Sorondo and Alvear massifs, is 40 km long and discharges a flow average of 19 $m^3 s^{-1}$ into the Beagle Channel. In Chile, Paralelo and Cóndor Rivers flow from Beauvoir massifs of the Almirantazgo Sound.

The basins of the Darwin Icefield, in Chile, present differential hydrological patterns, and then they should be identified as ice-covered basins, where ice is the dominant land cover and rivers are characterized by their short length from glacier terminals to fjords. Among these, the longest is the Yendegaia River, which flows along 13 km from the Stoppani glacier to the Yendegaia Bay, in the Beagle Channel. There is no available flow data about these watercourses.

The Chilean sector of this area is not populated. The main city in Argentina's southern region is Ushuaia, the capital of the Province of Tierra del Fuego, Antártida e Islas del Atlántico Sur (TDF-AIAS), which is widely recognized as a gateway to Antarctica. The most important water use is human domestic consumption, with the Pipo, Buena Esperanza, and Arroyo Grande rivers providing drinking water to the city.

West of Ushuaia, Tierra del Fuego National Park, is a popular tourist destination due to the natural beauty of its landscape, rivers, and lakes. Water is used recreationally as well as in the Park's tourism facilities. The most common use of water in the mountain valleys to the east of Ushuaia is also for tourism and recreation.

Transitional basins spread north and east of Lake Fagnano. There are no glaciers in these basins, and seasonal snow accumulation is unstable. The forest gives way to humid grasslands that become dry northward. River valleys acquire a longer development, flowing to the Atlantic or the Inútil Bay shore in western Chile. The main one is the Grande River, whose basin covers a fifth of TDF. South of Río Grande, the rivers Fuego, Ewan, San Pablo, Láinez, and Irigoyen Rivers flow to the Atlantic Ocean.

Steppe basins extend north of TDF, exposed to west winds. The area receives about 300 mm of yearly precipitation, resulting in a deficit summer water balance. However, it is more humid than the continental extra-Andean steppe. In Chile, the Baquedano massifs (300 m a.s.l.), which extend between Inútil Bay and the Magellan Strait, is the source of Verde, del Oro, Oscar, and Side rivers that discharge into the Magellan Strait. The oriental face of this massif feeds the Cullen River, which flows toward the Atlantic Ocean. Further south, the Chico River travels eastward from the Carmen Silva massifs to Argentina and then parallel to the coast, before ending at the Río Grande city shore.

Cattle ranches in the steppe basins use water to irrigate pastures through flood irrigation. HC firms employ water for oil exploitation during the secondary recovery stage (Urciuolo et al. 2020). The intense HC activities, as well as compaction produced by livestock, have important hydrological effects in these basins, such as increased surface water runoff and sediment production (Borrelli and Oliva 2001).

Porvenir is the most inhabited population in Chile and the capital of the Chilean province of Tierra del Fuego. Main water-dependent activities in the area relate to cattle ranches and growing tourism. There are many rights of water use for irrigation and HC companies. Finally, peatland-dominated basins occupy the oriental sector of TDF, where these wetlands become the dominant water regulation systems, with cover rates over 80% in some cases (Iturraspe et al. 2013). Main river basins in this area are Moat, López, Sudamérica, Irigoyen, Malengüena, Leticia Bueno, Luz, and Policarpo. For several years, NGOs and several sectors of TDF's population have demanded this vast peatland area be officially conserved as a Provincial Protected Area (see Chap. 10).

In 1946, beavers (*Castor canadensis*) were introduced in TDF to develop the local fur industry. Taking advantage of a predator-free and sparsely populated environment, they rapidly colonized almost all streams in TDF and adjacent islands (Lizarralde et al. 1996), causing severe environmental shifts. Beavers destroy the riparian forest by cutting and flooding, affecting also peatlands and grasslands. Peatland hydrology is often altered by dams that flood peat-bog areas as well as by canals they dig to move through the peatland (Iturraspe 2022). Beavers also caused biophysical transformations. Downstream of dammed river sections, Rodríguez et al. (2020) found decreasing dissolved organic carbon concentrations and increasing phytoplankton abundance and periphyton taxa richness. Anderson and Rosemond (2007) reported decreasing macroinvertebrate diversity but increasing abundance and productivity linked to beaver activity.

There is constant monitoring in the Chilean Magallanes region to prevent the species from invading the continent through the Magellan Strait. In 2008, a binational agreement was signed for the restoration of beaver-affected ecosystems. Both countries are conducting pilot projects for beaver eradication, funded by the Global Environment Facility (GEF) and national counterparts (Schiavini et al. 2019).

## Grande River Basin

The Grande River is the most expanded fluvial system in TDF. Its basin area reaches 8600 km$^2$, of which 55% extends in Chile and 45% in Argentina (Fig. 9.8). The forest-steppe ecotone dominates most of the basin.

The main course crosses almost all TDF at about 53° 50′ S latitude. The westernmost valley, at 300 m a.s.l., is dominated by *Sphagnum* bogs and *Nothofagus* forest, contrasting with the dry features of the oriental river mouth, wherein the steppe dominates, and the annual rainfall drops to 330 mm. The main water contribution comes from the slopes of the W-E oriented Beauvoir massif, which separates the Grande River basin from Lake Fagnano and Almirantazgo Sound.

In Chile, the area of Lynch, Chico, and Blanco lakes adds up to 226 km$^2$. Lake Blanco is the largest (149 km$^2$) and incorporates the discharge of Lake Chico. The rivers Riveros and Blanco, respectively, connect the Lynch and Blanco lakes to the Grande River. Forest-peatlands-grasslands mosaics are the typical land cover in this lake's zone. In Argentina, the first tributary from the south is the Rasmussen River, called Bellavista in Chile, with sources in the Karukinka Natural Reserve. The next tributary is de la Turb-Menéndez River that begins at the outlet of Lake Deseado. Most of the lake extends in Chile, in La Paciencia Valley, but its outlet is situated in Argentina. De la Turba River runs northward and discharges in the Grande River through a 4-km wide valley, covered by peatlands, wet meadows, and grasses. The next southern tributaries are Onas and Candelaria, both flowing from the Beauvoir massifs. Northern tributaries are the Herminita River, with sources in the northwestern hills, and the small and intermittent Moneta stream, which fully extends in the steppe.

The Grande River meanders in the wide and gentle flood plain, wherein it bifurcates along some sectors. After floods, its path shifts, and abandoned meanders remain. The longitudinal slope of the valley in the Argentinean section is 0.1%. Suspended sediment in mid-water at the lower stream varies between 1 and 25 mg $L^{-1}$ of silt and clay, and between 0.5 and 15 mg $L^{-1}$ of fine sand. Flow average in the section of Route Y85, Chile is 30.3 $m^3 s^{-1}$. Considering the contribution of downstream tributaries by flow data from the Dirección General de Recursos Hídricos de TDF, the final discharge of the river is estimated at 45 $m^3 s^{-1}$. The fluvial regime is pluvio-nival, with the maximum monthly average in September–October and minimum between January and March (Table 9.5).

This transboundary basin was prioritized by Argentina and Chile for the development of a General Utilization Plan (GUP). These plans are management instruments that must establish guidelines for optimal and harmonious use of water between both countries (Gaviño 1999). The Chilean sector of the basin is sparsely populated. Water rights have been granted for domestic use and irrigation in cattle ranches to the north of the Grande River. Water is also used for tourist activities and recreational fishing.

In Argentina, the Grande River provides drinking water to the city of Río Grande, the most populous in TDF, located at its mouth. Agricultural producers in the suburbs use water for irrigation on a small scale. Other basins' water sources are used for human consumption by cattle ranches and border posts. Oil companies also use both surface and groundwater water for HC operations. Tourism is relevant in sport fishing lodges of international level, which are located along the entire river (Urciuolo et al. 2009). Recreational water uses such as fishing and boating are very important to local people. Difficulty of gaining access by locals to public domain water on private lands for recreational fishing resulted in conflicts between fishing lodge owners and local fishermen claiming their right to use and enjoy public waters (Urciuolo et al. 2009, Chap. 17). Certain agreements were reached after a long and complex dialogue process, though a final solution has yet to be attained. The quality of the water is good, although livestock activity generates diffuse pollution and increased sediment due to erosion caused by overgrazing. Wetlands and soils have been affected by sheep grazing in the northern region and by clearings in the upper basin (Borrelli and Oliva 2001; Chap. 10).

### Fagnano-Azopardo River Basin

The Fagnano-Azopardo River basin (3534 $km^2$) is the only transboundary basin in TDF draining from Argentina to Chile (Fig. 9.8). Lake Fagnano extends 104 $km^2$ along a tectonic fault, between the South American and Scotian plates. The Azopardo River, 10 km long, runs from the lake outlet to the Almirantazgo Sound.

Repeated Pleistocene glaciations shaped the basin. The Fagnano paleoglacier generated Fuego, Ewan, and San Pablo ice lobes that drained into the Atlantic Ocean and originated the Lakes Yehuin and Cheppelmuth, respectively. Flow reversal through the Azopardo River happened after the last depositional stage of the Fagnano paleoglacier that occurred between 12.5 and 11.7 ka. (Coronato et al. 2009). In addition to these lakes, which amount to a surface area of 679 km$^2$, there are several smaller lakes in the basin, such as Escondido, San Ricardo, Santa Laura, Margarita, and Yakush.

Around 40 tributaries converge into Lake Fagnano, which regulates almost the entire flow of the basin. It is an oligotrophic lake with very low salinity and transparent water (Mataloni 2014). The Claro River is the main northern tributary, which collects discharges from lakes Cheppelmuth, Yehuin, and Yakush. Small rivers join from the south, among which the Betbeder supplies the main glacial contribution from its SW headwaters near the Darwin Icefield. Azopardo average flow is 49 m$^3$ s$^{-1}$, having a very good natural regulation. The lowest monthly flow average in September is only 16% below the annual average, and the highest one in December is 24% above the average. During the 2007–2020 period, annual precipitation at the river mouth was 1228 mm.

In Argentina, the Tierra del Fuego National Park occupies the basin's southwestern part, and the "Corazón de la Isla" Provincial Reserve is located north of the lake. Other minor provincial reserves on the north and south lake banks play an essential role in conservation since forest regeneration is hampered by the browsing of livestock (Ríos 2017). The lands on the southern shore of the lake have been zoned for productive purposes. Tolhuin, the basin's main settlement, is located near the lake's eastern edge and offers services linked to forestry, peat mining, and tourism. The main use of water in Tolhuin city is domestic/sanitation. Tourist/recreational water use is carried out in the area, including navigation and sport fishing. Lake Fagnano constitutes an environment of high scenic value, maintaining its pristine character and good water quality conditions. The precarious treatment of the Tolhuin sewage discharges (Romano et al. 2019) has caused organic pollution in the wetland that is the direct receiving body, which in turn drains to the lake. A new wastewater treatment plant is expected to solve this problem. In the Chilean sector, the population is scarce, and the main economic activities are linked to primary production and tourism.

## *4.18 Main River Basins Flow Summary*

The analysis of the major Patagonian rivers basins represents an up-to-date and comprehensive outline of these hydrological systems. Table 9.1 shows a flow synthesis for main rivers and their tributaries, while Table 9.3 presents features of the Patagonian lakes greater than 30 km$^2$, although there is a lack of bathymetric information in many cases, notably in the south. The flow discharged into the Atlantic continental Patagonian shore reaches 1680 m$^3$ s$^{-1}$ mostly accounted for by the Negro and Santa Cruz rivers with 87% of the total (Table 9.4).

**Table 9.3** Patagonian lakes over 25 km$^2$ extension

| Lake | River basin | Country | Area | Mean depth | Max depth | Altitude | Emissary | Lat S | Lon W |
|---|---|---|---|---|---|---|---|---|---|
| Aluminé | Negro | A | 57 | 69 | 165 | 1155 | Aluminé R. | 38° 56′ | 71° 07′ |
| Tromen | | A | 29 | 69 | | 1050 | Malleo-Aluminé | 39° 32′ | 71° 27′ |
| Huechulafquen- | | A | 78 | 142 | 464 | 895 | Chimehuin | 39° 46′ | 71° 22′ |
| Lolog | | A | 36 | | | 902 | Quilquihue R | 40° 02′ | 71° 26′ |
| Traful | | A | 77 | | ~300 | 806 | Traful | 40° 38′ | 71° 26′ |
| Nahuel Huapi | | A | 557 | 157 | 464 | 764 | Limay | 40° 50′ | 71° 30′ |
| Espejo Grande | | A | 38 | | | 772 | L. Espejo Chico | 40° 38′ | 71° 45′ |
| Mascardi | Puelo | A | 39 | 218 | 111 | 808 | Manso | 41° 20′ | 71° 34′ |
| Puelo | | A | 46 | 111 | 180 | 192 | | 42° 09′ | 71° 38′ |
| Menéndez | Futaleufú-Yelcho | A | 56 | 149 | 287 | 523 | L. Futalaufquen | 42° 41′ | 71° 49′ |
| Futalaufquen | | A | 66 | 101 | 168 | 516 | Amutui Quimey | 42° 50′ | 71° 38′ |
| Amutui Quimey | | A | 92 | 65 | | 483 | Futaleufú | 43° 04′ | 71° 42′ |
| Yelcho | | CH | 116 | | 135 | 42 | Yelcho R. | 43° 18′ | 72° 17′ |
| Palena-Vintter | Palena | CH-A | 142 | | | 928 | Carrenleufú | 43° 56′ | 71° 39′ |
| Rosselot | | CH | 34 | | | 71 | Rosselot R. | 44° 06′ | 72° 20′ |
| La Plata | Senguer | A | 73 | | 182 | 931 | L. Fontana | 44° 53′ | 71° 51′ |
| Fontana | | | 85 | 79 | 119 | 928 | Senguer | 44° 56′ | 71° 30′ |
| Musters | | | 435 | 20 | 39 | 270 | Endorheic | 45° 25′ | 69° 11′ |
| Colhué Huapi | | | 710 | 2 | 4 | 254 | Endorheic | 45° 32′ | 68° 44′ |
| Elizalde | Aysén | CH | 26 | 108 | 203 | 220 | Blanco | 45° 46′ | 72° 17′ |
| Blanco | | A | 28 | 3 | 4 | 560 | Mayo | 45° 53′ | 71° 13′ |

(continued)

**Table 9.3** (continued)

| Lake | River basin | Country | Area | Mean depth | Max depth | Altitude | Emissary | Lat S | Lon W |
|---|---|---|---|---|---|---|---|---|---|
| Lapparent | Baker | CH | 25 | | | 482 | Ibáñez | 46° 15′ | 72° 15′ |
| B Aires-Carrera | | A-CH | 1803 | | 596 | 217 | Baker | 46° 27′ | 71° 42′ |
| Bertrand | | CH | 49 | | | 202 | Baker | 46° 55′ | 72° 50′ |
| Posadas | | A | 41 | 31 | | 152 | Pueyrredón | 47° 30′ | 71° 50′ |
| Pueyrredon-Cochrane | | A-CH | 320 | | >460 | 152 | Cochrane R. | 47° 16′ | 72° 03′ |
| Belgrano | Pascua | A | 43 | | | 811 | L. Nansen | 47° 51′ | 72° 08′ |
| Nanzen | | A | 43 | | | 800 | Mayer-Carrera | 47° 58′ | 72° 21′ |
| Tar | | A | 30 | <10 | | 270 | L. San Martín | 49° 14′ | 71° 59′ |
| S. Martín-O’Higgins | | A-CH | 1050 | | 836 | 252 | Pascua | 48° 52′ | 72° 21′ |
| Viedma | Santa Cruz | A | 1205 | | >520 | 253 | La Leona | 49° 38′ | 72° 28′ |
| Argentino | | A | 1450 | | 680 | 179 | Santa Cruz | 50° 14′ | 72° 37′ |
| Nordenskjold | Serrano | CH | 44 | | 200 | 65 | Paine | 51° 02′ | 71° 58′ |
| Sarmiento | | CH | 84 | | 312 | 77 | Endorheic | 51° 03′ | 72° 45′ |
| Grey | | CH | 40 | | 200 | 45 | Grey R. | 51° 04′ | 73° 09′ |
| Del Toro | | CH | 195 | | 300 | 24 | Serrano | 51° 13′ | 72° 44′ |
| Curicó | Endorheic lakes | A | 45 | | | 110 | Endorheic | 40° 37′ | 65° 55′ |
| Carrilaufquen | | A | 60 | | | 785 | Endorheic | 41° 08′ | 69° 28′ |
| Ghio | | A | 65 | | | 376 | Endorheic | 47° 17′ | 71° 32′ |
| Quiroga | | A | 49 | | | 1047 | Strobel | 48° 25′ | 71° 33′ |
| Strobel | | A | 100 | | | 724 | Endorheic | 48° 26′ | 71° 11′ |
| Cardiel | | A | 345 | 35 | 76 | 276 | Endorheic | 48° 55′ | 71° 13′ |

(continued)

**Table 9.3** (continued)

| Lake | River basin | Country | Area | Mean depth | Max depth | Altitude | Emissary | Lat S | Lon W |
|---|---|---|---|---|---|---|---|---|---|
| Lynch | Grande (TDF) | CH | 52 | | | 150 | Riveros-Grande | 53° 58′ | 69° 24′ |
| Chico | | CH | 25 | | | 146 | L. Blanco | 54° 01′ | 69° 11′ |
| Blanco | | CH | 149 | | | 120 | Blanco-Grande | 54° 08′ | 69° 01′ |
| Cheppelmuth | Azopardo (TDF) | A | 38 | | | 50 | Indio-Yehuin | 54° 24′ | 67° 35′ |
| Yehuin | | A | 43 | 40 | 118 | 43 | Yakush-Claro | 54° 25′ | 67° 43′ |
| Fagnano | | A-CH | 592 | 90 | 206 | 26 | Azopardo | 54° 34′ | 68° 00′ |

**Table 9.4** Patagonian basins synthesis. Average flow, surface area, and flow yield

| Pacific slope | | | | Atlantic slope | | | |
|---|---|---|---|---|---|---|---|
| River | Flow $m^3\ s^{-1}$ | Basin area | Flow yield $L\ s^{-1}\ km^{-2}$ | River | Flow $m^3\ s^{-1}$ | Basin area | Flow yield $L\ s^{-1}\ km^{-2}$ |
| *Continental Patagonia* | | | | | | | |
| Colorado | 140 | 46,500 | 3.01 | Puelo | 625 | 9000 | 69.44 |
| Negro | 734 | 138,000 | 5.32 | Yelcho | 570 | 11,500 | 49.57 |
| Chubut-Chico | 34 | 59,800 | 0.57 | Palena | 821 | 13,300 | 61.73 |
| Deseado | 1,6 | 27,600 | 0.06 | Cisnes | 215 | 5400 | 39.81 |
| Chico-Chalía | 22 | 23,964 | 0.92 | Aysén | 530 | 12,000 | 44.17 |
| Santa Cruz | 724 | 28,400 | 25.49 | Baker | 1000 | 26,500 | 37.74 |
| Coyle | 4 | 16,300 | 0.25 | Pascua | 715 | 14,200 | 50.35 |
| Gallegos | 29 | 12,100 | 2.40 | Serrano | 408 | 8850 | 46.10 |
| Totals | 1689 | 352,664 | 4.79 | Totals | 4884 | 100,750 | 48.48 |
| *Tierra del Fuego* | | | | | | | |
| Grande | 45 | 8600 | 5,23 | Azopardo | 49 | 3534 | 13,89 |

From the mouth of the Puelo River to the south, the analyzed rivers discharge into the Pacific Ocean an average flow of 4884 $m^3\ s^{-1}$. However, this value does not include direct or almost direct discharges into the sea from the large western glaciers of the NPI and SPI, nor that of the numerous streams draining small mountain basins of the western Andean slope. We estimate these non-computed flows at around 60% of the reported value. Significant freshwater inputs from river discharges and direct ice melting affect seawater salinity along fjords and adjacent seawaters, having a potential influence on the ecology of early life stages of marine fish occurring in the area (Landaeta et al. 2012).

Among the four great rivers that drain the eastern NPI and SPI sectors, only Santa Cruz contributes to the Atlantic Ocean, whereas the Baker, Pascua, and Serrano rivers jointly discharge 2123 $m^3\ s^{-1}$ into the Pacific fjords, which partially explains the big difference between the total freshwater contributions to the Atlantic and Pacific oceans. Flow yields are on average ten times higher in basins of the Pacific slope than in those discharging into the Atlantic Ocean (Table 9.4). Maximum rates correspond to the Puelo and Palena river basins (69.7 L $s^{-1}$ $km^{-2}$ and 61.4 L $s^{-1}$ $km^{-2}$). Although Baker and Pascua rivers have the highest glacier contributions from the NPI and SPI, they partially extend on dry areas of extra-Andean Patagonia, which explains their lesser flow production rates.

Finally, a significant flow decline was observed north of 45° S in the upper basins of Patagonia over the last decade, almost halving historical rates in some cases (Table 9.2). Except for the Santa Cruz River, the rest of the Santa Cruz province rivers have also reduced flow.

## 5 Institutional Regulatory Framework for Water Management in Argentina and Chile

The Argentine Republic has a federal organization system, wherein environmental competencies are concurrent between the Federal Government and the provincial states, according to the principles of the National Constitution (NC) of 1994. The NC also establishes that the original domain of the natural resources within the national territory corresponds to the provinces. Therefore, the latter dictate the necessary norms for its regulation and exercise management (Formento and Ferrazzino 2003).

The Civil and Commercial Code (CCC) regulates water classification, as well as other aspects related to the use of a public good, such as water (Martín 2012). In 2003, the Federal Water Agreement was signed between all jurisdictions, by which the “Principios Rectores de Política Hídrica” (Guiding Principles of Water Policy) of the Argentine Republic were adopted (OECD 2020).

The Argentine provinces regulate the use of water resources (Araujo 2002) through organizations that act as the Water Authority in each jurisdiction: they grant concessions and permits for the use of water under certain conditions. They are also responsible for the inventory, evaluation, and planning of water, the environmental control, and the management of other resources associated with water in the basin (Martín 2012), such as wetlands. Water authorities in Patagonian provinces are organized in different hierarchical levels and places within each government structure (Pinto 2014). At the national level, the SIPH (Table 9.5) is the institution responsible for setting and executing the national water policy. Other areas of the national government imply environmental and sectoral issues of the use and control of water resources. The COHIFE is a Federal Council made up of the Provincial States, the Autonomous City of Buenos Aires, and the National State that oversees global, strategic, interjurisdictional, and international aspects of water resources.

Many basins extend over more than one province. This has called for the creation of interjurisdictional basin organizations in order to facilitate the integrated management of shared water resources, wherein the provincial jurisdictions involved

**Table 9.5** Institutions related to water management of Patagonian basins

| Initials | Name | Website |
|---|---|---|
| SIPH | Secretaría de Infraestructura y Política Hídrica de Argentina (Secretariat of Infraestructure and Water Policy) | https://www.argentina.gob.ar/obras-publicas/infraestructura-y-politica-hidrica |
| COHIFE | Consejo Hídrico Federal (Federal Water Council) | https://www.cohife.org |
| COIRCO | Comité Interjurisdiccional del Río Colorado.(Inter-jurisdictional Committee of the Colorado River) | www.coirco.gov.ar |
| AIC | Autoridad Interjurisdiccional de las Cuencas de los ríos Limay, Neuquén y Negro. (Interjurisdictional Authority of Limay, Neuquén, Negro River Basins) | http://www.aic.gov.ar/ |
| ACRS | Autoridad de Cuenca del Río Senguer (Senguer River Basin W.A.) | https://www.argentina.gob.ar/obras-publicas/hidricas/10-comite-de-cuenca-del-rio-senguer |
| ACRA | Autoridad de cuenca del río Azul. (Azul River Basin W.A.) | https://www.argentina.gob.ar/obras-publicas/hidricas/autoridad-de-cuenca-del-rio-azul |
| DGA | Dirección General de Aguas. Gobierno de Chile (General Water Directorate) | https://dga.mop.gob.cl/ |

and usually also the nation take part (Pochat 2005). The COIRCO Committee is the older water basin organization in Argentine, created in 1969, followed in Patagonia by the AIC Water Authority (WA), the ACRA WA, and the ACRS WA (Table 9.5).

Chile is a unitary country with regional governments and a presidential administration. The Water Code endorsed in 1981 states that a “water right” integrates the use, enjoyment, and disposal of water into the patrimony or domain of its owner. Although waters remain national goods for public use (Donoso et al. 2004), a new concept of “individual property” appears concerning water separate from land, thus permitting the holder of a water right to transfer it. As a result of allowing unrestricted purchase and sale of water, a “water market” was created (Larraín et al. 2010). The 1981 law was partially modified in 2005, establishing a canon or patent for the nonuse of water. There are additional regulations for environmental aspects of water and other related resources in the water basin, such as forests and wetlands.

The General Water Directorate (DGA) is the state agency in charge of promoting the management and administration of water resources. It has regional directorates throughout the country. There are also user organizations that are regulated by the DGA.

Argentina and Chile have numerous transboundary water resources, and therefore rules based on consensus choices were developed to enable a harmonic and equitable use of water (Gaviño 1999). As a consequence, an agreement on hydrological basins was achieved in Santiago in 1972, and both nations signed a Friendship

Treaty in 1984. This established a permanent Binational Commission whose efforts culminated in the Environment Treaty (1991) including an Additional Protocol on Shared Water Resources. This protocol envisions the creation of an Argentine-Chilean Working Group on Shared Water Resources and the development of general basin utilization plans (GUPs) including the actions and programs for the use of shared water resources (Gaviño 1999). This group, founded in 1996, defined a structure for the GUPs and established that they must be designed and implemented by local subgroups of experts.

## 6 Main Water-Related Conflicts in Patagonian Basins

Water is an element that stands out as a central axis in the field of political conflicts in the study basins (Bottaro et al. 2014). The expansion of traditional productive activities, as well as the introduction of economic development alternatives, resulted in a variety of effects on the region's water resources and wetlands. Based on information obtained on the biophysical characteristics and land/water uses in Patagonian basins, an analysis of the major conflicts linked to pressure on freshwaters is provided below.

The emergence of new economic activities entailed an acceleration in the expansion of the main Patagonian cities, which in many cases occurred without any planning or ability to respond to the needs of the population (Ciminari et al. 2005; Fank 2019). In Northern Patagonia, tension around land use in the Alto Valle of Río Negro and Neuquén is rising. The urban advance on rural productive lands and natural wetlands affects nearly all of the towns (Svampa 2016; Chap. 10). In addition to the reduction of the agricultural areas, this process has seriously impacted the irrigation system (Catoira 2017) by physical modifications in the channels, water pollution from effluent discharges, and deposition of solid urban waste (FAO 2015; Chap. 11), which resulted in a decrease in food production. Rural and urban land-use coexistence also generates conflicts derived from common farm practices such as the use of fertilizers and pesticides (FAO 2017) or the use of active frost control methods, which may affect the nearby inhabitants (FAO 2015).

In Southern Patagonia, the rapid expansion of several coastal towns has come at the expense of existing wetlands, as is the case of the city of Río Grande, TDF, which has lost a third of its wetlands in the last 50 years. Primary issues include waterlogging by urbanization on wet meadows that obstruct natural drainage, damage to coastal zones, and the loss of a substantial part of its beaches (Iturraspe et al. 2021). Due to its urban dynamics, the city of Río Gallegos (Province of Santa Cruz) has lost 246 hectares since the 1960s. Estuary wetlands are still being harmed, even though they are public goods with recreational value due to their biodiversity (Diez et al. 2015). In main touristic cities, high-value wetlands show signs of water pollution and deterioration (Blanco et al. 2017). These problems gave rise to multiple claims by NGOs for the protection of wetlands and urban freshwater environments, which in several cases were resolved in court (Burgos 2020; Urciuolo et al. 2009; Rojas Quezada 2020). The Chilean Parliament enacted Law 21202/2020, intending to protect urban wetlands and improve their precarious regulatory situation (Valenzuela 2021).

The hydrocarbon (HC) activity, of great relevance in several Patagonian basins, has been the cause of surface and groundwater pollution by recurring spills that have reached water bodies (Scandizzo 2018), and soil degradation by pipe constructions and dense road networks connecting oil wells, buildings, and installations (Váquez and Mazzoni 2004). These outcomes resulted in local socio-environmental conflicts, notably in zones where water is scarce. The expansion of HC production on native communities' territory triggered a series of disputes in the early 2000s when they protested against the health and environmental consequences of hydrocarbon extraction (Scandizzo 2018; Chap. 16).

Although agriculture and hydrocarbon production have coexisted for decades in the region (Guibert et al. 2018), the pressures on land and water resources and the impacts generated by the exploitation of hydrocarbons in some areas increased the loss of productive soil dedicated to fruit growing under irrigation (Svampa 2016; Rodil 2015). Gas exploitation amidst farms entails several hazards and negative effects on the productive area (Catoira 2017); in recent years, there have been oil spills, broken water pipes, explosions, and fires among other deleterious effects (Rodil 2015).

The discovery of UHC resources generated new conflict scenarios in the region (Acacio and Svampa 2017). Their exploitation entails more civil works than previously existed, truck traffic, and an expanding number of facilities, resulting in severe soil degradation and a variety of socio-environmental undesired effects (Sosa 2014). Continuous expansion of UHC exploitation implies increasing water consumption and the generation of return volumes that cause environmental impacts on water resources (FAO 2015). Burton et al. (2014) identify the potential effects on aquatic natural resources as well as the hazards to water quality. Therefore, although shale production still consumes a relatively modest quantity of water in Patagonia, quality issues resulting from industry effluent may compromise future water availability in particular water-scarce subregions (Forni et al. 2021).

The HC production was put in tension at various times by territorial conflicts, but after the development of the fracking technique (Howarth et al. 2011) for UHC exploitation, criticism on this activity has been addressed by the socio-environmental agenda (Scandizzo 2018). In 2013, there was a large mobilization in Neuquén under the slogan "No to fracking" due to the environmental risks associated with this activity (Acacio and Svampa 2017). Since then, strong opposition movements have arisen in regions with no history of oil and gas extraction (Scandizzo 2018). The role of national and local governments in ensuring compliance with security measures that safeguard water is vital for the effective development of this activity.

During the last two decades, there have been a series of socio-environmental disputes concerning mining expansion in Patagonia, supported by new and significant investments, with water serving as "the core axis and common denominator" of these conflicts (Bottaro et al. 2014). Mining demands large amounts of water in the chemical processes used to produce and transport mineral concentrates, competing for access to water (Budds and Hinojosa 2012). It also involves the risk of water contamination, both by the chemicals used and by acid mine drainage (Higueras et al. 2004). Also, mining operations near glacial environments cause dust deposition on the glacier surface, which reduces albedo and increases melt processes and glacier recession (Chap. 4).

Conflicts related to mining and water acquire similar features in Chile and Argentina. The discourses in opposition to mining in the public sphere on both sides of the mountain range highlighted the impact of mega-mining as a threat to life and focused on pollution and water scarcity. The "water-life" relationship emphasizes the value placed on water by mobilized actors – assemblies, social organizations, and communities – who regard water as a common good endangered by mining operations (Bottaro et al. 2014; also see Chap. 17).

The water legal framework is critical for conflicts between mining companies and communities. In Chile, water privatization has been a major contributor to the country's controversial relation between mining and water. The socio-environmental movements against mega-mining mainly focus their claims on the lack of water for other uses due to rights granted to mining companies ("The thirst for gold will leave us without water"). The national network of organizations that defend water, created in 2009, took the name *Coordinadora por la Defensa del Agua y la Vida* (Coordinator for Defense of Water and Life), questioning the legitimacy of property rights over water (Bottaro et al. 2014).

Along with the mega-mining expansion in Argentina, major socio-environmental conflicts have emerged across many provinces. The high environmental and economic effects of mega-mining along with its extractive character have sparked a wide variety of organizations and social movements (Galafassi 2008). The "self-convened neighborhood assemblies" have rejected the expansion of the mining model in their territories, expressing concerns about its effects on water. Patagonian Esquel City was the first place in Argentina wherein people's action forced a mining exploitation project to halt operations. Through a popular consultation in 2003, the citizens of the region responded no to mining proposals in the mountain range (Galafassi 2008). The slogans "water is worth more than gold" and "no to the mine" became generalized and later replicated in other Patagonian provinces. As a result of this process, laws forbidding open-pit mining were passed in many of these provinces (Schweitzer 2019).

The physical properties of many Patagonian rivers indicate their potential for hydroelectric power, which has led to confrontations between governments, local populations, and nongovernmental groups. Hydroelectric generation companies acquired the property of more than 90% of the water in the Aysén region over a long time, as a result of rights granted by the Chilean state for hydropower development initiatives in the 1990s. This situation resulted in unfulfilled water demands among local riparian owners who did not have access to water resources

(Romero and Toledo 2009). Hidroaysén, led by Spanish and Chilean companies, was the main of these projects, involving the construction of five mega-power stations on the Baker and Pascua rivers (Universidad de Chile 2016; Reyes and Rodríguez 2015). Generated electricity should be sent to the north of the country through the Central Interconnected System. Faced with the possible impacts in the region, various actors from the Aysén community expressed their concern, by stating that the project would have no regional advantages and would drastically alter the natural conditions of both River basins and local ecosystems. Arising organizations converged in a movement known as *Patagonia sin represas* (Patagonia without dams), which attempted to build alliances between the local civil society, indigenous peoples, and national, regional, and even international nongovernmental organizations (Schaeffer 2017). This movement put pressure on the executive power's intentions for Aysen's resources since it affects more than just the residents of the region, to the point that Aysén is regarded as a "life reserve" (Reyes and Rodríguez 2015). After more than 10 years of legal controversies and conflicts between the regional administration and the organized community, the project was rejected by the ministerial committee and the environmental court, and finally, the companies renounced their water rights by late 2017. After the failure of the Hydroaysén project and due to the high cost of keeping many inactive water rights, in 2018 several companies returned water rights obtained for hydropower and canceled projects for new ones (Peña 2018). The social movement involved in this process reflected a cultural change that clearly showed how environmental and social concerns are intimately linked (Romero 2014).

Another paradigmatic case is the one concerning the dam projects on the Santa Cruz River in Argentina. The proposed dams' location and size generated socio-environmental conflicts due to the possible impacts on the aquatic ecosystem and potential changes to the dynamics of glaciers in contact with Lake Argentino, as well as the lack of a proper environmental impact study (EIS). Shortly after the dam's construction started, NGOs filed a precautionary lawsuit, which was accepted by the Supreme Court, and the suspension of the works was ordered in early 2016 (Acevedo 2017). In 2017, the Government presented an EIA as well as a project reformulation reducing the number of turbines and the height of the dams, thus decoupling its influence on Lake Argentino to avoid impacts on the glaciers (EBISA 2017b). Subsequently, the Court authorized the resumption of work based on the new project; however, at present, NGOs and other sectors still argue EIA's deficiencies and uncertainty about the magnitude of negative effects that the dams will produce on the aquatic environment.

Beyond differences between the positions on the case, it is worth highlighting the social learning that this conflict implied, since it managed to make audible voices usually silenced and articulated territorially distanced actors with different interests. Institutional advances and new networks contributed to strengthen the socialization of information and the demand for public participation in decision-making granted by the National Constitution (Borsellino and Pereira 2020).

## 7 Climate Change and Water Availability in Patagonia

Significant changes in mountain glaciers of the world gave, in the 1970s, the first warnings of climate change at the global level. In Patagonia, glacier recession and disappearance occurred along the Andes as a result of rising temperature and precipitation decline, as well as noticeable changes in large glacier terminals of the Patagonian Icefields. The glacial lake area at the periphery of the NPI increased by 64.9% between 1945 and 2011 because of glaciers' front retreating (Louriaux and Casassa 2013).

Droughts that have persisted in the Andes in recent decades have hampered both extensive and intensive rural production, particularly in North Extra-Andean Patagonia, by reducing natural pasture productivity and by interrupting water supply for irrigation in critical situations when the ecological flow and water supply to populations have been threatened. There is a close relation between precipitation in the Andes mountains and hydroelectric generation which have significant consequences for the dam's operation and therefore energy supply. In the Comahue Region, which includes the Limay and Neuquén river basins and involves 37.8% of the hydropower installed capacity in Argentina, a trend of −97.9 GWH year$^{-1}$ was observed in 1994–2016 and −403.4 GWH year$^{-1}$ regarding the 2001–2016 period (González et al. 2021). The average hydroelectric generation during 2011–2016 dropped by 37% regarding that of 2001–2006. This trend still persists in 2021. Projections (Table 9.6) indicate that these basins, as well as that of Chubut and Senguer rivers, will be the most affected in Patagonia by global warming in the late twenty-first century (Barros et al. 2014).

All glacier studies foresee an important reduction of glacier volume in the Andes by the end of the current century, and even the disappearance of many glaciers, particularly small ones (Masiokas et al. 2020; Chap. 4). Despite uncertainties inherent to the modeling of future water yield projections, which are subject to internal variability in the Global Circulation Models and the particular nature of atmospheric

**Table 9.6** Projected temperature increase in Patagonia under two GHG emission scenarios

| IPCC representative concentration pathway (RCP) | Temperature increase relative to (1986–2005) as the reference period | |
|---|---|---|
| | Near term | Long term |
| RCP 4.5<br>Assuming decreasing GHG emissions from 2040 on | 0.5–1 °C in continental Patagonia<br><0.5 °C in TDF | 1.5–2 °C in the upper Colorado and Neuquén River basins<br>1–1.5 °C in the rest of Patagonia |
| RCP 8.5<br>Assuming continuous increase in GHG emissions during the twenty-first century | | 3–3.5 °C in the upper and medium river basins of Argentina north 46° S<br>2.5–3 °C in the rest of continental Patagonia<br>2–2.5 °C in TDF |

Source: Barros et al. (2014)

processes, Pessacg et al. (2020) found clear evidence of critical decrease in yearly water yields of Limay and Chubut river basins in the far future.

Research on projections of seasonal flow changes for this area, wherein the summer is markedly dry, is necessary. Snowpack decline is an expected result of increasing temperature and potential winter precipitation reduction, and it would raise the intensity and frequency of severe winter floods, as well as extended summer flow shortages in months with higher water demand. These foreseeable effects that so far have not been systematically evaluated represent a potentially high impact over extra-Andean Patagonia which will aggravate the present conflicts due to the increasing water demand in the most productive areas. The proper management of this situation in the next decades will be the greatest challenge for sustainable development in Northern Patagonia.

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# Chapter 10
# Patagonian Wetlands: *Vertientes*, *Vegas*, *Mallines*, *Turberas*, and *Lagunas*

**Luis B. Epele, Elizabeth Mazzoni, Rodolfo Iturraspe, Carolina León, Erwin Domínguez Díaz, María Laura Miserendino, and Gabriela Mataloni**

## 1 Wetland Types: Definitions

As mentioned in Chaps. 1 and 2, Argentinian and Chilean Patagonia show a wide variety of habitat types, some of which are rare and unique to each country. However, and taking into account that Patagonia stands as a single Region among those defined in the wetland regions of Argentina (Benzaquén et al. 2017), in this chapter, these environments will be addressed independently from their location regarding political borders and the three subregions described in that work (Buria 2017; Mazzoni 2017; Mataloni 2017), and unless clarified, Patagonia is here regarded as a single geographical region beyond the border between countries (Coronato et al.

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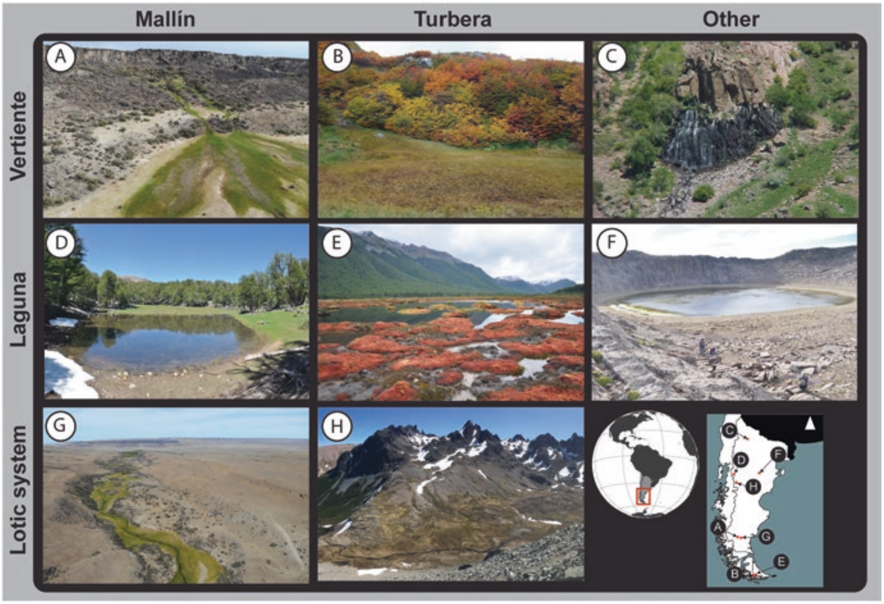

**Fig. 10.1** Examples of mallines, turberas, lagunas, and vertientes from Patagonia and their interactions. This figure should be interpreted as a double-entry table. For example, mallines and turberas (first and second column) can be associated with different waterbodies (i.e., vertientes, lagunas, and lotic systems; displayed in rows). However, not all the vertientes and lagunas are associated with mallines or turberas, with some examples displayed on the "other" column. (**a**) Vertiente-fed mallín from a volcanic plateau. (Photo by J. Fenoglio); (**b**) turbera from the Vinciguerra valley, surrounded by autumnal *N. pumilio* forest. (Photo by R. Iturraspe); (**c**) "piedrasmeonas" a vertiente that does not generate neither a mallín nor a turbera in its immediate margins. (Photo by E. Mazzoni); (**d**) montane laguna with adjacent mallín. (Photo by L.B. Epele); (**e**) turbera from Tierra del Fuego Island in Sphagnum matrix. (Photo by L.B. Epele); (**f**) temporary laguna from a basaltic plateau in Talagapa ranch. (Photo by L.B. Epele); (**g**) mallín from an alluvial plain located in the middle Santa Cruz basin. (Photo by J. Fenoglio); (**h**) montane turbera from central Patagonia. (Photo by L.B. Epele)

2008). Although more wetland types could be found in Patagonia, this chapter is focused on the most common and widely distributed ones (Fig. 10.1).

## 1.1 Mallines and Vegas

Most Patagonian wetlands, turberas excepted, are locally called "mallines" in plural and "mallín" in singular, words that derive from the mapudungun (Mapuche nation language), and means swampy area or lowland area where water accumulates. "Vegas" is normally used as a synonym of mallines (Movia 1984) by southern Patagonia inhabitants. Although it is sometimes reserved for mountain wetlands (Reboratti 1982), here we assume both terms as synonyms and hereafter will be jointly referred to as mallines.

Mallines are defined by Movia (1984) as humid grasslands of high density and floristic richness, whose genesis is associated with the presence of water-saturated soils, mostly groundwater. They are characterized by a central area of high moisture dominated by hydrophilic plant species and a periphery dominated by mesic species (Boelcke 1957) varying according to its geomorphological location and water source (Mazzoni and Rabassa 2013). This higher water availability leads to the development of azonal types of soils and plant communities that are more productive than the surrounding forest (Gandullo and Schmid 2001) or steppe (Irisarri et al. 2012). Within the steppe matrix, mallines constitute oases containing not only a large proportion of the aquatic and wetland-specific biodiversity but also providing environmental resources for many other non-wetland flora and fauna species (Epele et al. 2021a) (see Chap. 15). In turn, most mallines from forests are located on the Andes mountains and thus normally used as summer pasturelands (Perotti et al. 2005). Besides providing pastures for livestock, all mallines contribute to people with materials, fishing activities, and recreation areas, and many have importance for Mapuche culture and heritage (Chaps. 2 and 16). Mallines are distributed across Patagonia, from the Pacific to the Atlantic Ocean (Blanco and de la Balze 2004; Kandus et al. 2008; Filipová et al. 2013; Benzaquén et al. 2017) but are scarce in arid Argentinean Patagonia (Crego et al. 2013). The most appropriate environments for their development are the glaciolacustrine and glaciofluvial plains, alluvial plains, and slopes with springs, particularly in basaltic plateaus (Mazzoni and Vazquez 2004).

## *1.2 Turberas*

The "turberas" (peatlands) are wetlands with a naturally accumulated, well-defined, at least 30-cm-thick organic matter layer on its surface termed "peat" (Joosten and Clarke 2002). Environmental settings such as permanent waterlogging and poor water circulation through the peat substrate result in anoxic conditions that, together with low temperatures and pH, impede decomposition of local dead plants, thus forming in turn more peat (Rydin and Jeglum 2006), which usually also includes animal remains and mineral sediments as minor components. A mire is a peatland wherein peat is actively formed. According to their genetic origin and surface morphology, mires are mainly classified into bogs (peatlands raised above the landscape and receiving water only from precipitation) and fens (depressional peatlands fed by water that has been in contact with mineral bedrock or soil) (Joosten and Clarke 2002).

Peatlands fulfill key functions and values. Among them, the most recognized is their role as carbon sink and carbon storage: while peatlands and organic soils cover 3% of the global land area, they contain 30% of its total carbon (Parish et al. 2008), thus having the potential to affect global carbon cycle regulation, and in turn, climate change (Humpenöder et al. 2020). They also supply water regulation and control erosion and sediment transport. Mires support a particularly adapted biodiversity and make up scenic landscapes (Iturraspe and Urciuolo 2021). Peat has different applications, although its extraction requires peatland drainage and hence implies the degradation of the ecosystem (Joosten and Clarke 2002). While turberas can be found from northern to southern Patagonia, they reach their maximum extension and diversity in latitudes higher than 44° S (Pisano 1977; Blanco and de la Balze 2004; Schlatter and Schlatter 2004; Díaz et al. 2015; Benzaquén et al. 2017), with 95% of the Argentinian turberas located in Tierra del Fuego Island (Rabassa et al. 1996) (see Chap. 6). In northern Patagonia, in turn, they are sparsely distributed along the Andes and mainly associated with montane environments (Malvárez et al. 2004) (Fig. 10.1h).

### *1.3 Lagunas*

The Spanish word "laguna" (or lagunas in plural) can be used as a synonym of small lake, shallow lake, pond, or depressional wetland. Unlike the terms "mallín" or "turbera," the word refers to the waterbody, regardless of the surrounding landscape. Therefore, a laguna could be part of a mallín, a turbera or standalone surrounded by a matrix of typical forest or steppe vegetation (Fig. 10.1). Their hydrology is highly dependent on the water input by precipitation and the loss by evapotranspiration and also on the inflows and outflows by lotic and ground waters (see Chap. 2). Therefore, Patagonian lagunas can be permanent or temporary (those that dry every year or are eventually flooded) (see Fig. 10.1d–f). These habitats can support tourism and recreational activities and harbor some endangered species like the Hooded grebe (*Podiceps gallardoi*) (see Chap. 15), and many are used for native and exotic species fishing (see Chaps. 13 and 14). Additionally, across the extensive interfluvial dryland areas between large Patagonian rivers (e.g., Coyle and Gallegos rivers) as well as in basaltic plateaus, lagunas in endorheic depressions (locally named "bajos sin salida") are usually the unique freshwater sources (Fig. 10.2) (Mazzoni and Rabassa 2018). Lagunas are distributed across the Patagonian region, from the forests to the steppe, being particularly dense and numerous in volcanic landscapes such as basaltic plateaus (Fig. 10.1f).

### *1.4 Vertientes*

Vertientes (sometimes called "manantiales") is the Spanish word for springs. There are plenty of definitions for this wetland type, with some based on the immediate physical and chemical environment, and others focused on the communities of organisms living there. These definitions stem from different disciplines. From a hydrological viewpoint, a "vertiente" represents a concentrated point of natural discharge of groundwater that is sufficiently high to maintain flow on the land surface

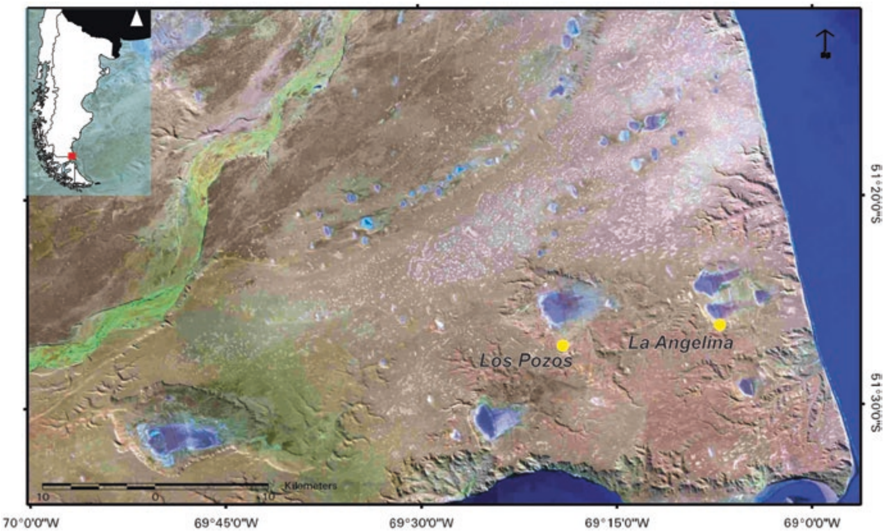

**Fig. 10.2** Satellite image showing the distribution of temporary lagunas in the interfluvial area between Coyle and Gallegos rivers. See that two ranches (yellow dots) are located close to the larger lagunas. Satellite image Landsat 8 228–96, OLI sensor, bands 6,5,3 (April 15th, 2021). (Source: USGS (https://earthexplorer.usgs.gov/))

(van Everdingen 1991). The flow of groundwater from a vertiente often comprises a mixture of water that has infiltrated the subsurface at different times and places (van der Kamp 1995). The rate of this recharge varies according to the balance between precipitation, infiltration, runoff, and evapotranspiration, and as the contributions from different recharge points change, so will the output of the spring vary, yet is normally persistent along the year. The constant supply of water produced by these vertientes can generate mallines, turberas (i.e., spring-fens and percolation-fens), and small lagunas in lowland areas (Fig. 10.1) (Melendi et al. 2020). For example, mallines produced by vertientes are typical features of volcanic landscapes (Mazzoni and Rabassa 2018) but could also be found in other Patagonian landscapes (i.e., Andes mountains, steppe, etc.).

Vertientes are widely appreciated for their relatively good water quality, being used as drinking water sources for people, livestock, and native biodiversity (Mazzoni and Rabassa 2018; Morales et al. 2020). This kind of wetland has been recognized for its role supporting specialized biota, mostly in harsh environments (i.e., Patagonian steppe) wherein their relatively stable conditions contribute to maintaining and enhancing the local biodiversity (Williams 2016). However, the biodiversity of Patagonian vertientes has been widely neglected, and much more comprehensive studies are needed to understand their current distribution and role for the regional biodiversity.

## 2 Nature Contributions to People

The benefits that human beings obtain from wetlands are vital to understand human-environment relationships, in order to design proper management actions for their sustainable use (see Chap. 2). In the last decades, the valuation of wetland ecological services has significantly risen for the expression of values assigned by people to various functions of ecosystems (Moomaw et al. 2018). Nevertheless, the ecosystem services conceptualized by the Millennium Ecosystem Assessment (2005) have reached far into the mainstream, embraced into 18 categories under the concept of *nature's contributions to people* (IPBES 2019). These categories are grouped into three types, including regulating, material, and nonmaterial contributions (see Table 10.1).

The ecosystem services provided by the Patagonian wetlands have been poorly assessed, with just a few available studies dealing with specific wetland types (Iturraspe and Urciuolo 2021; León et al. 2021) or including wetlands into watersheds (Delgado et al. 2013). Therefore, the information on Table 10.1 is based on expert judgment of the authors on nonspecific literature from the region, in an effort to highlight the numerous contributions to people provided by Patagonian wetlands. Three IPBES categories that would not be contributed to by freshwater Patagonian wetlands are excluded (i.e., pollination and dispersal of seeds, regulation of ocean acidification, regulation of hazards, and extreme events). It is worth noting that the provision of wetland contributions to people will depend on their water and landscape quality, their biodiversity conservation status (i.e., species composition, diversity, and richness), and the scale and extension of changes in their habitats, topics that will be discussed in the following sections.

## 3 Wetland Water Quality and Biodiversity

Environmental Patagonian heterogeneity, driven by north-south (e.g., air temperature) and west-east gradients (e.g., precipitation) and different geological features (see Chap. 1), generates patterns of wetland water quality and freshwater biodiversity rigorously studied at local and regional scales (Fig. 10.3), despite still needing more research efforts. Their knowledge is growing rapidly, and here some findings are summarized per wetland type.

### *3.1 Mallines, Their Lagunas, and Vertientes*

As mentioned above, mallines could be associated with different geomorphic environments and climatic conditions, as long as they receive enough water inputs to preserve the hydromorphic soils. In particular, the origin or genesis of

**Table 10.1** Relative magnitude of Patagonian wetland type contributions to people, subdivided into regulating, material, and nonmaterial contributions

| Wetland contributions to people | M* | T | L | V | References from Patagonia |
|---|---|---|---|---|---|
| *Regulating* | | | | | |
| Habitat creation and maintenance | ↑ | ↑ | ↑ | → | Mataloni (2016), Mataloni (2017), Epele et al. (2018), Mazzoni and Rabassa (2018), Morales et al. (2020) |
| Regulation of air quality | ? | ? | ? | | |
| Regulation of climate | → | ↑ | → | | Iturraspe (2016), Holl et al. (2019), León et al. (2021) |
| Regulation of freshwater quantity | ↑ | ↑ | → | → | Iturraspe (2016), Morales et al. (2020) |
| Regulation of freshwater quality | ↑ | ↑ | → | ↑ | Manzo et al. (2020b), Morales et al. (2020) |
| Formation and protection of soils | ↑ | → | → | → | Ciari (2009), Roig and Roig (2004) |
| Regulation of detrimental organisms | → | | | | Grech et al. (2019) |
| *Material* | | | | | |
| Energy (+) | | ↑ | → | | Rabassa et al. (1996), León et al. (2021) |
| Food and feed | ↑ | → | ↑ | → | Mazzoni and Rabassa (2018), Castillo et al. (2021) |
| Materials and assistance (+) | | ↑ | | | Rabassa et al. (1996), León et al. (2021) |
| Medicinal and genetic resources | → | → | ? | ? | Molares and Ladio (2014) |
| *Nonmaterial* | | | | | |
| Learning and inspiration | → | → | → | ? | Van Bellen et al. (2014), González Garraza et al. (2019), Epele et al. (2021b) |
| Experiences | ↑ | ↑ | ↑ | → | Pascual et al. (2009), Iturraspe (2016) |
| Supporting identities | ↑ | ↑ | → | ↑ | Iturraspe (2016), Ceballos et al. (2012), Morales et al. (2020), Epele et al. (2021a) |
| Maintenance of options | ↑ | ↑ | ↑ | ↑ | Epele et al. (2018), León et al. (2021) |

Adapted from IPBES (2019) and Brauman et al. (2020)
*M* mallines; *T* turberas; *L* lagunas; *V* vertientes
*Mallines excluding lagunas; →: low provision; ↑: high provision;?**:** not known; blank cells indicate that the service is not considered applicable to the wetland type. +: These potential contributions to people imply the draining and consequent destruction of the wetland

mallines seems to be not only a reliable classification scheme but also a good predictor of their aquatic diversity (Mazzoni and Rabassa 2013; Manzo et al. 2019).

Mallines from glacial plains, dominated by fine and homogeneous substrates, concave profiles, and receiving the highest levels of precipitations, are usually the most extensive and developed across the region (Fig. 10.4a) (Mazzoni and Rabassa 2013). The water of these mallines tends to exhibit circumneutral pH and low values of electric conductivity, alkalinity, and nutrients (Chimner et al. 2011; Epele et al. 2018). However, these patterns also depend on the aridity of

**Fig. 10.3** Patagonian wetland biodiversity. (**a**) *Keratella valdiviensis*, a rotifer from a turbera. (Photo by P. García); (**b**) *Coscoroba coscoroba*, from a semiarid Patagonia laguna. (Photo by E. Fernández Clark); (**c**) *Sphagnum* sp. and *Drosera* sp., typical plants from turberas. (Photo by L.B. Epele); (**d**) *Nannophryne variegata*, typical from turberas located in the Aysén Region, Chile. (Photo by E. Domínguez); (**e**) *Lynceus mallinensis*, from an ephemeral mallín located in semiarid Patagonia. (Photo by L.B. Epele); (**f**) *Frustulia yaganiana*, a diatom described from a turbera. (Photo by V. Casa); (**g**) Gomphidae dragonfly from a mallín located in humid Patagonia. (Photo by L.B. Epele); (**h**) *Hippuris* sp. a macrophyte from a mallín located in humid Patagonia. (Photo by L.B. Epele); (**i**) *Percichthys trucha*, a native fish typical from lagunas and some Patagonian rivers. (Photo by L.B. Epele)

the surrounding areas, with arid and semiarid ones leading to higher values of the abovementioned variables. Besides, glacial plain mallines support high cover and diversity of aquatic plants (alpine mallines excepted; Epele pers. obs.), and diversity of aquatic invertebrates (Manzo et al. 2019; Manzo et al. 2020a).

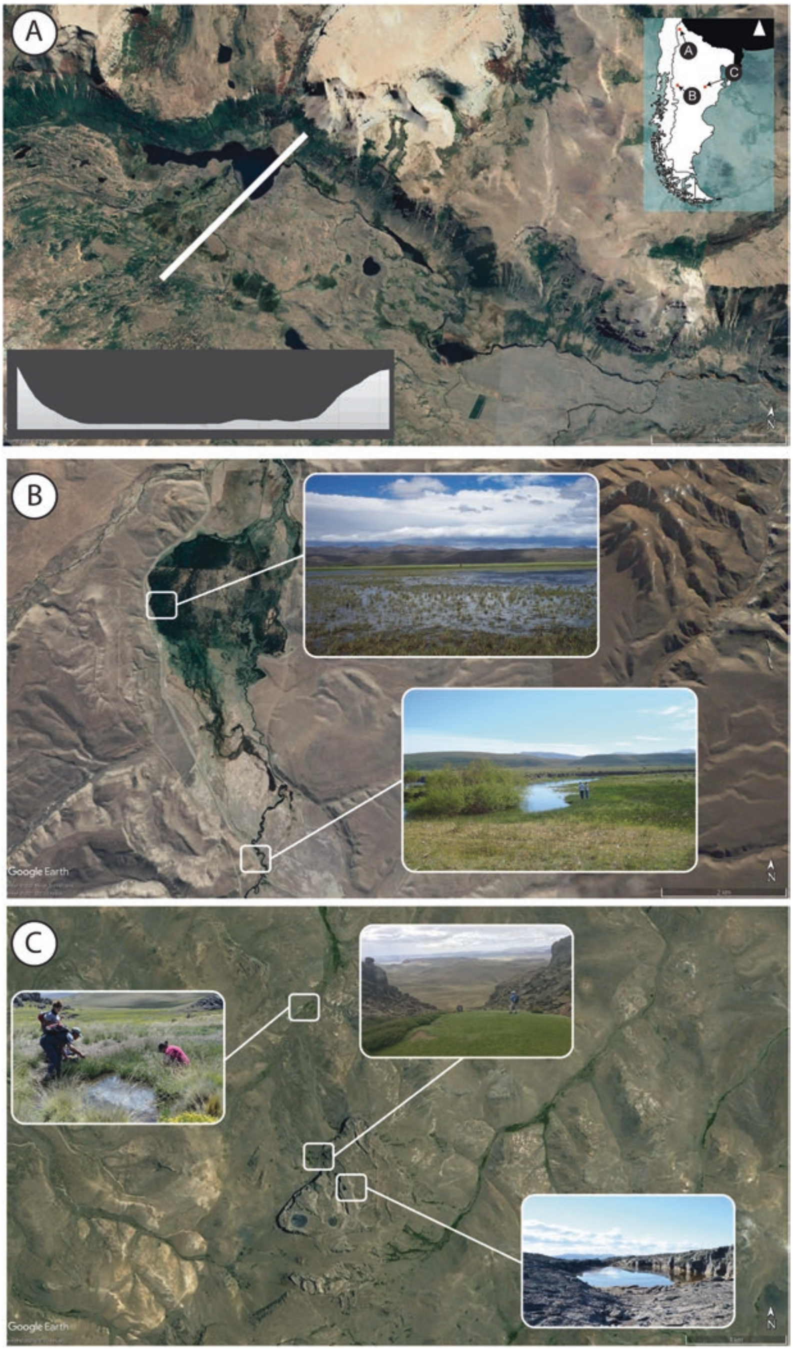

**Fig. 10.4** Three typical mallines from Patagonia. (**a**) Glacial plain mallín from Trollope Lake valley, showing its concave profile through the white line and the lower left profile representation; (**b**) mallín located in fluvioglacial and alluvial plains from Tecka river. (Upper photo by M.L. Miserendino and lower by L.B. Epele); (**c**) mallínes from Talagapa ranch that are fed by many vertientes (see left photo). Lower photo shows a high-altitude basaltic laguna. (Photos by L.B. Epele)

Mallines located in fluvioglacial and alluvial plains have a longitudinal development following the direction of the water course flow (i.e., streams or rivers) as well as a transverse gradient to the valley (Figs. 10.1g and 10.4b). Fluctuations in the water table usually depend on the regime of the lotic system but also on the microtopographic variations of the fluvial deposits (Mazzoni and Vazquez 2004). When compared with other mallines that are closely located but isolated from lotic systems, they usually exhibit lower nutrient levels and similar values of electric conductivity and pH (Epele et al. 2018, 2019). They support similar alpha diversity than their isolated counterparts (Kutschker et al. 2014), but as their hydrological connectivity is higher, they tend to support lower beta diversity (Epele et al. 2019).

Vertientes-fed mallines constitute a third type. These mallines are mainly located on the slopes of the basaltic plateaus, in the arid and semiarid Patagonia (Fig. 10.4c). Depending on the morphology of the slope and the water flow received, they acquire different shapes and sizes (Mazzoni and Rabassa 2018; Melendi et al. 2020). The surface of these mallines is usually relatively small (a few hectares), unless they receive the contribution of several vertientes reaching hundreds of hectares. These vertientes have a water quality good enough for human consumption, although organic pollution has been sometimes recorded (Morales et al. 2020). Their water has low electrical conductivity values ($< 1500\ \mu S\ cm^{-1}$) and neutral pH values (Melendi et al. 2020; Morales et al. 2020). Regarding its biodiversity, on southern Somuncurá plateau and montane Andes, Epele (unpublished data) has registered the presence of several groups of aquatic invertebrates (e.g., leeches, mollusks, amphipods, bugs, beetles, caddisflies, and nonbiting midges), as well as the presence of amphibians (see Chap. 15).

Mallines hydroperiod is also an important driver of water quality, with temporary ones exhibiting higher nutrients and alkalinity, but less aquatic plant cover than permanent (Epele et al. 2018). In a recent paper that identified priority areas for wetland biodiversity conservation, the authors found that wetlands with hydroperiods shorter than 3 months (i.e., wetlands that only hold water in the spring and early summer or after heavy rains) contributed to regional invertebrate diversity with rare species, whereas permanent ones contributed by increasing taxon richness in general (Epele et al. 2021a). However, these findings could depend on particular taxon patterns. For example, aquatic beetles exhibited similar diversities across mallines hydroperiods (Epele and Archangelsky 2012).

### *3.2 Turberas and Their Lagunas*

In Patagonian turberas, as well as in other types of wetlands, the chemistry of the water is a key factor driving the microbial and invertebrate diversity and the trophic web structure of stagnant waterbodies (Mataloni 2016 and cites therein, García et al. 2017). However, these relationships have been addressed at local scales, with regional studies still lacking. Some punctual studies such as of León et al. (2018) found significant relationships between surface-water chemistry and the composition of nonvascular flora of Chiloé peatlands. Nutrient inputs by sea spray are another crucial factor explaining the dominant bog vegetation type from Chile, but less frequent in Argentina (Kleinebecker et al. 2007; Rodríguez 2015). Like mallines, precipitation is another important predictor of peatland vegetation (Arroyo et al. 2005). For instance, in sites with annual precipitation, >2000 mm cushion plants such as *Astelia pumila* and *Donatia fascicularis* are dominant. Between 500 and 1500 mm, ombrotrophic bogs are dominant (Iturraspe 2016), although they have been also found in areas with higher annual precipitation values (Rodríguez 2015). One of the most used classification criteria is based on the dominant vegetation communities. Pisano (1977) describes mainly three groups: (i) raised bog communities, dominated by cushions of *Sphagnum magellanicum* moss; (ii) non-raised bog communities, or cushion plants bog dominated by *Donatia fascicularis* and *Astelia pumila*; and (iii) graminoid tundra communities dominated by species of *Carex*, *Schoenus*, and *Carpha*. As demonstrated by Savoretti (2018) for two turberas located in the SE Tierra del Fuego Island, changes in the dominant vegetation of these systems have tracked environmental changes that occurred since the last glacial maximum, confirming their value as reliable indicators of climate change.

The biodiversity of Patagonian turberas is very specialized (Mataloni 2017, and references therein). High endemism levels have been recognized for bryophyte and lichen species (Díaz et al. 2008; León et al. 2014; Souto et al. 2015) but also for protists such as ciliates (Küppers et al. 2016), testaceans (Burdman 2019), and diatoms. For the latter group, a large number of species have been described from two *Sphagnum*-dominated ombrotrophic bogs (Casa et al. 2017, 2018, 2019). Therefore, it should be noted that the loss of species at a local level would imply its extinction at a global scale (León et al. 2021).

### 3.3 Lagunas from Basaltic Plateaus and Endorheic Depressions

As well as lagunas from humid Patagonia are usually part of mallines and turberas, those of arid and semiarid zones are usually the endorheic depressions that harbor permanent and temporary lagunas (Fig. 10.1f). There are countless endorheic depressions (i.e., >1000 just in the Strobel Lake plateau; Izaguirre et al. 2018) of various sizes and shapes, whose distribution and morphology are related to the age and lithology of the geological formations in which they are located (Mazzoni 2001; see Chap. 5). In most cases, fluctuations in their level are associated with the precipitation/evaporation ratio, while they rarely receive contributions from underground or permanent drainage lines (Mazzoni and Rabassa 2018). In general, these lagunas exhibit alkaline waters that can reach pH values higher than 11 (Epele et al. 2018; Melendi et al. 2020) and from moderate (around 1000 μS $cm^{-1}$) to high electric conductivity (even more than 30 mS $cm^{-1}$) due to their endorheic nature (Ramos et al. 2022). As other wetlands from the region, the limiting nutrient is commonly nitrogen. The levels of water nutrients are not particularly high (Epele et al. 2018;

Izaguirre et al. 2018), although values that equal those found by Manzo et al. (2020b) in wastewater treatment plants (>25 mg $l^{-1}$) were found by Epele and Grech (pers. comm.) in lagunas from southern Somuncurá basaltic plateau that were not altered by any land use. The biodiversity of these waterbodies has been extensively documented, standing out the presence of different species of microorganisms, plants, macroinvertebrates, amphibians, and birds (Lancelotti 2009; Perotti et al. 2005; Porcel et al. 2019; Manzo et al. 2020a, Epele et al. 2021a) (see Chap. 15). Moreover, these lagunas are the habitat of a particular crustaceans called ostracods that have been widely used for paleoenvironmental reconstructions (e.g., Coviaga et al. 2018).

## 4 Threats to Wetland Biodiversity and Functioning

In the Anthropocene, global freshwater biodiversity is disproportionally threatened by multiple stressors (Reid et al. 2019), including direct exploitation and environmental change (e.g., climate change). Compared with other temperate regions that are subjected to water drainage (Zedler 2003), intensive agriculture (Gleason et al. 2008) or cattle rising (Sica et al. 2018), extensive urban developments (Gledhill et al. 2008), or simultaneous changes in land use (Aquino et al. 2021), Patagonian environments are relatively well conserved (Rosas et al. 2021; Peri et al. 2021), and the wetlands are not the exception (Epele et al. 2018). However, after more than a century of regional extensive livestock farming (Collantes and Faggi 1999), strong local effects of urbanizations, and interactions among them and with other stressors, Patagonian wetland biodiversity is threatened, as well as their function and contributions to people (see Chaps. 11 and 15).

### *4.1 Livestock Breeding and Agriculture*

Across Patagonia, livestock breeding is not only the most frequent and distributed land use but also the oldest threat to wetland biodiversity since Europeans colonized the region between 1880 and 1920. Current management is based on extreme overuse in space and time, inducing profound changes in vegetation structure and increasing soil fragility. Through trampling, livestock reduces the plant cover and produces soil compaction, leading to denudation with consequent erosion in extreme situations (Gaitán et al. 2014; Motta et al. 2020). Livestock land use (Fig. 10.5a, b) can compromise wetland water quality, increasing sedimentation rates (i.e., trampling) and nutrient inputs (i.e., manure deposition) (Epele and Miserendino 2015). However, wetland responses would depend on the aridity of their locations. Thus, wetland water quality and biodiversity seem to be more vulnerable in arid and semiarid environments than in humid and subhumid zones (Kutschker et al. 2014; Epele and Miserendino 2015; Epele et al. 2018; Motta et al. 2020). Dryland wetlands have

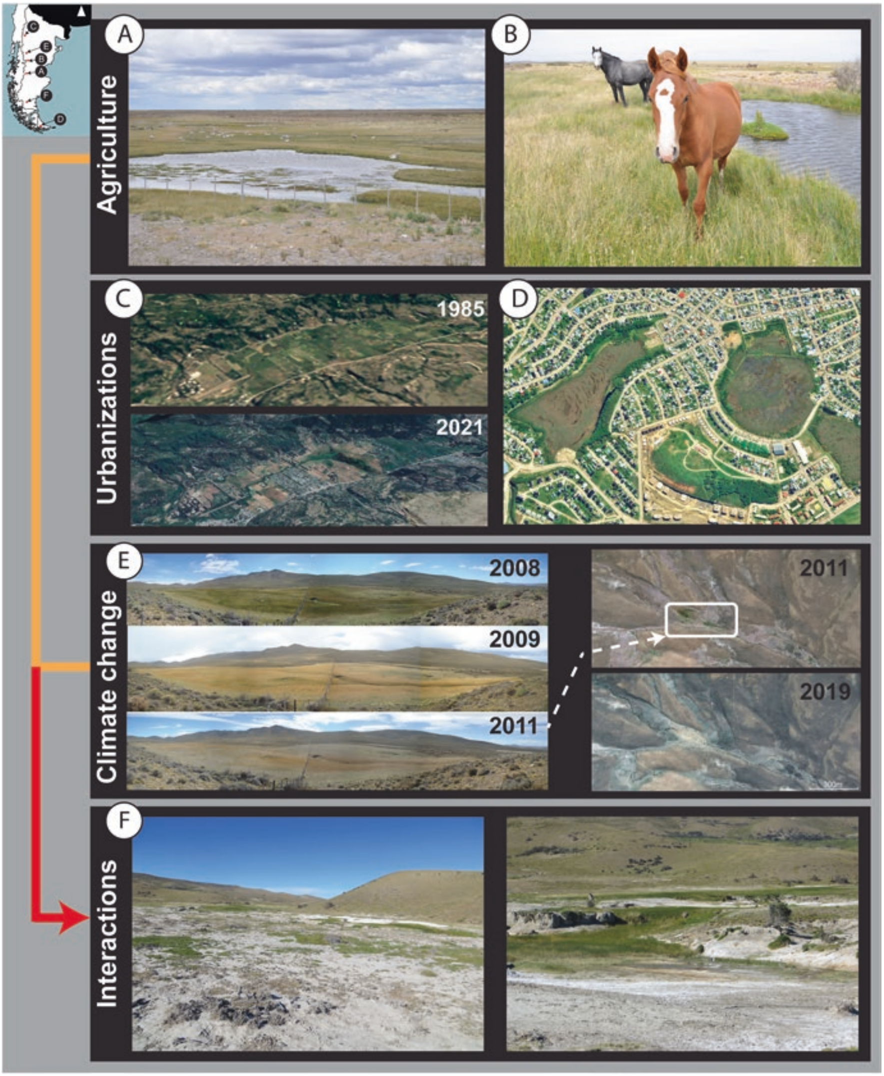

**Fig. 10.5** Patagonian wetlands threatened by different stressors: (**a**) and (**b**) livestock land use as an example of agricultural impacts. (Photos by L.B. Epele); (**c**) and (**d**) urbanizations threatening a mallín (from San Martín de los Andes) and turberas (from Ushuaia); (**e**) effects of droughts on Crespo mallín (Gobernador Costa). In 2019, the mallín has practically disappeared. (Photos by L.B. Epele); (**f**) interacting effects of climate change (i.e., higher temperatures and evapotranspiration and lower precipitation) and overgrazing leading to the fragmentation and loss of the mallines. (Photos by E. Mazzoni)

been the main providers of water supply and forage for livestock breeding. Consequently, these rangelands have been overexploited, reaching a point in which recovering their livestock and rangeland productivity would take decades (Utrilla et al. 2005) (see Chap. 11).

Other anthropic activities as wetland drainage for agricultural purposes (e.g., mostly fruticulture and horticulture production) occur mainly at the floodplains of larger rivers of northern Patagonia (e.g., Chubut and Negro) but have also been observed at certain localities on the Andes mountain valleys (Miserendino 2009). In intensively developed areas of northwest Patagonia, the application of fertilizers and pesticides is also associated with these agricultural activities, indirectly altering the water quality of surrounding wetlands (Macchi et al. 2018) (Chaps. 9 and 11).

## *4.2 Urbanizations*

This is one of the most important stressors causing wetland loss and species extinction (Holtmann et al. 2019). Urban development generates wetland drainage, decreases vegetation cover, and increases physical habitat degradation, sedimentation, hydrographic alterations, temperature oscillations, increasing contaminant, and nutrient concentrations (Patenaude et al. 2015). The most obvious impact is the loss of biodiversity and wetland functioning through reductions of the natural wetland area. Despite the low density of inhabitants per km$^2$, most studies in Patagonia have found that urbanizations decrease wetland water quality. In a regional approach, Epele et al. (2018) recorded strong relationships between urbanization cover and nutrient levels in wetland water. Similarly, Macchi (2018) found higher water conductivity and total phosphorus at urban wetlands than at reference ones (San Carlos de Bariloche, Argentina) (see Chap. 11).

There is no doubt that wetlands can play a crucial role in the urban water cycle (Kampf and van den Voomen 2013). For this reason, many Patagonian towns (e.g., Esquel, El Bolsón) are using constructed wetlands as solutions for wastewater treatment and flood control (Manzo et al. 2020b). Nevertheless, the accelerated urban expansion is a concerning problem in certain areas (Iturraspe et al. 2021). For example, in San Martín de los Andes (northern Patagonia), urbanization expanded over the mallín of Lacar Lake, draining more than 320 ha (Fig. 10.5c), and in Ushuaia (southern Patagonia) (Fig. 10.5d), this expansion produced the destruction of several turberas (e.g., during the 1990s decade, an important peat bog complex located in the geographic center was drained). Moreover, an increasing number of private buildings with construction of drain ditches and no wastewater treatment are irregularly occupying public lands on the boundaries or within protected peatlands such as Rancho Hambre, with an uncertain impact on their diversity and ecosystem functioning (see Chap. 6). These rapid increases in the impervious cover (e.g., roads, pavement, and residential developments) can reduce the water infiltration eluding wetland groundwater replenishment that in turn alters their hydroperiod and ultimately their existence (Chaps. 9 and 11).

### 4.3 Invasive Species

Patagonian wetlands are particularly susceptible to anthropogenic disturbances that generate favorable conditions for the establishment of exotic species (Raffaele 2004; Domínguez et al. 2020; León et al. 2021). Some species introduced hundreds of years ago, as, for example, all livestock species treated before, produced very profound changes in the wetland ecosystems of Patagonia. Other invasive vertebrates that altered wetland biodiversity and functioning include the ecosystem-engineering American beaver (*Castor canadensis*) (Huertas Herrera et al. 2020, and cites therein) and the American mink (*Neovison vison*) (Grootjans et al. 2014), as well as exotic fish introduced in lagunas (e.g., trouts; see Chaps. 13 and 14). There are just a few studies regarding invasive aquatic invertebrates. For example, the invasive snail *Physa acuta* is becoming more frequent in the region (Brand et al. 2018, see Chap. 11). In particular, this species appears very successful inhabiting anthropogenic landscapes (e.g., constructed wetlands, urban ponds, etc.) (Macchi et al. 2018; Manzo et al. 2020a). Another example is the introduced snail from the *Lymnaea* genera, which inhabits wetland margins. It is an intermediate host of the parasitic trematoda *Fasciola hepatica*, whose final hosts are sheep, cows, and humans, and causes the common liver fluke.

Wetlands are also subjected to the invasion of exotic plant species. Their relative contribution in terms of taxonomic richness can be quite high across the region (i.e., higher than 20% per wetland) and frequent across wetlands (i.e., 50–80% of localities recording their presence) (Gaitán et al. 2011; Kutschker et al. 2014; Manzo et al. 2020a). Cuassolo et al. (2012) also found that invasive aquatic plants could also modify the stoichiometry of the wetlands (see Chap. 11).

With regard to invasive microorganisms, the periphytic diatom *Didymosphenia geminata* was first detected in rivers from the Andean section of Patagonia as of 2010 and presently is expanding not only its geographic distribution but also its range of ecological preferences (Chaps. 11 and 12), thus posing a potential risk to oligotrophic vertientes with porous rock substrates.

### 4.4 Human Threats to Turberas: Peat Extraction, Drainage, and Tourism

In the Chilean Magallanes Region, historically there were eight turberas under exploitation, with a total area of 444 ha, approximately 10% of the exploited area in the Argentinean Tierra del Fuego (Cardone and Worman 2008). At present, though, only two projects for peat extraction have been approved by the National Environmental Impact Assessment System (SEIA), involving just 178 ha, about 40% of the historically exploited area in the region. Using peaty soils for agricultural activities requires draining and fertilizing. Nevertheless, according to the current Chilean legislation regarding the protection of *Sphagnum* (Supreme Decree

N°25 of the Agriculture Ministry) and that of the environment (Supreme Decree N°40 of the General Environmental Law), such practices are deemed illegal, since they turn turberas from $CO_2$ sinks to sources. Contrarily to emissions associated with forest clearance, those from drained peatlands continue for as long as the peatland remains drained. For this reason, peat bog restoration through rewetting is the keystone of Scotland's actions to reach net-zero greenhouse gas emissions by 2045 (Gewin 2020). Though a model developed by Humpenöder et al. (2020) recognizes that both restoration and conservation measures should be tackled to transform the global land system to a net C sink by 2100, all in all, conservation of turberas in their pristine state is the best cost-effective way to prevent emissions rise (Joosten et al. 2012). The Land Use Planning Resolution 401/2011 issued by the Secretary for Sustainable Development and Environment of Tierra del Fuego Province (SDSyA-TDF) protects most turberas from peat mining and, in turn, determines a "sacrifice area" for peat extraction under regulated conditions. Yet, increasing recreational activities in Tierra del Fuego peatlands triggers local degradation in trekking trails and along the coast of Peninsula Mitre (Iturraspe, personal obs). To ensure conservation of such extremely fragile wetland ecosystems, tourism and recreation should be carefully planned and controlled (Inostroza Pino 2008).

## *4.5 Climate Change*

Climate change predictions for the region include increasing water temperatures and a reduction in the annual precipitation (Barros et al. 2015; see Chap. 1). These changes are currently impacting wetlands through the alteration of precipitation regimes, snow accumulation, lagunas levels, water table depths, and vegetation productivity (Izaguirre et al. 2018; Dussaillant et al. 2019; Lancelotti et al. 2020; Pessacg et al. 2020; Castillo et al. 2021). Moreover, Patagonian studies indicate that rising temperatures could alter wetland species distribution, produce disease outbreaks, and change in their phenology and survival (Jara and Perotti 2018; Grech et al. 2019; Jara 2019).

In the short term, the effects of climate change will strongly affect the arid and semiarid Patagonia. For example, Crego et al. (2013) predicted significant reductions in the distribution of areas suitable for arid and semiarid mallines. Likewise, Epele et al. (2021a) after identifying priority wetlands for biodiversity conservation found that more than 35% of high priority wetlands (i.e., environmental resources for aquatic and wetland-specific biodiversity as well as other non-wetland flora and fauna species) are predicted to disappear by 2050. Despite being less threatened, wetlands from humid and subhumid Patagonia (e.g., turberas) are being altered by the same changes in climate, with a stronger impairment in $CO_2$ uptake in raised *Sphagnum*-dominated bogs than in cushion bogs dominated by vascular plants (Holl et al. 2019). Climate change could also lead to significant alterations in their microbiota (Chap. 6) and eventually to mire degradation with the consequent release of greenhouse gasses into the atmosphere (León et al. 2021).

### 4.6 Interactions Among Threats

Up to this point, we have detailed the effects of multiple stressors on Patagonian wetland biodiversity, and despite analyzing their individual effects on wetlands, most of them can and do interact yielding synergistic effects at local and regional scales. For example, the wild board (*Sus scrofa*) produces stronger impacts on wetlands when coexisting with other typical livestock animals (i.e., cattle and horses) (Motta et al. 2020). Moreover, all stressors (i.e., agriculture, urbanizations, beaver damming) would interact with regional climate change. This could generate synergistic effects between them but also expand the scale of the negative effects produced by water-consuming land uses (i.e., oil and mineral extraction; see Chap. 11) and thus impact the functioning of wetlands (e.g., avoiding groundwater recharge). However, the paramount example of multiple stressors interactions in Patagonia is one of the main socio-ecological problems produced by human activities (del Valle et al. 1998; Andrade 2013): the desertification (see Chap. 11). More than a century of unmanaged (or wrongly managed) livestock breeding (mainly sheep) has resulted in a desertification process, forcing the landowners to move from extensive livestock practices to the intensive use of more productive areas such as mallines pastures (Perotti et al. 2005; Mazzoni and Vazquez 2009). This process is being enhanced by the severe regional drought generated by climate change. In the context of rural communities from arid and semiarid environments, water shortages and reduction of agricultural production are of main concern (Solano-Hernandez et al. 2020), altering not only the wetland contributions to people (e.g., pasturelands; Daza et al. 2019; Castillo et al. 2021) but also their habitat availability for local biodiversity (e.g., aquatic invertebrates, amphibians, waterbirds, and fish; Sandi et al. 2020; Epele et al. 2021a; see Chaps. 13 and 15). The long-term ecological responses of biological communities to the aforementioned stressors are difficult to predict but would probably lead to species distribution shifts (e.g., moving to west), phenological changes, and/or local extirpations. The future of wetland biodiversity would also depend on their spatial connectivity, the ability of species to colonize new habitats (e.g., dispersing against prevailing westerly winds, Epele et al. 2021b), and management decisions of land use.

In summary, wetlands from Patagonia are threatened by the rise in exploitation of its natural resources (i.e., the demand for land for agriculture and livestock breeding), accompanied by the rapid development of infrastructure, severe changes in climate (e.g., intense droughts), and a low percentage of effectively conserved wetlands).

## 5 Wetland Conservation Status

Among freshwater ecosystems, wetlands are probably in the highest need of conservation (Reis et al. 2017). As we mentioned, Patagonian wetlands host a large and specific biodiversity and provide numerous natural contributions to people that support the regional economy. However, the Patagonian conservation status is still far from reaching global conservation targets. Many national protected areas include wetlands, but just a few ones were specifically created to conserve wetlands, including Patagonia National Park (Argentina), Laguna Blanca National Park (Argentina), Laguna Parrilla National Reserve (Chile), and Magallanes National Reserve (Chile). Furthermore, as most Argentinian and Chilean protected areas are located in humid zones, the situation is critical for wetlands located in arid and semiarid Patagonia.

The conservation status of Patagonia is highly dependent on the policies of each country's government (Argentina and Chile); thus, for this section, the conservation status is described separately.

## *5.1 Argentinian Patagonia*

Most National Parks were created to conserve habitats located in the Andes mountains, including glaciers, lakes, many headwater basins, and terrestrial forests species (i.e., *Fitzroya cupressoides* in Los Alerces National Park) (Martín and Chehébar 2001). Despite this, these protected areas are appropriate for the conservation of a priori nontarget wetland species (e.g., aquatic invertebrates, Epele et al. 2021a); they tend to underrepresent wetland distribution across the region. For example, in the Argentinean sector of Tierra del Fuego, only a small fraction of the turberas area is located within provincial reserves, the Ramsar Site No. 1886 (Glaciar Vinciguerra y Turberas Asociadas), and the Tierra del Fuego National Park. In a country, where non-protected wetland land use is unregulated, and management actions are scarce or do not exist at all, the future of their biodiversity is highly dependent on landowners' decisions. Those decisions are in general governed by natural (e.g., in a dry year many sheep can die, decreasing the density per hectare) and economic changes (e.g., sheep stocks depend on the international price of the wool). Additionally, wetlands currently protected might not need immediate conservation actions. They are located in humid areas of Patagonia, where climate change would be less harsh than in the prevalent dry areas. However, since these wetlands contain high percentages of rare species, and protected areas are not completely exempt from threats (Maiorano et al. 2015), they should be managed adequately and monitored.

The lack of protected areas is evident in the arid and semiarid Argentinean Patagonia, which constitutes most of the regional area, but has less than 3% of their wetlands under protection (Crego et al. 2014). An exception is the Patagonia National Park (Argentina), which has been established to preserve basaltic plateau wetlands (i.e., lagunas), and their endemic and endangered species (i.e., *Podiceps gallardoi*; see Chap. 15). Epele et al. (2021a) suggested that the current network of protected areas in Patagonia should be expanded to adequately preserve wetland biodiversity. Nevertheless, the available economic resources for the creation of protected areas are scarce (Morea 2014) and deemed to diminish due to increasing inflation. A large number of projects for a National Law of Minimum Budgets for the conservation of wetlands have been discussed – even simultaneously – over the last decade (2013, 2015, 2020/21), and lost parliamentary status in circumstances not very different from those characterizing the debate around the Water Law in Tierra del Fuego Province (see Chap. 17). Furthermore, a national law on its own does not guarantee wetland conservation. Rather, current and projected climate change impacts demand concerted management and conservation actions based on reliable studies (i.e., long-term wetland monitoring) to halt wetland deterioration and promote their recovery. As of March 2021, The Ministry of Environment and Sustainable Development (MAyDS) created the Wetlands Programme, which contemplates the necessary funding to advance the national wetland inventory and to assist planning and management of the Ramsar sites network, among other actions. Within this framework, technical collaboration agreements have been signed with Tierra del Fuego and Río Negro Provinces. Future bold action in this direction is paramount in order to improve the conservation status of the Patagonian wetlands in the coming years.

## *5.2 Chilean Patagonia*

In Chile, conservation of wetland freshwater ecosystems is still far from appropriate. For example, although a large number of protected areas, including three Ramsar sites (i.e., Humedales de Monkul, Carlos Anwandter Sanctuary, and Bahía Lomas), have been created to protect regional biodiversity, none of the abovementioned sites were created to protect the ecosystems dealt with in this chapter but rather aimed at maintaining the diversity of coastal wetlands. Furthermore, although the Chilean National Plan for Wetlands Protection (2018) recognized more than 40,000 wetlands across the country, this inventory did not include mallines and peatlands. Turberas are nevertheless protected through three regulatory frameworks. First, peat is considered a concessional fossil substance, and its exploitation is regulated by the Mining Code (Law 18,248). Second, the Supreme Decree N° 25 of the Ministry of Agriculture (2017) establishes protection measures for *Sphagnum magellanicum* moss seeking to regulate their harvest and prohibiting turberas drainage. Third, there are two regulation projects currently in the pipeline: Project 12017-12 (Cámara de Diputadas y Diputados 2021) seeks to generate an exception in peat mining concessions and subject both peat extraction and live moss collection to the SEIA, and Project 11672-12 (Cámara de Diputados 2018) prohibits the extraction of material from the turberas of Chiloé Province. In addition, a recent regulation defines the minimum criteria for the sustainability and management of urban wetlands (Chilean Law No. 21202; Environment Ministry 2020).

In summary, it is of prime importance that future national (or regional) inventories include not only a detailed account of the different types of freshwater wetlands but also their latitudinal and altitudinal distribution in order to improve their management and conservation.

## 6 Knowledge Gaps

Throughout this chapter, we have only been able to detail a small portion of the vast knowledge generated for the freshwater wetlands of Patagonia. However, there are many gaps in knowledge of wetland distribution, biodiversity, and functioning. Some are here summarized in order to make recommendations for future research and guide government decisions for the allocation of economic resources.

Plans of regional conservation of wetlands are urgently needed, mainly for arid and semiarid Patagonia. To efficiently allocate limited resources and ensure the conservation of wetland biodiversity and their contributions to people, priority areas should be identified including wetland water quality and hydroperiod, landscape connectivity, their resilience, current and potential anthropogenic threats, and models of future species distributions in the context of global environmental change.

Moreover, regional wetland inventories are currently recognized as essential tools for designing management and conservation policies for their wise use (Zedler and Kercher 2005; Kandus and Minotti 2018). In Argentinian Patagonia, wetland inventory and characterization have proved difficult on account of its very large area (i.e., 1,000,000 km$^2$) and wide environmental and geographic heterogeneity (Ayesa et al. 1999; Mazzoni and Vazquez 2004; Benzaquén et al. 2017, among others). For this reason, its execution depends on government decisions at national and provincial levels to allocate human and financial resources to this much-needed task.

Finally, in addition to particular national efforts, it would be desirable to strengthen and encourage the international partnership between Chile and Argentina, jointly addressing regional scientific and management tasks that tend to resolve the peculiarities of the Patagonian wetlands, as well as their relationships with human well-beings, geography, and climate. An international Patagonian wetland monitoring program considering their spatial and temporal variability would be a key strategy in this direction.

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# Chapter 11
# Land-Use Effects on Aquatic and Wetland Ecosystems: An Overview of Environmental Impacts and Tools for Ecological Assessment

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## 1 Introduction

Unlike in many other territories, the colonization process in Patagonia is relatively recent; however, important human-induced changes have occurred since the arrival of the first settlers. With regard to aquatic resources, there has been frequent irresponsible management driven by the idea that the supply of water is unlimited. Despite their low population density (2.5 inh km$^{-2}$), some areas in Patagonia are seeing an important increase in population, at a higher rate than many other regions in Argentina (INDEC 2010). This expansion is promoting an intensification of the land use (Morello et al. 2018; Nanni et al. 2020) and is having significant environmental impacts on the region's aquatic resources.

According to Morello et al. (2018), three different ecoregions can be distinguished in the vast Patagonian territory (Fig. 11.1). A narrow strip of forests over and along the Andes and at the headwaters of all Patagonian basins forms the Patagonian Forest ecoregion. It is extended on the occidental part of Neuquén, Río Negro, Chubut, Santa Cruz province, and the austral strip of Tierra del Fuego, Antarctic, and South Atlantic Islands. The ecoregion also exhibits a wide proportion

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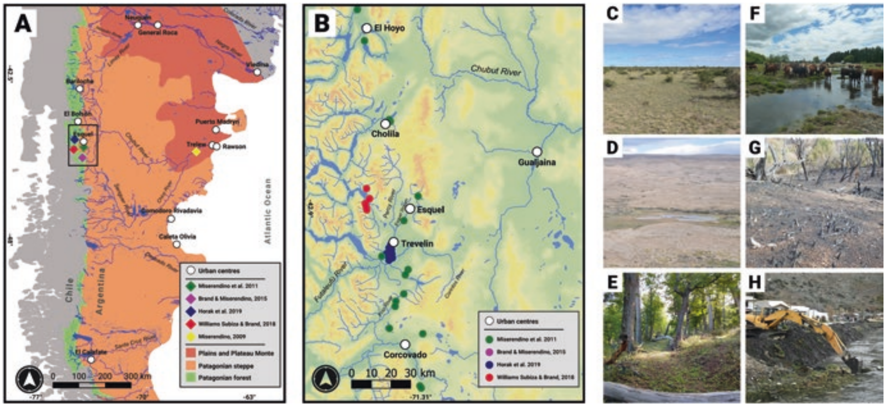

**Fig. 11.1** (**a**) Patagonian ecoregions (Sensu Morello et al. 2018), main drainage systems and urban centers. (**a**, **b**) Locations of studied sites mentioned in the scientific works compiled in this chapter. Pictures of landscapes of the Patagonian ecoregions: (**c**) Monte, (**d**) Steppe, and (**e**) Andean forest. Examples of different land uses and disturbances: (**f**) livestock breeding, (**g**) forest fires, and (**h**) urbanizations

of shrubs, rocks, and continental ices. The temperate semi-deciduous rainforest dominates, with species changing with elevation, location (exposure), and latitude. The Patagonian Steppe ecoregion is a vast area that comprises Southwestern of Mendoza, west of Neuquén and Río Negro, most of Chubut and Santa Cruz and north of Tierra del Fuego. The main vegetation forms are shrub-grass species mostly adapted to extreme aridity conditions, low temperature, strong winds, and frequent frosts. It is composed of short shrubs, with cushion forms, prickly, with reduced leaf or even leafless. The herbaceous steppe with xerophytic grasses appears in a lower proportion. Wetlands are conspicuous in the continental sector and include meadows, marshes, bogs, forests, and shrubs (see Chap. 10). The Plains and Plateaus Monte ecoregion spreads from the Andes Cordillera to the Atlantic Ocean and includes great part of Mendoza, Neuquén, La Pampa, and a little portion of Río Negro and Chubut provinces. The landscape is dominated by plains and staggered plateaus wherein aridity is a prevailing feature (see details in Chap. 1).

The main land-use practices in Patagonia include extensive livestock breeding, pasture conversion, forestry, pine plantations, mining and oil extraction, and to a lesser extent urbanization and industrial development. These practices have led to major ecosystem stressors, defined as any physical or chemical change promoted by anthropogenic action that causes significant modifications in biological components, patterns, and relationships in natural systems (Sundermann et al. 2013). According to Dudgeon et al. (2006), five major threats or stressor categories have the potential to impact on freshwater biodiversity: overexploitation, water pollution, flow modification, habitat degradation, and species invasion. However, new threats emerged during the last few decades, with some of them disproportionately impacting freshwater systems (Reid et al. 2019). The consequences or impacts of these stressors are quantifiable through the assessment of the attributes of biological components. Among organisms that respond consistently to different stressors are periphyton, macrophytes, and macroinvertebrates (Allan et al. 2021).

Biodiversity in Patagonia is globally significant due to the region's pronounced endemism, habitat heterogeneity, and biogeographic location (Morrone 2006). There is strong concern from a conservation standpoint, since both ongoing and emerging land-use practices are rapidly changing the ecological status of freshwater resources in the region.

## 2 Land-Use Effects on Aquatic Environments

### *2.1 Agricultural Practices*

#### Pasture Conversion and Livestock Breeding

The transformation of forest into pasture for the extensive grazing of livestock has been a generalized practice in Patagonia mountains since the 1960s. Several forested areas have been burned to accelerate this land-use change. In earlier times, these practices were attributed to the region's indigenous inhabitants, who used them as a strategy for the hunting of *Lama guanicoe* (guanaco) (Carabelli and Scoz 2008). This landscape modification has occurred on both the Patagonian mountains and valleys (Subantarctic forest), which are characterized by *Nothofagus* sp. forest.

The loss of vegetation in a basin can result in profound modifications of the environmental features of lotic systems. Changes in water temperature, primary productivity, sediment inputs, and nutrient enrichment, as well as alterations in stream hydrology and habitat structure, are examples of the effects that pasture conversion may exert on stream environments (Allan et al. 2021). Additionally, natural riparian cover also reduces the progressive erosion of the catchment and, at a local scale, of the stream bank during floods (Magner et al. 2008). Erosion is extremely detrimental for the biota; it is linked to the entrainment and deposition of inert fine particles (Larsen et al. 2011), an increase in turbidity, and a reduction in primary production. It also increases the amount of light available for visual predators and changes the substratum structure and benthic habitat by filling interstices. This phenomenon has been strongly associated with the decrease in richness and density of several sensitive taxa (Buendia et al. 2013).

Brand and Miserendino (2015) compared streams into pasture lands and streams in nonimpacted native-forested catchments. They found strong evidence of sediment deposition in streams draining pastures and also increased levels of suspended solids. In addition, these sites had significantly higher water temperature and conductivity values, as well as lower dissolved oxygen concentrations than forested sites. Other authors also detected profound changes in the riparian ecosystem (Kutschker et al. 2009) and in the in-stream habitat condition, in contrast with other land-use categories (Fig. 11.2).

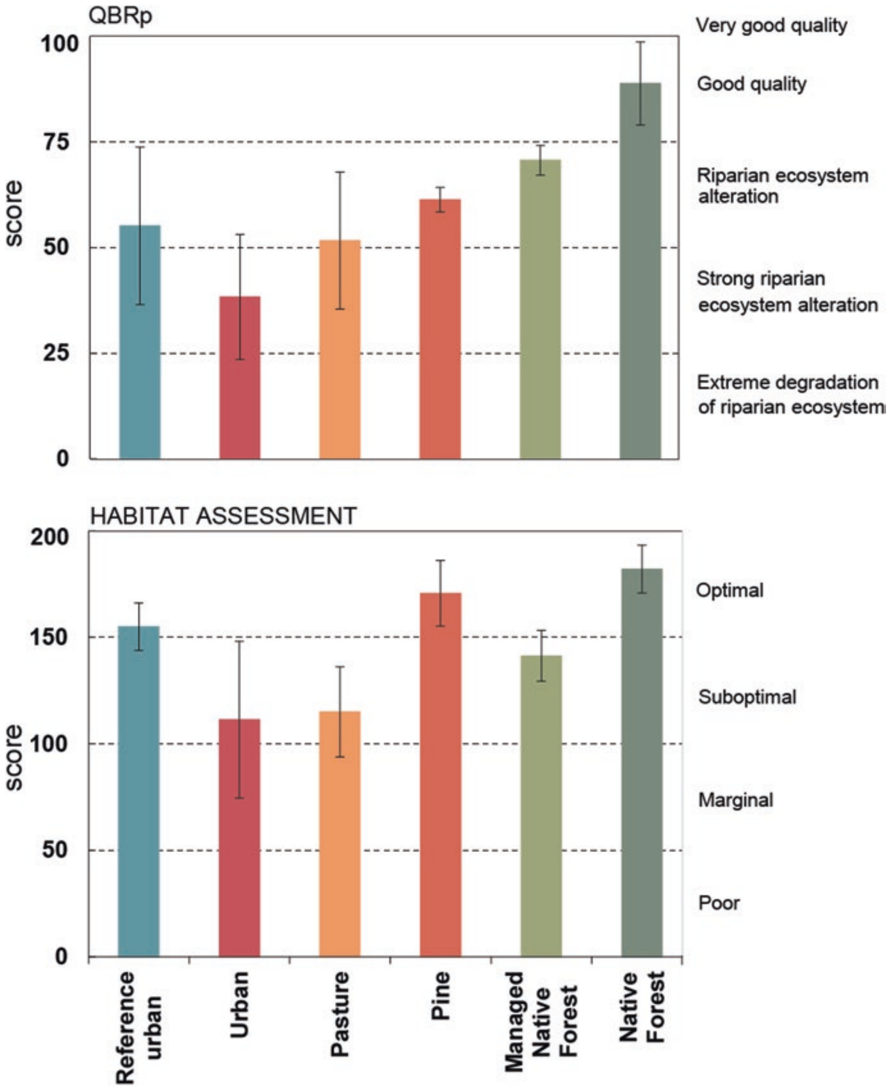

**Fig. 11.2** Riparian ecosystem condition (QBRp) (upper) and habitat condition index score (lower) per land use (*n* = 3). Data obtained from 18 mountain streams in Northwest Chubut, during February 2005. Mean values (±SD) are consigned. (Adapted from Miserendino et al. 2011)

Responses in macroinvertebrate communities were significant, and several macroinvertebrate metrics proved useful in assessing disturbance. Of the metrics studied, Shannon diversity, Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness; Plecoptera richness; percentage of EPT taxa; percentage of dominant taxon; shredder richness; and percentage of predators proved to be the most consistent measures and displayed the strongest responses to impairment (Fig. 11.3). Both studies

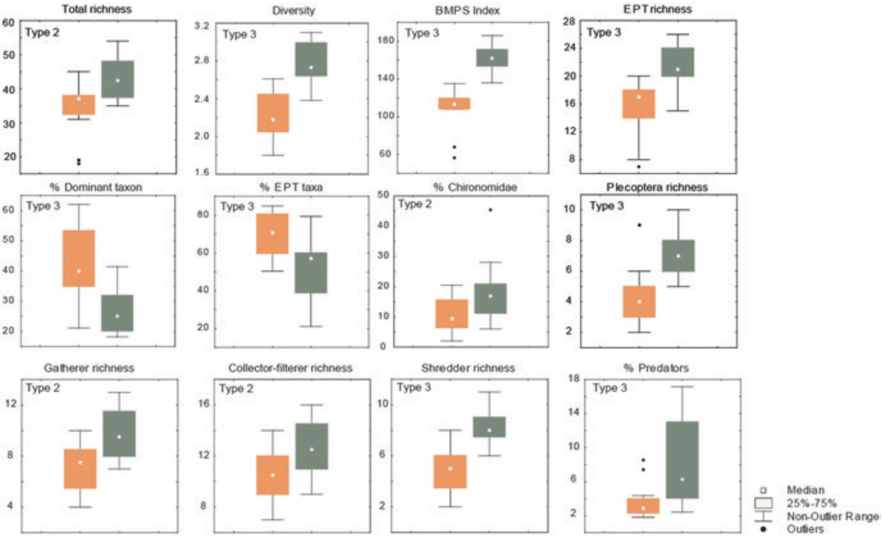

**Fig. 11.3** Distribution of values of selected metrics based on macroinvertebrate attributes at pasture (black bars) and reference sites (white bars) from six Patagonian mountain streams. Range bars show non-outlier range, boxes represent interquartile ranges (25–75%), small squares represent median values, the sensitivity type 3 (very strong: if no overlap exist in the interquartile range) or 2 (strong: if there is some overlap that do not extend to the medians) for each metric is indicated in each figure. Metrics based on richness, density, composition, and functional groups are included. (Adapted from Brand and Miserendino 2015)

(Brand and Miserendino 2015; Kutschker et al. 2009) agree that the conversion of forest into pasture for livestock decreases macroinvertebrate biodiversity, potentially changing the functioning and dynamics of Patagonian mountain streams.

A different situation appears on the Patagonian steppe ecoregion, which is dominated by grasslands, shrub-grass steppes, shrublands, and semideserts (Morello et al. 2018). On the steppe, overgrazing and agricultural practices have increased the desertification process greatly, and vast areas have been affected (Gaitán et al. 2014). The extent and severity of desertification has been properly quantified and mapped (del Valle et al. 1998; Mazzoni and Vázquez 2009). According to del Valle (1998), the main environmental and ecological problems associated with desertification are vegetation degradation, wind and water erosion, salinization and alkalization, soil crusting, and compaction; all of which have known negative impacts on aquatic ecosystems and wetlands (Rasmussen et al. 2013; see also Chap. 10).

Miserendino (2004) assessed the effect of landscape configuration and desertification process on macroinvertebrate assemblages in rivers from six important basins in the Patagonian mountains and steppes. Although the best predictors for the composition of macroinvertebrate assemblages were latitude and variables associated with river size, the desertification status negatively affected diversity (H´) and the richness of EPT groups. Rivers located in forested areas with a low degree of desertification contained communities with a higher diversity than those in areas with moderate or severe desertification. However, some sites located in basins within a grassland matrix showed severe desertification and intermediate diversity values, probably as a result of having nearby strips or patches of native riparian shrubs, which mitigated the negative impact of desertification. These patches probably buffered nutrient inputs, supplied allochthonous organic matter, improved in-stream habitat, and provided habitats for the adult terrestrial stages of insects with stream-dwelling larvae. The preservation or restoration of patches of riparian vegetation could be a sound mitigation measure for the conservation of freshwater resources in this ecoregion.

The impact of different grazing intensities on communities in wetlands has also been assessed. In a study conducted in 30 wetlands, including connected and isolated ponds (northwestern Patagonia), Kutschker et al. (2014) documented the consequences of different livestock densities (low, medium, and high) on hydrophytic plant assemblages. The approach also considered specifics such as land carrying capacity scores in the area. The main findings were that macrophyte assemblages consistently responded to a gradient of disturbance represented by increasing salinity and ammonium that was associated with cattle presence (urine and feces). There was also a correspondence between community composition and the intensity of grazing pressure, mostly at isolated wetlands, with total species richness, richness of natives, and hydrophytic plant coverage significantly decreasing at the most disturbed sites. The study also identified three species (*Distichlis spicata*, *Xanthium spicatum*, and *Eleocharis melomphala*) as indicators of wetland deterioration, since they were recorded in ponds subjected to strong erosion processes that increased the natural salinity of the soil. Epele and Miserendino (2015) identified in the same wetlands the number of insect families as the most robust metric showing consistent and accurate responses to livestock grazing impact. Recently, Manzo et al. (2020) assessed the main environmental constraints driving variation in macrophyte assemblages (106 taxa) and trophic status in 29 ponds in Austral Patagonia. Most assemblages were dominated by native plants, but exotics were present in 24 ponds, dominating plant cover (>45%) in 15% of them. Pond eutrophication symptoms (high phosphorus contents and chlorophyll-*a*) were associated with extensive cattle grazing and urbanization, and the richness of emergent and endemic species was related with good pond condition. The authors emphasized the crucial role that buffer zones play at wetlands and suggested specific conservation measures to be included in the current legislation.

## Emergent Land-Use Practices

Historically, beef production has mostly taken place on natural forages in extensive modalities, but in the last 20 years and because of different factors (increased demand, profitability, climatic constraints, etc.), cattle intensification in confined systems has arisen in marginal areas (Arelovich et al. 2011). This type of cattle breeding has recently become an alternative for farmers in northwest Patagonia.

There is strong concern about how these practices can alter the health and integrity of streams. Although mitigation measures for livestock impacts have recently been incorporated into the legislation (e.g., livestock exclusion from streams and proper waste and effluent disposal), these actions were not implemented yet.

A recent study (Horak et al. 2019) examined environmental and ecological changes at three low order tributaries of the Corinto River (Futaleufú Basin) (Sect. 8.4.5) associated with different modes of animal production management. The production systems examined were extensive cattle management, semi-intensive mode (non-confined), and feedlots (confined), which were contrasted with reference sites (best conditions, minimum impact). Compared to the reference sites, intensive and semi-intensive practices produced low to moderate impacts with regard to oxygen saturation, conductivity, and nutrient levels but resulted in a strong alteration of the riparian ecosystem and in-stream habitat conditions. Also, *Escherichia coli* counts exceeded the recommended critical value for recreational purposes, indicating a decline in water quality (Fig. 11.4). Although the bacteriological status of running water is regularly assessed in different basins by governmental agencies, this aspect of water quality is perhaps neglected in rural catchments and should be considered in the future. In addition, while the concentrations of nutrients were moderate, some forms of nitrogen (nitrates, ammonium) and phosphorus (phosphates) can lead to eutrophication processes in areas that are naturally oligotrophic. Water quality changes were evident in terms of dissolved oxygen (range: 7.2–13.5 mg $L^{-1}$) and nutrients (soluble reactive phosphorus, 80–850 μg $L^{-1}$; nitrate plus nitrite, 1.79–7.93 mg $L^{-1}$; and ammonium, 5–230 μg $L^{-1}$). In the study, Plecoptera and Trichoptera richness significantly decreased at the most impacted sites, especially during the low water period.

A comparative assessment of the effects of livestock, pasture, and horticulture production on piedmont streams suggested that the most severe impacts on in-stream habitats and riparian ecosystems occurred at those sites associated with livestock production (Horak et al. 2020). Farms with high animal stocking rates could cause habitat damage and loss of suitable habitats mostly as a consequence of trampling (Miserendino and Masi 2010). According to Horak et al. (2020), the types of disturbances detected on the stream corridor included, among others, signs of livestock foraging, narrowing of the riparian buffer, and changes in forest composition (e.g., native *Nothofagus* replaced by exotics *Salix* species).

Among the metrics, EPT richness, number of insect families, density of tolerant taxa, collector-gatherers, and total invertebrates all displayed significant responses to water quality. The overriding stressor in explaining changes in the community with respect to intolerant taxa (EPT) was ammonium. Values between 25 and 50 μg $L^{-1}$ of ammonium resulted in a significant decrease of this group (~up to 14% of the EPT average value). Regarding functional feeding groups, shredder abundance decreased at impacted sites, probably in response to the decreased availability of the woody debris and leaf litter that provide refuges and dietary resources.

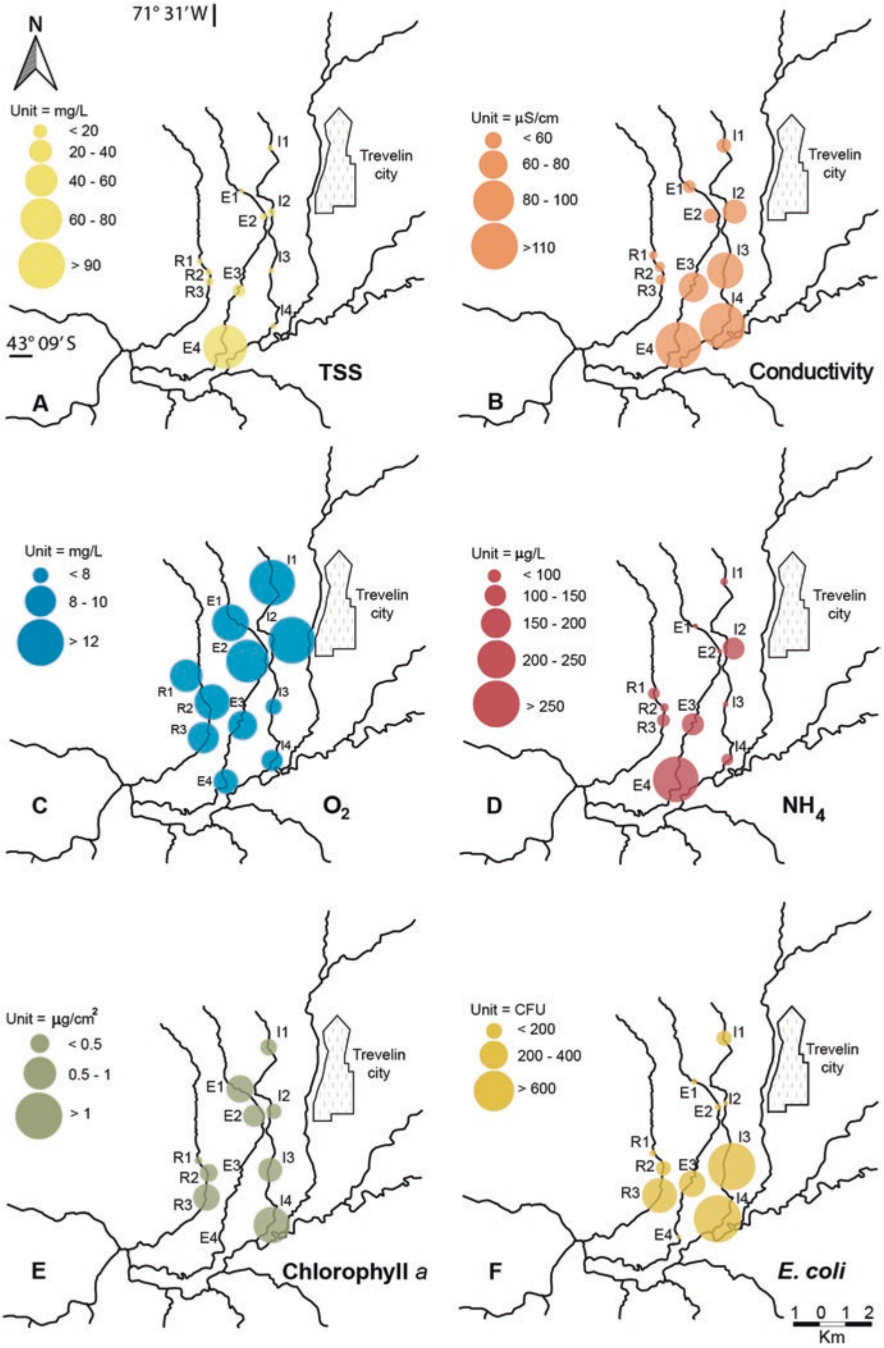

**Fig. 11.4** Trends of water quality variables for: (**a**) total suspended solids, (**b**) conductivity, (**c**) oxygen, (**d**) ammonium, (**e**) chlorophyll a, and (**f**) bacteria (*E. coli*), for Ruca (R), Enna (E), and Ingram (I) systems (Norwest of Chubut Province, Patagonia, Argentina). R1, I1, and E1 represent reference sites. Animal production modalities: R2, E3, and E4: semi-intensive; R3 and I4: intensive. (Adapted from Horak et al. 2019)

## Use of Agrochemicals

Agriculture represents the second most important economic activity in the northern Patagonian region of Argentina (Macchi et al. 2018). Most studies evaluating the consequences of the use of agrochemicals on aquatic environments have been conducted in the area where most agro-industries (fruticulture, horticulture, viticulture, among others) are based.

In a previous study, Tosi et al. (2009) assessed pesticide hazards on the surface waters in and around Negro River, Northern Patagonia, derived from agriculture and associated activities, such as fruit and vegetable production and packaging industries (see Sect. 8.4.2.3). Effects on aquatic organisms were assessed by assays of acute toxicity, available references, and analytical results. The authors indicate the probability of acute effects of azinphos-methyl (AM) in 37% of the studied species in the river, mainly fish and zooplankton. In samples taken from drainage channels, the value increased to 42% of the studied species. Additionally, carbaryl residues found in drainage channels affected 14% of the studied species, while in the river samples, the impact was negligible.

An assessment of the effects of the herbicide Magnacide® H on benthic invertebrates was carried out by Albariño et al. (2007) in irrigation channels connected to the Colorado River. They evaluated the effects of the use of Magnacide® H to control aquatic weeds in various channels of the lower basin. The authors found that even the recommended concentrations according to regulatory guidelines have a high toxicological impact on the aquatic ecosystem and kill most of the aquatic invertebrates. However, they also found that the presence of the substance in the channels was temporary. After applications, they documented a significant reduction in the number of taxa (−58%), abundance (−57%), and community diversity (−67%) of benthic assemblages compared to untreated channels in the first year of the study. They also recorded a reduction in the density of Oligochaeta, Chironomidae, Ostracoda, and the snails *Biomphalaria peregrina* and *Heleobia parchappii*.

Other studies conducted by Anguiano et al. (2012) evaluated the acute toxicity and biochemical effects of the organophosphorus pesticide AM on the amphipod *Hyalella curvispina*. The species inhabits ponds and irrigation channels of an intensive fruit-producing region in Río Negro and Neuquén valley, northern Patagonia. They found two coexisting subpopulations of *H. curvispina* with different susceptibilities to AM. The analysis of species sensitivity distribution showed that both subpopulations of *H. curvispina* were more tolerant to AM than most amphipod species. According to their results, the maximum concentration of AM in drainage water within the fruit-producing area would greatly affect most of the amphipod species and other local taxa.

Similarly, Guerreño et al. (2016) found differential susceptibility to AM in two fish species (*Odontesthes hatcheri* and *Jenynsia multidentata*) in the Río Negro valley. They documented probable acute effects on *O. hatcheri* in river and irrigation channels, while the risk was very high in superficial drainage water. In contrast, *J. multidentata* was exposed to minimal risk in river or channel waters and probable risk in drainage water.

Recently, Macchi et al. (2018) assessed the impact of nonselective insecticides (AM, chlorpyrifos, and carbaryl) on different attributes of macroinvertebrate communities in the water channels of the Río Negro and Neuquén Valley. They found significant changes in the abundance and richness of sensitive taxa in the affected drainage channels. The decrease in taxonomic richness was associated with chlorpyrifos and azinphos-methyl peak concentrations. The study revealed a decrease in sensitive taxa such as Baetidae and an increase in some tolerant taxa such as Chironomidae and Gastropoda. For all three pesticides, the acute hazard quotient exceeded the risk criteria for invertebrates. The authors concluded that macroinvertebrate assemblages in drainage channels were highly affected by chlorpyrifos levels.

## Forestry and Pine Plantation

Forest management involves several practices such as harvesting, deforestation, planting, and the replacement of native species by exotic commercial ones. In its different stages, forestry can include road building, harvesting, yarding, and site preparation for replanting – all actions that can represent a real and pervasive threat to the biodiversity and conservation of lotic ecosystems (Richardson and Béraud 2014; Allan et al. 2021). Widespread removal of basin vegetation has been shown to alter flow characteristics, change the amount of sediment introduced to stream systems, decrease infiltration, and increase surface runoff. In turn, an increase in nutrient levels can result in the overgrowth of algae and aquatic plants, which potentially alter habitat suitability for endemic fauna (Dodds and Smith 2016). Other consequences are changes in wood and detritus supply, habitat structure and stability, light reaching the stream, temperature, trophic resources, and also in water quality (Richardson and Béraud 2014).

Forest exploitation (e.g., wood collection for fuel, carpentry) and the substitution of native forest for pine plantations are common land-use practices in the Patagonian mountains. The impact of pine plantations on stream processing dynamics was explored in a study by Albariño and Balseiro (2002). They analyzed the effects of leaf litter quality and macroinvertebrate size on in situ litter breakdown. The main observations showed that the decay rates of the native *Nothofagus pumilio* leaves were twice as fast as *Pinus ponderosa* needles. They found similar abundances of macroinvertebrates colonizing both leaf pack types; however, macroinvertebrate biomass was shown to be higher in the *N. pumilio* packs. Moreover, shredders fed only on *N. pumilio* leaves. The presence of macroinvertebrates on *P. ponderosa* litter packs is probably because the packs offer refuge, as well as fine particulate organic matter for feeding, and biofilm resources. The authors concluded that the whole litter processing mechanism would be affected by the replacement of native forest by exotic pine forestation.

The question of what role pine needle litter plays in relation to stream invertebrates was investigated by Márquez et al. (2017) in central Argentina. Through different experiments, they analyzed the taxonomic and trophic structure of invertebrate assemblages colonizing both pine needles and plastic filaments (mimicking the physical structure of needle accumulations) that were placed in streams in afforested and natural grassland. It was concluded that needle litter functioned as habitat or refuge, but not as a food source. The authors observed that the taxonomic and trophic composition of the invertebrate assemblages which colonized the needles and the plastic filaments were mostly similar. Although these studies were carried out outside Patagonia, we consider that these findings could shed some light about the invertebrate community responses to pine needle litter at watercourses in the Patagonian steppe, wherein pine plantations are being promoted by national and provincial laws.

A comparative study conducted at mountain streams subjected to different land uses identified that difference in energy resources resulted in changes in community attributes (Miserendino et al. 2011). The assessment included native forests, pine plantations, harvest forests, urban and urban reference sites, and pastures. Shredder richness was higher in native and harvest forests than in exotic pine plantations; collector-gatherer density was consistently high in harvest sites, and total density was significantly higher in urban areas and harvest forests. Macroinvertebrate assemblages were different in forested sites (either with native or exotic species), as compared to those in riparian modified areas (pasture, urban, and harvest sites) (Fig. 11.5). The environmental variables with explanatory power on macroinvertebrate assemblages were mostly related to detritus availability (wood and leaf biomass). A complementary study revealed that among guild structure measures,

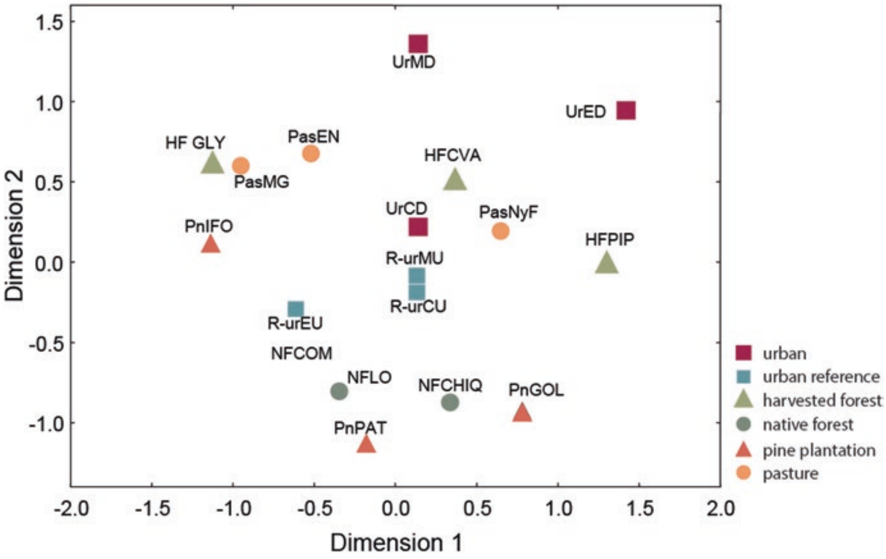

**Fig. 11.5** Multidimensional scaling ordination based on macroinvertebrate community data collected from 18 sampling sites subjected to different land uses (Chubut, Argentina). □: urban sites, △: harvested forest, ○: pasture, ■: reference urban sites, ●: reference native forest and ▲: pine plantations. (Adapted from Miserendino and Masi 2010)

indicators based on benthic community functional attributes, shredder richness, and collector density were good candidates for assessing land-use impacts (Miserendino and Masi 2010). As expected, differences in the allochthonous detritus supply and the autotrophic production (bryophytes and macrophytes) were significant between land-use practices.

Masi and Miserendino (2009) found that the overall mass of benthic particulate organic matter (POM) was higher in native and harvest forests than at pastures, whereas fine fraction (FPOM) was higher in harvest forests than in pastures. At the habitat scale, riffles supported higher leaf biomass and bryophytes than pools, suggesting differences in the distribution of the particulate organic matter and autochthonous resources.

## Forest Fires

Wildfires are considered major agents of landscape change; however, although their effects on terrestrial communities are rapidly visualized, their impacts on aquatic and wetland ecosystems are less known (Lyon and O'Connor 2008). According to Mellon et al. (2008), fire has a stronger influence on headwater streams than on large rivers because a greater proportion of the catchment is burned, and the ratio between stream margin and water volume is higher. The main direct effects arise from increased water temperatures, alterations in water chemistry, and food quality. Long-term changes include chemical responses, increased erosion processes, and changes in the abundance of macroinvertebrate species (Koetsier et al. 2010). The recovery of these ecosystems to prefire conditions is variable but mostly occurs within a few years or decades (Rhoades et al. 2011).

Few studies have been conducted on Patagonia regarding postfire effects on running waters. In an early study, Temporetti (2006) documented the long-term effect after forest fires at two mountain streams (Chalhuaco and Ñireco) in Nahuel Huapi National Park. Effects in the water were evident even 3 years after the fire, mostly in terms of nitrate concentration, which was significantly higher in the sites affected by forest fires. On the contrary, pH, conductivity, and suspended solids were within the same order of magnitude of those of unimpacted rivers and streams of the region. Measured concentrations of total phosphorus, nitrite, and ammonium in water were low at impacted sites. According to the author, the concentration of nitrate in the water appeared to be the most effective way to evaluate the long-term effect of forest fires.

More recently, a study conducted in Los Alerces National Park (Argentine Patagonia) assessed the effect of fire on a mixed *Nothofagus* forest. The fire occurred during March–April 2015, when 1228 ha were affected. In this work, Williams-Subiza and Brand (2018) compared the physicochemical variables and community structure of headwater streams from burned and unburned areas. The authors had the opportunity to sample shortly after the fire was extinguished. They found that water temperature, total dissolved solids, total suspended solids, and conductivity were significantly higher in burned sites, whereas nutrient, chlorophyll-*a*, and oxygen concentrations exhibited no major differences. Regarding macroinvertebrate communities, the authors did not report changes in total richness, abundance, or diversity between streams, but taxonomic richness and densities of sensitive taxa were markedly lower at burned streams, while noninsect taxa (mainly oligochaetes and amphipods) were significantly more abundant in the disturbed reaches (Fig. 11.6a). An analysis of functional feeding groups revealed that the relative abundances of shredders and collector-filterers were lower at burned sites, possibly reflecting a shift in detrital input (Fig. 11.6b).

## *2.2 Hydrological Interventions*

Energy demand in Argentina has markedly increased during the last few years. Some of the hydroelectric power plants that provide electricity through the national interconnected network are located in Patagonia. The main large-river dams are located in Neuquén/Río Negro province (Limay River, five dams; Neuquén River, one dam), Chubut Province (Futaleufú River, one dam; Chubut River, one dam), and Río Negro/La Pampa Provinces (Colorado River, two dams). Further important dams are planned for the next decade in Neuquén Province (Neuquén River, three dams; Limay River, one dam), Chubut Province (Carrenleufú River, one dam), and Santa Cruz Province (Santa Cruz River, two dams). In fact, with the construction of Condor Cliff and Barrancosa dams on the Santa Cruz River, no large watercourses in the region will remain unregulated (see Chap. 9).

Changing river flow through dam construction can produce different impacts on environmental conditions and on the biota (Arthington et al. 2018). A study conducted on the Chubut River, where the hydrological regime is modified in the lower section by an artificial reservoir (Florentino Ameghino Dam), revealed that total suspended solids, total phosphorus, and particulate organic matter decreased, whereas soluble reactive phosphorus (SRP) and chlorophyll-*a* increased below the dam (Fig. 11.7). Macroinvertebrate density increased threefold in this area, possibly due to habitat improvement and enhanced trophic resources. Mean species richness did not change downstream of the dam; however, the community was dominated by gastropods, chironomids, and flatworms (Miserendino 2009). In terms of relative abundance, collector-gatherers and scrapers/grazers were the prevailing groups. Collector-filterers comprised more than 25% of abundance at the pre-regulated site and were practically absent right downstream of the dam (Miserendino 2007).

Condor Cliff and Barrancosa dams (Santa Cruz River) are currently under construction and would supply 16% of Argentina’s hydropower. An intensive survey conducted at regular 6 km intervals (over 310 km downriver) in the Santa Cruz River (Tagliaferro et al. 2013) revealed homogeneous habitat structure along the river despite some particular characteristics at the local scale. In turn, macroinvertebrate density and richness (38 genera) proved to be among the lowest, compared with 42 other Patagonian watercourses. The reduced flow variation and the lack of bed-scouring flows would have a direct and negative effect on the heterogeneity of

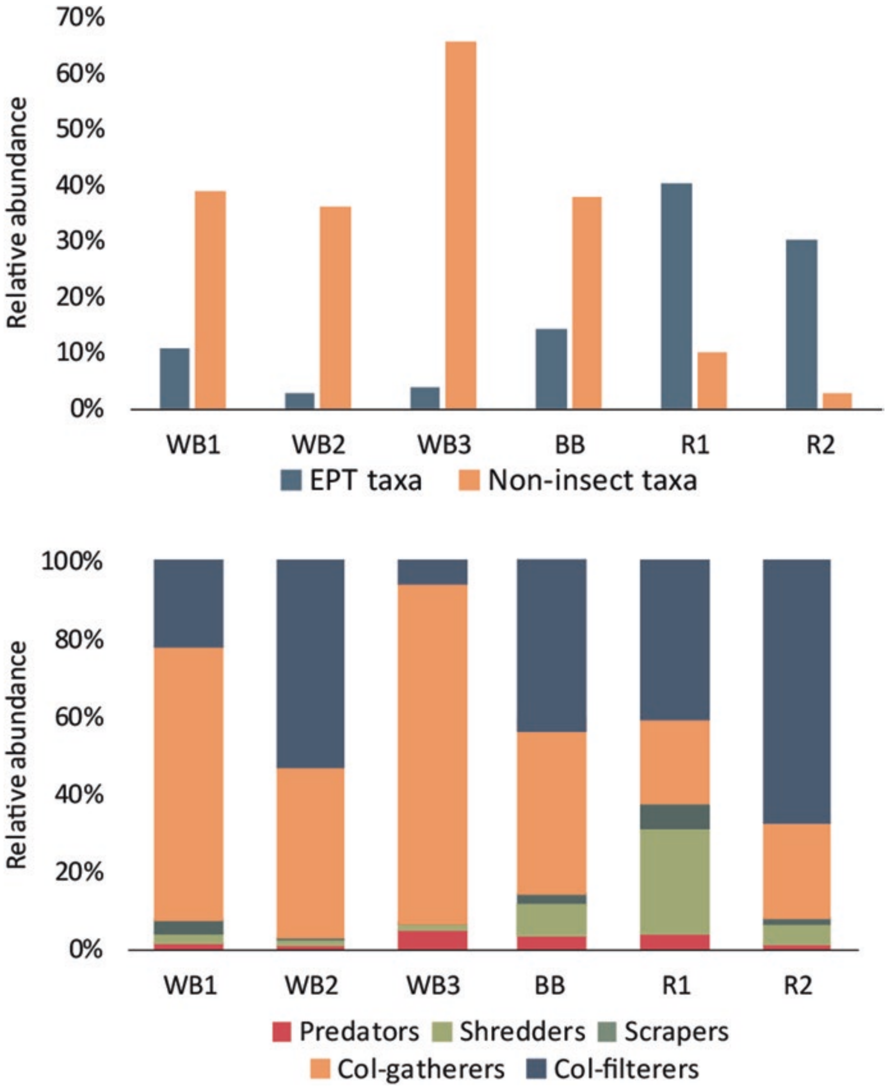

**Fig. 11.6** Abundance patterns of macroinvertebrates at different stream reaches affected by forest fires. (**a**) Relative abundance of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT) and non-insect taxa. (**b**) Relative abundance of functional feeding groups at the six studied reaches ($n = 4$). WB within burn, BB below burn, R reference. (Adapted from Williams Subiza and Brand 2018)

riverbeds and banks. According to the authors, high turbidity may also contribute to shorter food webs by affecting autotrophic production, general trophic structure, and overall macroinvertebrate productivity and diversity. They predicted that the planned dams would obliterate 51% of the lotic environment, including the most productive sections of the river. Tagliaferro and Pascual (2016) pointed out these

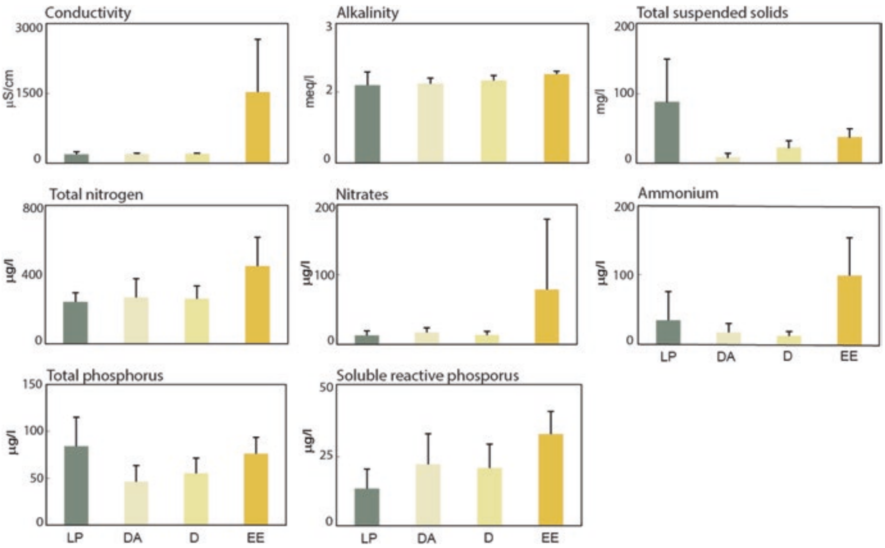

**Fig. 11.7** Mean values (±SD, *n* = 4) of physicochemical variables at pre- (LP: Las Plumas) and post-impoundment sites (DA: Florentino Ameghino Dam, D: 28 de julio, EE: Chubut estuary) in the Chubut River, Patagonia, Argentina, during a seasonal study conducted during 2004. (Adapted from Miserendino 2009)

dams would modify in-stream habitat conditions, leading to changes in macroinvertebrate community structure and local fish abundance due to loss of key prey taxa. Quiroga et al. (2015) hypothesized significant habitat loss of *Oncorhynchus mykiss* juvenile, an exotic recreationally species (see Chap. 13). Predictions were based on data collected through an intensive field survey of fish and environmental variables along 310 km of river. Based on this model, the authors estimated that habitat supporting 86% of current juvenile trout production would be lost due to the river sector that both dams will flood (see details in Chap. 13).

## *2.3 Urbanization*

Urbanization is one of the most fast and deep types of land-use change that degrades water bodies and wetlands and contributes toward decreasing their ecological integrity and health (Paul and Meyer 2001; see also Sect. 4.2). Urban land use induces substantial changes in multiple physicochemical properties of freshwaters and local loss of biodiversity (Utz et al. 2009; Rojas et al. 2019). Typical alterations include increase of impervious surfaces (altering infiltration rates and runoff), filling of wetlands (loss of ecosystems or impaired water quality), channelization of watercourses (loss of habitat, banks, and connectivity), loss of connectivity, and discharge of domestic and industrial effluents (impaired water quality). Compared to other land uses, the effects of urbanizations appear to be more severe (Allan et al. 2021). However, this phenomenon has probably received little ecological attention, especially in remote areas. This section documents the main results of studies associated with the urban areas of the Patagonian mountains.

Discharges of untreated domestic effluents can have a significantly detrimental impact on physicochemical conditions and aquatic biota. Earlier studies conducted in northwest Chubut have assessed the environmental impact of organic pollution on urban streams after the installation of wastewater treatment plants (WWTP) (Miserendino 1995; Pizzolón et al. 1997). In some small towns, the ecological conditions of urban streams were rapidly improved. Nevertheless, the construction of planned facilities was delayed in many areas due to financial problems. In addition, in some areas, this problem is growing fast because of accelerated urban development.

A seasonal study conducted in the Patagonia mountains (northwest Chubut) that included streams at Cholila, Esquel, and Corcovado towns (2190, 30977, and 1848 inhabitants, respectively) revealed important environmental impacts at post-urban reaches (Miserendino et al. 2008). The most impaired site was downstream the largest city (Esquel), where high values of ammonium, conductivity, and TSS were important in structuring invertebrate assemblages. Overall, urbanization resulted in varying degrees of habitat degradation, sedimentation, and nutrient enrichment at most urban sites, all of which affected macroinvertebrate communities. Insects (*Senzilloides panguipulli*, *Meridialaris chiloeensis*, *Polypedilum* sp., and *Rheotanytarsus* sp.) and Oligochaeta (*Nais communis*) dominated assemblages at reference and moderately impaired sites. The strongly polluted reach downstream Esquel, which had low flow in summer, showed a community dominated by Annelida (*Limnodrilus* spp., *Helobdella* spp.) and Amphipoda (*Hyalella* spp.). In contrast, fish density and biomass varied in a non-systematic manner among sites. More recent studies have demonstrated a strong impoverishment in water quality in Esquel Stream, related to the poor functioning of the wastewater treatment plant (Assef et al. 2014b; Manzo et al. 2020). The nutrient concentrations of treated effluents exceeded permissible standards for discharging into surface waters according to regional regulations. The functioning of the wastewater plant could be improved through proper design and management, including the construction of new modules, enlarging of the primary treatment system, completion of water-storm network, and reuse of treated effluents.

During 3 consecutive years, Mauad et al. (2015) surveyed seven sites in the Challhuaco-Ñireco stream system (Nahuel Huapi National Park) that runs through San Carlos de Bariloche City (150,000 inhabitants). The study evaluated ecological features from the headwaters to the stream outlet on the Nahuel Huapi Lake. The species assemblages at headwaters were mostly represented by sensitive insects, whereas tolerant taxa such as Tubificidae, Lumbriculidae, Chironomidae, and the crustacean *Aegla* sp. were dominant at urban sites. Regarding functional feeding groups, collector-gatherers were dominant at all sites, and this pattern was consistent over the 3 years. Macroinvertebrate communities responded primarily to temperature and elevation gradients but were also associated with variables related to human impact (conductivity, nitrate, and phosphate contents). As for macroinvertebrates, total richness, EPT taxa, Shannon diversity index, and Biotic Monitoring Patagonian Stream (BMPS) biotic index were shown to be fairly consistent and indicated different levels of disturbances in the stream. BMPS was developed by Miserendino and Pizzolón (1999), who were the first to present a biotic index for water quality assessment in lotic systems of Patagonia mountains. This index was an adaptation of the Biological Monitoring Working Party index and is computed by adding pollution sensitivity scores (1–10) for all invertebrate families present at a site for a potential range of 0 (most disturbed) to >150 (least disturbed).

The consequences of anthropogenic pressures at different sections of a Patagonian mountain river (Percy River system) were recently evaluated using a set of environmental and biological measures (Miserendino et al. 2016). The study also assessed the composition and structure of the riparian corridors, where discharge, intactness of the river channel, floodplain morphology, conservation status, and percentage of urban areas were the most important drivers of plant community composition. The lower, urbanized basin associated with the town of Trevelin (10,000 inhabitants) showed an increase in conductivity and nutrient values, with a poor riparian quality and habitat condition. The main variables affecting benthic assemblages were ammonium concentration, elevation, current velocity, and habitat conditions. Present land use in the basin is not currently deemed to pose a significant risk of soil erosion; however, unsustainable practices that promote the substitution of forest by shrubs at catchment scale could lead to severe consequences including sedimentation and soil loss. In turn, the increase in the severity of fine sediment pulses on macroinvertebrates survival and sensitive-species extirpation constitutes a major threat.

## *2.4 Mining*

Mining is recognized as one of the most impactful activities on aquatic ecosystems, causing severe environmental degradation, damage to the biota, and a reduction of nature's contributions to people they provide (Winterbourn et al. 2000; Magrin et al. 2014). Despite that, Patagonia has a long tradition in mining activities (oil, gold, silver, iron, aluminum, coal, among others), and only few studies have assessed the impact of this industry on freshwater environments. Extraction of gravel from riverbeds for construction purposes is also important and has direct impacts on the in-stream habitat and at the reach scale.

The Turbio River sub-basin (Gallegos River basin, Santa Cruz Province) is subjected to the mixed impact of different anthropogenic activities, mostly coal mining and urbanizations (Río Turbio and 28 de Noviembre) (Sect. 3). Recently, Asueta (2016) showed that the water quality of the Gallegos River varied between being heavily contaminated in the proximity of urbanizations and probably having incipient disturbance for those sites distant from urban and industrial pollution. Parameters such as conductivity, total solids, hardness, turbidity, COD (chemical oxygen demand), and concentration of iron increased in river segments associated with industrial activities derived from coal mining. Oligochaetes and Chironomidae were indicators of negative impact, whereas Ephemeroptera, Plecoptera, and Trichoptera appeared to be associated with better environmental quality. Nevertheless, the author found that total richness was low at most sites, indicating that macroinvertebrate communities can be naturally poor in these environments. This should be considered in future biomonitoring approaches.

Research conducted in coastal areas (San Antonio Bay) assessed the impact of long-period exposures (30 years) to mining waste (Marinho et al. 2017). Authors documented the concentrations of different metals (cadmium, lead, copper, and zinc) in the soil pile next to the Encerrado channel (protected area). These exceeded guidelines for agricultural, residential, and industrial land uses. In the bay, the sediments closest to the pile (mudflat and salt marsh) exceeded sediment quality guidelines for the protection of biota. They concluded that pollution is a persistent problem as a consequence of mining wastes.

## 2.5 Biological Invasions in Patagonian Freshwaters

Biological invasions constitute a major component of anthropogenic global change and threaten native biodiversity and ecosystem functioning all around the world. The introduction of non-native species into lotic systems in Patagonia has attracted the attention of stream ecologists and society at large (Sanguinetti et al. 2014). Species have successfully invaded different environments including riparian areas, streams, lakes, and wetlands. The level of disturbance produced is well known for some species and has resulted in profound modifications of the dynamic and functioning of ecosystems. On the other hand, other invasive species have only recently been detected, and their effects are still being investigated (see Chaps. 12, 14, and 15).

In northern Patagonia, riparian ecosystems are highly modified due to alien plant invasions. Willows of the *Salix alba–Salix fragilis* complex, native to western Eurasia, represent typical invaders of floodplains, and their distribution has increased significantly along the rivers (Budde et al. 2010). Thomas and Leyer (2014) assessed the impact of the establishment of non-native willows and poplars on *Salix humboldtiana*, which is the only native willow in the floodplains and ecotonal areas of the steppe. They analyzed age structure and growth performances within mixed adult forest stands along the Negro River, concluding that invasive willows significantly outperform native ones. However, they also observed that willow stands are usually removed by the river dynamics, which also creates new sediment bars and islands. This process enabled *S. humboldtiana* seeds to germinate and compensate for both its inferior growth performance and any potential impacts by the invasive willows and poplars. The question of how exotic trees invading riparian areas can impact aquatic systems was investigated by Serra et al. (2013). They conducted a field survey of benthic invertebrates and leaf litter and an in situ experiment to examine the effects of the exotic *Salix fragilis* (crack willow) on the structure and functioning of small streams. The study showed that crack willow leaves decomposed slower than the native *Ochetophila trinervis* (chacay) and that the contribution of benthic leaf litter was similar between the two riparian vegetation types. In stream reaches dominated by crack willow, leaves of this species represented 82% of the total leaf litter. However, the effect on benthic invertebrates appeared low and differed between stream reaches, mostly in terms of species composition.

*Didymosphenia geminata* (didymo) is a stalked freshwater epilithic diatom, which forms large blooms in ultra-oligotrophic conditions, and it has been declared a nuisance species worldwide. Its presence has been verified in Argentine Patagonia since 2010 (Sastre et al. 2013) (see Chap. 12). The assessment of its impact on a community of macroinvertebrates was conducted by Brand and Grech (2020) in a seasonal study at Futaleufú River. The main finding was that macroinvertebrate communities differed between didymo infected and control sites. The observed changes were mostly in terms of total density, tolerant group (Orthocladiinae and noninsect taxa) density, and total richness, which were all higher at didymo-affected sites. An analysis of trophic structure revealed that abundances of shredders and collector-gatherers also increased in invaded sites. The authors posited that the diatom blooms may constitute a new complex microhabitat providing refuge from harsh current and predators as well as new foraging opportunities, producing a bottom-up effect on the benthic community.

Añón Suárez and Albariño (2020) examined the impact of didymo algal blooms on the distribution of the benthic crustacean *Aegla riolimayana* in the littoral zone of the upper Limay River. The species is a key component in the transfer of matter and energy in the lakes and rivers of Patagonia. The authors conducted a 3-year study (summer of 2016, 2017, and 2018) at different locations and habitats (shore, shallow, and deep) of the river with and without the diatom. As they found no correlation between *A. riolimayana* abundance and percentage didymo cover, they suggested that there was no negative effect of didymo on the abundance and distribution of this crustacean at the reach scale.

Recently, Brand et al. (2018) assessed the population patterns of the exotic *Physa acuta*, a Northern-Hemisphere gastropod, in post-urban reaches of the Esquel-Percy system. The first record in this region occurred in 2005 in Esquel Stream. From 2005 to 2017, the authors seasonally studied riffles and pools of this river system. The highest densities of *P. acuta* were observed downstream from the wastewater treatment plants of Esquel and Trevelin, and it was observed that population density increased by a factor of 160 between 2005 and 2017, indicating that the population of *P. acuta* is currently expanding.

## 3 Bioindicators and Biomarkers in Ecological Assessments

There have been strong efforts to implement the use of bioindicators to assess water quality in Patagonia. In this context, a group of experts, managers, and governmental agencies started to collaborate in establishing main directives for biomonitoring

(Miserendino et al. 2020). Biotic indices and metrics can be based on a variety of freshwater organisms, such as algae and macroinvertebrates. Through the years, researchers have adapted existing biotic measures for their use in Patagonia, which requires a detailed knowledge of regional species pools and endemisms of each taxonomic group. Miserendino and Pizzolón (1999) were the first to develop a biotic index for water quality assessment in lotic systems of Patagonian mountains. Since then, and as described above, several studies have implemented the use of bioindicators in the ecological assessment of aquatic systems in the region. Most of them were based on macroinvertebrates and successfully evaluated the impact of different land-use practices. In general, the richness of sensitive insect families and the EPT richness appear as the most robust measures in evidencing anthropogenic disturbances (Miserendino 1995; Miserendino et al. 2008; Brand and Miserendino 2015; Williams-Subiza and Brand 2018; Horak et al. 2019). Recently, algal communities and macrophytes have also been incorporated into biological assessments of water quality (Manzo et al. 2020).

The use of biomarkers (functional measures of exposure to stressors expressed at the suborganismal, physiological, or behavioral level; McCarty and Munkittrick 1996) as surrogate measures of biological impact within laboratory and field studies has also been included in recent regional studies. Different systems protect aquatic organisms from xenobiotics by reducing the accumulation of toxins in cells. P-glycoprotein (P-gp)-mediated multixenobiotic resistance (MXR) is a defense mechanism analogous to multidrug resistance (MDR) that has been observed in several aquatic organisms inhabiting the watercourses of Northwest Chubut. As this phenotype is modulated by natural and man-made chemicals, MXR is used as a biomarker of the organisms' exposure to environmental contamination. The mechanism has been characterized in the exotic freshwater snail *Physa acuta* (Assef et al. 2014a; Horak and Assef 2017). In addition, the presence and activity of MXR were analyzed in different freshwater macroinvertebrate species (leeches, *Helobdella michaelseni*, *H. simplex*, *Patagoniobdella variabilis*; amphipod, *Hyalella curvispina*; and midge, *Chironomus riparius*). In this study, *H. michaelseni*, *H. simplex*, and *P. variabilis* displayed activity in the MXR system, suggesting their suitability for in vivo biomonitoring (Assef et al. 2014b).

The MXR mechanism was also evaluated in fish species that are widespread in Patagonia, native *Hatcheria macraei* (Patagonian catfish) and exotics *Salmo trutta* (brown trout), *Oncorhynchus mykiss* (rainbow trout), and *Oncorhynchus tshawytscha* (Chinook salmon) (Assef et al. 2019). Results from different tissues such as liver, gills, muscle, and epidermis indicate that the fish studied have different species-specific levels of MXR activity, with the gill and liver tissues displaying greater detoxifying activity. The induction of MXR transport activity was also identified in liver tissue of rainbow trout from an urban stream, suggesting its suitability for the biomonitoring of aquatic environments subjected to urban contaminants. According to the authors, these are promising approaches for assessing the potential risk of exposure to different xenobiotics and new emerging contaminants.

## 4 Mitigation Measures

Given the extensive and diverse land-use practices existing in Patagonia (Sects. 3 and 4), it is necessary to implement biomonitoring programs that help to identify the response of aquatic ecosystems to different stressors caused by such activities. These programs would benefit from the experience developed so far to assess the degree of deterioration at different spatial scales (reach, stream, and basin) and to propose mitigations measures accordingly. Based on the scientific knowledge discussed here and other relevant literature (Allan et al. 2021), we detail actions that can be implemented in order to mitigate ecosystem alterations resulting from anthropogenic drivers. These measures are proposed to protect freshwater ecosystems, their ecological integrity, and aquatic biodiversity (Table 11.1).

Some mitigation actions are very simple and low cost. For example, for those agricultural practices that result in livestock intrusion to watercourses, it is possible to limit livestock access to the riverbanks and water bodies using electric fences or provide drinking water points away from riverbanks. Other important measures, such as the early detection of invasive species and long-term biomonitoring programs (e.g., pollution control and erosion risk), may require a significant investment and the articulation of multiple actors (e.g., scientists, managers, and governmental agencies).

## 5 Conclusion

We synthesized studies that examined the influence of the main land-use practices on the characteristics of aquatic and wetland environments, as well as on the organisms, populations, and communities of different ecoregions of Argentinean Patagonia. These studies indicated that freshwater integrity is being threatened by anthropic pressure. The implementation of management and conservation actions on these valuable but fragile ecosystems is urgently needed. We propose different mitigation measures to minimize the impacts produced by different land-use categories. The promotion of measures to maintain the adequate functioning of aquatic and wetland ecosystems and to protect their biodiversity should be conducted through collaborative work, involving governmental agencies, scientists, landowners, and local communities. The success of this enterprise also depends on the continuity of bioassessment programs with long-term approaches for monitoring the evolution of impacted ecosystems. In this context, changes in the quality and quantity of water of freshwater ecosystems are expected. If protective measures are not carried out, a worsening of environmental problems associated with current land-use practices could well occur.

**Table 11.1** Suggested mitigation measures according to different land-use practices for the protection of Patagonian freshwater ecosystems and wetlands and their aquatic biodiversity

| Agriculture | Hydrological interventions | Mining | Urbanizations | Biological invasions |
|---|---|---|---|---|
| ***Pasture conversion, livestock and agricultural practices***<br>Promoting water protection, restoration, and conservation of riparian ecosystems<br>Replanting native riparian vegetation<br>Limiting livestock access to the riverbanks and water bodies (electric fences, drinking water points)<br>Reinforcement of riverbanks at crossing areas (with cobble/gravel)<br>Treatment of animal waste (pond farms)<br>Regulation and control of fertilizer application<br>Landowner involvement in conservation (economic incentives)<br>***Agrochemicals***<br>Improving control on the application conditions<br>***Forestry and pine plantation***<br>Maintaining a buffer zone<br>Proper disposal of postharvest material (leaves, branches)<br>Reforestation with native woody species and retention of native forest patches in riparian corridors<br>***Forest fires***<br>Replanting native riparian vegetation<br>Excluding cattle in the early stages of succession to allow revegetation and prevent erosion | Maintaining water regulation in order to replicate the natural seasonal flow cycle<br>Maintaining ecological flow of watercourses in order to ensure the support of aquatic organisms' life requirements (invertebrates, fishes) | Regulating water consumption in developments<br>Implementation of monitoring of water quality during all stages of development (e.g., post-mining period)<br>Avoid/limit dredging on river channels for gravel extraction | Improvement of domestic effluent treatments<br>Appropriate design of wastewater treatment plants<br>Reduction of external nutrient input<br>Prevention of runoffs<br>Restoring aquatic habitat in heavily modified areas<br>Stormwater mitigation measures<br>Pluvial network systems from domestic effluent<br>Reduction of impervious cover<br>Minimize channel dredging actions | Monitoring programs for early detection of potentially invasive species<br>Prioritize areas of hydrographic isolation, high conservation, or economic values<br>Eradication of invasive species<br>No action |

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# Chapter 12
# Invasive Species: The Case of *Didymosphenia geminata* in Neuquén, Argentina

**Guadalupe Beamud, Lorena Laffitte, Julieta Muñiz Saavedra, Gustavo Baffico, Mónica Diaz, Betina Laurenzano, and Fernando Pedrozo**

## 1 Introduction

The diatom *Didymosphenia geminata* (Lyngbye) M. Schmidt, a native species to the Northern Hemisphere, is a very aggressive invasive phytobenthic organism, inhabitant of rivers, streams, and lakes of the world, which can cause harmful environmental and economic impacts (Spaulding and Elwell 2007; Blanco and Ector 2009; Reid and Torres 2014). *D. geminata* cells are attached to the substrate by an extracellular polysaccharide stalk (Gretz 2008) produced in the apical pore at the base of the cell. Over time, the stalks lengthen and branch during cellular division. The result is a dense mat that can be up to several centimeters thick and cover extensive portions of the riverbed. These mats significantly impact the aesthetic appearance of waterbodies (Spaulding and Elwell 2007), habitat structure, and trophic web dynamics (Kilroy et al. 2009). The average maximum linear dimension (MLD) of *D. geminata*

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cells is 120 μm (± 12.7 μm) and the average width is 37.5 μm (± 3.5 μm) (Fig. 12.1d). *D. geminata* stalks are usually colonized by small diatoms, mainly *Achnanthidium* sp. but also *Navicula* sp. and *Tabellaria* sp. (Fig. 12.1e) (Beamud et al. 2013).

*D. geminata* presents an ecological paradox which challenges our understanding of the fundamental processes of river ecosystems since it has the apparently unique ability to rapidly produce large algal biomass developments or blooms under conditions of very low nutrient concentrations (oligotrophic environments). How can it produce so much biomass so quickly under conditions of low or undetectable nutrients and colonize environments with such high turbulence that causes the detachment of the algal mats? This paradox is known as "the didymo paradox" (Sundareshwar et al. 2011). The explanation is not clear yet, but it is likely to include aspects of ecology, hydrology, biogeochemistry, and geomorphology. A wide range of niche parameters have been reported for the species (summarized in Cullis et al. (2012)). Although *D. geminata* cells have been observed in waterbodies with a wide range of mean temperatures (Spaulding and Elwell 2007), temperature is considered an important factor in defining its habitat. Kumar et al. (2009) and Hix and Murdoch (2019) showed that colder temperatures were positively correlated with *D. geminata* blooms and density. Low water temperature could be important in the production of the stalk and in the activity of the enzymes that catalyze the hydrolysis of organic phosphorus (phosphatases), which are located in the stalks (Whitton et al. 2009; Bray et al. 2016a). Another possible hypothesis is that the effect of temperature is secondary and that different characteristics of oligotrophic waterbodies, such as low nutrients and high availability of light, are the conditions that give *D. geminata* the competitive advantage over other periphytic algae (Cullis et al. 2012). For instance, light availability has a positive impact on *D. geminata* cell division rate (Bothwell and Kilroy 2011) and on stalk height (Kilroy and Bothwell 2011). Rivers and streams of transparent and turbulent waters are generally located at the highest altitudes and latitudes, in the coldest areas of the planet. In recent studies, it has been shown that other chemical variables can play an important role in *D. geminata* growth. Iron (Fe), sulfur (S), and dissolved organic carbon (DOC) are considered important in bloom occurrence and in complex biogeochemical processes that take place within the mats, where the release of phosphorus (P) from particulate P and its consequent accumulation is promoted in stalks (Sundareshwar et al. 2011; Bothwell et al. 2012; Bray et al. 2017). On the other hand, Rost et al. (2011) showed that the increase in the concentrations of calcium (Ca) and sulfates ($SO_4$) in water was positively correlated with the presence of *D. geminata*, probably due to the fact that these elements are components of stalk material (Gretz 2008).

Blooms often occur in environments with very low soluble reactive phosphorus (SRP) concentrations (Bothwell et al. 2014), at or below the concentration detectable by many analytical techniques. According to West et al. (2020), <11 μg P $L^{-1}$ would be necessary for the formation of blooms, although there are differences according to geographic areas. These authors determined that in the Southern Hemisphere, *D. geminata* blooms do not occur above 4 μg P $L^{-1}$ while in North America, there may be blooms at sites with SRP <2 μg P $L^{-1}$ and up to concentrations of 8.5 μg P $L^{-1}$.

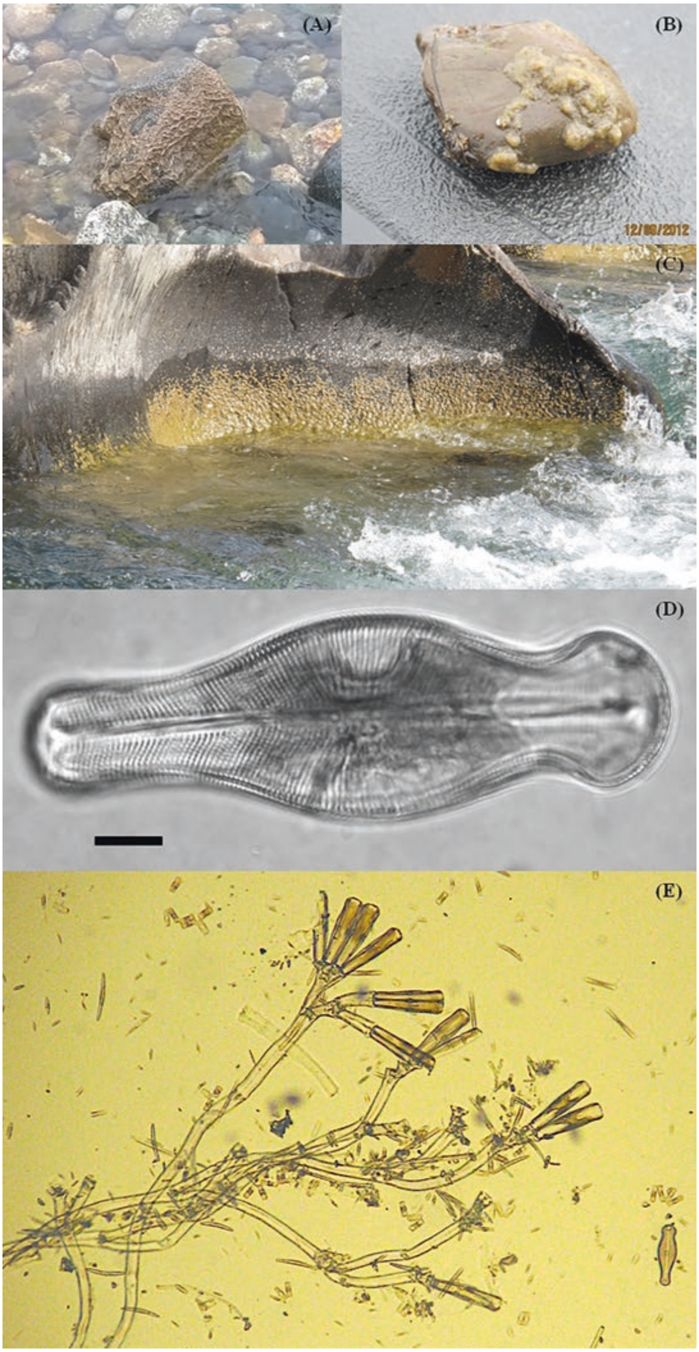

**Fig. 12.1** (**a**) Detail of the proliferation of *D. geminata* in Traful Lake, (**b**) close-up of the development of *D. geminata* on stones in Chimehuin River, (**c**) detail of the proliferation of *D. geminata* in the Collon Cura River; (**d**) *D. geminata* cell, and (**e**) detail of *D. geminata* stalks. Scale bar = 10 µm. ((**a**–**c**) Courtesy of DGBA; (**d**) modified from Beamud et al. (2013); (**e**) courtesy of G. Baffico)

In the Patagonian lakes and rivers of Argentina and Chile, the importance of P in photosynthetic rates, growth, and occurrence of *D. geminata* blooms was demonstrated (Beamud et al. 2016). However, the highest P values in these Patagonian environments with massive *D. geminata* blooms do not coincide with the global pattern indicated by Bothwell and Spaulding (2008) for this species (<20 μg P $L^{-1}$). On the other hand, Bray et al. (2016b) identified not only P as an important nutrient for *D. geminata*, as concentrations of nitrates + nitrites greater than 2 mg $L^{-1}$ limit the biomass of this species.

The problem of *D. geminata* is relevant on a continental scale in South America, since the range of dispersion expected includes Chile, Argentina, Uruguay, Perú, Ecuador, Bolivia, and Colombia (Spaulding and Elwell 2007; Reid et al. 2012). Since the only previous records of *D. geminata* in South America only date from rivers and lakes in Chile (Asprey et al. 1964; Rivera and Gebauer 1989) and studies describing periphytic communities in Argentinean Patagonia are very scarce, the alga status as an invasive species has been debated (Taylor and Bothwell 2014). Unlike in Chile, where *D. geminata* has been validated as an invasive species by Cayupe et al. (2018) with the introduction of multiple lineages or with a rapid genetic differentiation in this species, in Argentina, two possible situations may have taken place. According to Jones et al. (2019), the appearance of these blooms may have been due to the recent introduction of multiple lineages, as in Chile, or the species was already present, and given certain favorable conditions, it became more abundant.

The first *D. geminata* blooms in South America were found in the binational basin (Argentina-Chile) of the Futaleufu River (Los Lagos Region, Chile) (CIEP 2010), west of the Andes in May 2010 (see Fig. 9.1). Later, it was found in the same basin of the Futaleufu River but east of the Andes (Chubut, Argentina) in August 2010 (Sastre et al. 2013). In November 2011, it was recorded by the Provincial Department of Waters of Río Negro (Alemanni, "personal communication"), in the Manso River (Río Negro, Argentina), also belonging to a binational basin, the Manso-Puelo watershed. Afterward, it was recorded in the Chimehuin and the Collon Cura Rivers (Neuquén, Argentina; Abelli Bonardi et al. 2012). In February 2013, *D. geminata* was first observed in the binational Grande River in Tierra del Fuego Province (Sala et al. 2013); and in October 2014, it was found in the River de las Vueltas (Santa Cruz Province) (Braccalenti 2016). These rivers are important for touristic and recreational activities, and some are in binational basins, so actions and management in one country can affect ecosystems in the other country. In January 2013, the species was detected on the shores of the Nahuel Huapi Lake (Beamud et al. 2013), this being the first bloom of *D. geminata* recorded for a Patagonian lake. The Nahuel Huapi Lake is the largest and most important in population and tourism development of the homonymous national park (Chaps. 9 and 17). This lake drains into the Limay River, which runs through North Patagonia from the west to east, after crossing five chained reservoirs that are mainly used for irrigation and hydroelectric generation.

The most significant impacts of *D. geminata* on the host ecosystem are probably mediated by changes in biological and physical processes (substantial increase in algal biomass sediment retention and variation in hydrodynamics of rivers) and biogeochemical processes (changes in pH, phosphorus absorption dynamics) within the algal mats during blooms (Reid and Torres 2014). The potential displacement of native communities, as a consequence of the settlement and dominance of an invasive species, can have a significant effect on river nutrient cycles and ecosystem functioning, strongly impacting on the productive and recreational activities in the area (tourism, fishing, irrigation, etc.). Dense mats of this alga can detach during periods of high flow and clog water intake filters, which are costly to clean, reducing the efficiency of irrigation or hydroelectric channels.

*D. geminata* was declared an invasive exotic species in the Argentine Republic by the Secretariat of Environment and Sustainable Development (SAyDS), National Resolution No. 991/2012. Moreover, the Patagonian provinces affected by the invasion also legislated in this regard by declaring the state of emergency for all the aquatic systems affected by *D. geminata* and establishing the monitoring and prevention of exotic algal species on a multidisciplinary and interinstitutional basis as a priority public policy in environmental matters (Chubut, Law XI No. 58 (10/18/2012); Neuquén, Provincial Emergency Provision (Provincial Decree No. 0137/2012); Río Negro, Law No. 4801 (10/19/2012)).

## 2 *D. geminata* Presence: Management and Prevention Actions in the Neuquén Province, Argentina

Nature plays a critical role in providing contributions to people such as food and feed, energy, medicines and genetic resources, and a variety of materials fundamental for people’s physical well-being and for maintaining culture (IPBES 2019). In the Province of Neuquén, activities or services encompassed in the Nature’s Contributions to People (NCP) concept are of great environmental and economic importance and provide opportunities for physically and psychologically beneficial activities, for healing, relaxation, leisure, and aesthetic enjoyment based on close contact with nature; therefore, the province watches over the preservation of its natural systems.

For this reason, Neuquén was the first province to implement actions against the potential occurrence and dispersion of an invasive exotic species (IES), with state agencies making the greatest effort to monitor *D. geminata* presence throughout the region. Since 2010, the General Directorate of Aquatic Biology (DGBA) and the Provincial Directorate of Water Control (DPRH) have worked to establish surveillance guidelines and to identify the presence of the invasive exotic alga *D. geminata* in aquatic environments to set up protected areas free of the presence of the alga, as well as actions to be implemented in provincial territory.

The first regional meeting coordinated by Neuquén was held with national, provincial, and municipal organizations as early as in 2010. It was concluded that it was essential to act with national organizations in order to accomplish actions that were effective both at the national and interprovincial jurisdiction levels. For the province of Neuquén, the situation was worrying due to the slow pace of administrative actions and the rapid advance of the invading alga. Various actions were taken to understand the dynamics of this IES and prevent its spread. Thus, in January 2011, Neuquén organized a training course delivered by specialists in periphyton from the National University of La Plata (UNLP), with the participation of all the local, provincial, and national water resource agents. The objective of this training was to establish a monitoring protocol including the methodology for taking biological and physicochemical samples and characterizing the habitat at relevant sites, environmental representatives of different areas of the basins, and potential habitats for the settlement and dispersal of *D. geminata*. In January 2012, the Ministry of Territorial Development (MDT) and the Ministry of Energy, Environment and Public Services of Neuquén (MEAySP) passed Resolution No. 24/12, which made it mandatory to disinfect every object coming into contact with provincial waters (such as fishing gear and boating equipment and machines), before carrying out sports, control, scientific, and technical activities. In this sense, Neuquén Province set up disinfecting stations at which the corresponding certificates were issued (Fig. 12.2).

Subsequently, the Provincial Emergency was declared due to the detection of the invasive alga in the Neuquén Province by Provincial Decree No. 137/12, and collaboration agreements were also concluded with municipalities for the installation of biosafety posts in these districts. Likewise, the DGBA carried out training and talks in the educational community and in the chambers that bring together tourist providers, fishermen, etc., in order to make the issue known, provide tools to prevent the spread of the invasive alga, and create awareness about the importance of the preservation of natural resources. This directorate, through the Secretariat of State, Environment and Sustainable Development of Neuquén (SEAyDS), promoted the generation of a provincial legal framework that would make it possible to highlight the problem of IES in aquatic environments and facilitate actions in this area. In December 2013, Provincial Law No. 2880 was sanctioned. Its first article establishes the multidisciplinary and interinstitutional monitoring and prevention of IES in aquatic environments as an environmental priority public policy, in order to prevent the settlement and dispersal of this species in waterbodies.

After 2 years of initiating the provincial actions, the First Argentinean Regional Meeting on the Evaluation of Regional Measures to Control the Expansion of *D. geminata* was held in San Carlos de Bariloche on April 17, 2012, organized by the National Secretariat of Environment, highlighting the delay of interjurisdictional actions that only favored the invasion of the alga. In this meeting, the creation of the Regional Technical Commission (CTR) was proposed to follow up on the problem. The CTR was formed by the environmental authorities of the provinces of Neuquén, Río Negro, Chubut, Santa Cruz, and Tierra del Fuego; the SAyDS; the representatives of the Interjurisdictional Authority of the Limay, the Neuquén and the Negro River basins; and the National Parks Administration. The declaration of the alga as invasive exotic species (IES) in the Argentine Republic (SAyDS Resolution No. 991/12, June 18, 2012) gave the official framework to the formation of the CTR working group – with the possibility to obtain fundings. The national

ALERTA HIDROBIOLÓGICA

Alga *Didymosphenia geminata*

Es OBLIGATORIO implementar las medidas de bioseguridad en todos los elementos que estén en contacto con el agua en territorio provincial, a fin de evitar su dispersión.

CUIDEMOS NUESTROS RÍOS Y LAGOS
EVITEMOS LA PROPAGACIÓN DE DIDYMO

Sr. Pescador o visitante:
Protejamos nuestros ambientes acuáticos de organismos invasores.

Help us protect our aquatic environment from invader organisms.

SUS WADERS, BOTAS DE VADEO Y EMBARCACIONES PUEDEN TRANSPORTAR ACCIDENTALMENTE ORGANISMOS INVASORES

YOUR WADERS, FORD BOOTS, AND WATERCRAFT CAN ACCIDENTALLY CARRY MICROSCOPIC INVADER ORGANISMS.

NEUQUÉN PROVINCIA | JUNTOS PODEMOS MÁS

El contacto o la ingesta no es perjudicial para la salud de las personas

SECRETARÍA DE DESARROLLO TERRITORIAL Y AMBIENTE
SUBSECRETARÍA DE AMBIENTE

El Alga Didymosphenia geminata, es una especie invasora de alto riesgo para la naturaleza, ha sido declarada Plaga Hidrobiológica a nivel internacional.

Por favor colabore cumpliendo las siguientes indicaciones de bioseguridad:

Please help us following the instructions

NEUQUÉN PROVINCIA | JUNTOS PODEMOS MÁS

"El contacto o la ingesta no es perjudicial para la salud de las personas"

1

REMOVER restos de algas, vegetación, barro, agua y sedimentos de los equipos y embarcaciones antes de abandonar lagos y ríos.

CHECKING before leaving rivers or lakes, remove any organic remains and sediments from your fishing gear and boat. Please dispose of them in trash cans (do not flush into the home sewage sys).

2

LAVAR todo lo que estuvo en contacto con el agua durante al menos un minuto en un recipiente de agua con lavandina al 2% (1 vaso cada 10 litros de agua), o sal al 5%(2 vasos cada 10 litros de agua) por 30 minutos.

CLEANING. Whatever might have been in contact with water should be soaked and rubbed for at least one minute in any of the following solutions:
I-Bleach at 2% (a small glass in 10 l of water)
II-Salt at 5% (500g of salt in 10 l of water) for at least 30 minutes.

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3

SECAR si lavar no es posible, dejar secar totalmente los equipos al sol y esperar 48hs. antes de volver a usar.

DRYING. When the cleaning process is not possible, drying is effective providing items become absolutely dry ( that is to say no humidity can be felt to the touch on their inside or outside surfaces. Once dry, gear can only be used after 48 additional hours have elapsed.

De este modo usted colabora con el "Plan de Prevención y Monitoreo de *Didymosphenia geminata* y evita la dispersión de este organismo exótico invasor en el país"

**Fig. 12.2** Some examples of control and educational measures of the Neuquén Province: a cleaning station located in Lonco Luan Place and a hydrobiological alert poster (triptych) about *D. geminata*

SAyDS convenes and coordinates the CTR meetings and leads the organization with the different jurisdictions. One of the tasks of the CTR was to elaborate and coordinate a regional strategy against *D. geminata*, which was periodically reviewed and updated along with an action plan focused on monitoring, biosecurity, communication, and legal framework. The SAyDS managed the collaboration in terms of budget and integrated work, among national organizations located in the provinces (National Gendarmerie, National Parks, Argentine Naval Prefecture, Customs, National Service of Health and Food Quality (SENASA), etc.), provincial entities, the Secretariat of Tourism, the General Directorate of Wildlife Resources and General Directorate of Natural Protected Areas, and interjurisdictional organisms.

As a consequence of the state of emergency declared by the Provincial Decree No. 0137/2012, since 2012, the Neuquén Province has developed the Surveillance and Monitoring Program by the DGBA and the DPRH. Eleven sampling campaigns were performed, in autumn (May) between 2012 and 2019 and in spring (November) between 2012 and 2014, at several sites on streams, rivers, and lakes (Table 12.1 and Fig. 12.3). In order to adjust to local conditions, the sampling protocol used by DGBA and DPRH was adapted from the EPA Rapid River and Stream Assessment Protocols (Barbour et al. 1999; Flotemersch et al. 2006). Sampling sites were selected taking into account human activities; the feasibility of the species arrival, settlement, and development; and the accessibility and the sociocultural value of the site. Different parameters were recorded at each sampling site: water temperature (Temp), pH, electrical conductivity (EC), dissolved oxygen (DO), current velocity (flow meter), and order of rivers and streams. The order of rivers and streams is a parameter that establishes a hierarchy defined as follows: rivers of the first order are those without tributaries, those of the second order are formed by joining first-order rivers, those of the third order are formed by joining second-order rivers, and so forth. The habitat index (PEH) at each site was additionally characterized according to protocols previously established by DGBA-DPRH (2012).

Water samples were taken for laboratory analysis: total and dissolved phosphorus (TP and SRP, respectively) and total nitrogen (TN); alkalinity; total suspended solids (SST); ions; Na, Cl, K, and Mg; and Fe, Si, and $SO_4$. All these determinations were performed according to APHA (1999) with the exception of Si, which was determined following Hach® method N°8185. Biological multi-habitat samples were collected by scrapping different periphytic substrates at each site. From these samples, we determined *D. geminata* presence/absence and quantitative information about *D. geminata* cells and the entire periphytic algal community using a Leica inverted microscope following the Utermöhl (1958) method. For further details on laboratory methods and determinations, see Beamud et al. (2019). Sections 3, 4, and 5 are a summary of the results obtained from data collected in the Surveillance and Monitoring Program between 2012 and 2019.

**Table 12.1** Sampling site abbreviations and names, locations, and main characteristics: basin, type of environment, order, and presence or absence (P/A) of *D. geminata*

| Sampling site code | Sampling site name | Latitude | Longitude | Basin | Type of environment | Order | DG |
|---|---|---|---|---|---|---|---|
| ACHACHI | Chapelco Chico small stream, Pio Proto | 40°07′15.90″ | 71°14′17.70″ | Hua Hum | Stream | 1 | P |
| ACHACHIARR | Chapelco Chico small stream, upstream Pio Proto | 40°7′6.80″ | 71°14′11.00″ | Hua Hum | Stream | 1 | A |
| ACHACHIGOL | Chapelco Chico small stream, Golf field | 40°7′32.00″ | 71°13′00.02″ | Hua Hum | Stream | 1 | A |
| ACODIHPTE | Codihue stream, bridge | 38°28′31.09″ | 70°29′37.09″ | Neuquén | Stream | 3 | P |
| AELONCINF | El Once stream | 40°47′04.60″ | 71°38′03,70″ | Nahuel Huapi | Stream | 1 | A |
| AGRANDINF | Grande stream at Lake Lacar inflow, Quila Quina site | 40°10′13.24″ | 71°26′48.28″ | Hua Hum | Stream | 1 | P |
| AMACHPAR | Machico stream, Port Arrayan | 40°36′47.19″ | 71°32′49.78″ | Limay | Stream | 1 | A |
| APOCAUINF | Pocahullo stream at Lake Lacar inflow | 40°09′34.60″ | 71°21′42.60″ | Hua Hum | Stream | 2 | P |
| ATRENTINF | Tren stream at Lake Lacar inflow, Catrite beach | 40°10′25.78″ | 71°24′10.66″ | Hua Hum | Stream | 1 | A |
| CQUILQPIO | Quilquihue River channel, Pio Proto ascent | 40°07′8.86″ | 71°14′13.51″ | Hua Hum | Channel | 3 | A |
| EPICHIBAL | Pichi Picun Leufu Reservoir, Piedra del Aguila City | 40°03′07.80″ | 70°01′52.30″ | Limay | Reservoir | 3 | A |
| LLACARCOS | Lake Lacar, coastal | 40°09′43.20″ | 71°21′30.10″ | Hua Hum | Lake | 2 | A |
| LNAHUECUM | Nahuel Huapi Lake, Cumelen Bay | 40°47′12.10″ | 71°37′2.91″ | NahuelHuapi | Lake | 1 | A |
| LTRAFUCAM | Traful Lake, Traful Village at camping site Paloma Araucana | 40°39′58.00″ | 71°21′25.70″ | Limay | Lake | 1 | A |
| LTRAFUCAT | Traful Lake, Catarata stream beach | 40°37′37.10″ | 71°29′37.30″ | Limay | Lake | 1 | P |
| LTRAFUCTO | Traful Lake, the piers (Village Center) | 40°39′15.00″ | 70°23′55.10″ | Limay | Lake | 1 | P |

(continued)

**Table 12.1** (continued)

| Sampling site code | Sampling site name | Latitude | Longitude | Basin | Type of environment | Order | DG |
|---|---|---|---|---|---|---|---|
| LTRAFUMAQ | Traful Lake, La Maquina Beach | 40°39′52.60″ | 71°21′36.30″ | Limay | Lake | 1 | A |
| LTRAFUMAR | Traful Lake, Las Tres Marías Beach | 40°38′45.50″ | 71°26′19.30″ | Limay | Lake | 1 | A |
| LTRAFUPCH | Traful Lake, Pichi Traful | 40°30′00.60″ | 71°34′58.30″ | Limay | Lake | 2 | A |
| LTRAFUPTO | Traful Lake, Port Arrayan | 40°36′53.60″ | 71°32′49.10″ | Limay | Lake | 1 | A |
| RAGRIOHUA | Agrio River, Huarenchenque stream inflow | 38°16′45.14″ | 70°36′30.82″ | Neuquén | River | 3 | P |
| RAGRIOLAJ | Agrio River, upwaters Las Lajas bridge | 38°31′03.80″ | 70°21′43.00″ | Neuquén | River | 4 | P |
| RAGRIOLON | Agrio River, Loncopue | 38°04′35.80″ | 70°36′34.40″ | Neuquén | River | 3 | P |
| RALUMIALU | Alumine River, Alumine | 39°14′00.20″ | 70°54′42.90″ | Limay | River | 4 | P |
| RALUMIGAU | Alumine River, Piedra Gaucha site | 39°08′12.90″ | 70°57′05.60″ | Limay | River | 4 | P |
| RALUMINAC | Alumine River, headwaters | 38°58′3.63″ | 71°3′1.44″ | Limay | River | 3 | P |
| RALUMPILO | Alumine River, Pilo Lilplace | 39°33′17.40″ | 70°57′27.00″ | Limay | River | 4 | P |
| RCALEUHIT | Caleufu River, milestone | 40°23′52.40″ | 70°44′11.80″ | Limay | River | 3 | P |
| RCATAINF | Lower Catan Lil River | 39°53′19.40″ | 70°50′14.80″ | Limay | River | 3 | P |
| RCCURABAL | Collon Cura River, Balsa Vieja | 40°07′07.10″ | 70°44′57.00″ | Limay | River | 5 | P |
| RCCURARIN | Collon Cura, La Rinconada bridge | 39°59′55.00″ | 70°50′00,7″ | Limay | River | 4 | P |
| RCHIMEJUN | Chimehuin River, Junin de los Andes City EPAS | 39°58′46.40″ | 71°04′11.50″ | Limay | River | 3 | P |
| RCHIMERIS | Chimehuin River, Tipiliuque cliffs | 40°06′22.80″ | 70°57′27.00″ | Limay | River | 4 | P |
| RCHIMESUP | Chimehuin River mouth | 39°47′49.90″ | 71°12′39.70″ | Limay | River | 3 | A |
| RCHIMETIP | Chimehuin River, Tipiliuke | 40°06′23.25″ | 70°58′09.33″ | Limay | River | 4 | P |
| RCOLOPTON | Colorado River, El Porton | 37°07′36.70″ | 68°38′47.60″ | Colorado | River | 4 | A |
| RCOLORDLS | Colorado River, Rincón de los Sauces | 37°22′47.80″ | 68°54′37.30″ | Colorado | River | 4 | A |
| RCORREPTE | Correntoso River | 40°44′11.10″ | 71°40′25.80″ | Limay | River | 2 | A |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| RCURCHOSM | Curi Leuvu River, downwaters Chos Malal bridge | 37°22′14.00″ | 70°16′32.00″ | Neuquén | River | 3 | A |
| RCURRUSUP | Currhue River, headwaters | 39°54′14.90″ | 71°20′06.40″ | Limay | River | 1 | A |
| RCUYININF | Cuyin Manzano River, downwaters outflow | 40°42′53.00″ | 71°07′30.80″ | Limay | River | 2 | A |
| RFHHUMINF | Meliquina River, Filo Hua Hum | 40°28′51.90″ | 71°13′36.40″ | Limay | River | 2 | P |
| RGUAÑAPRE | Guañacos River, Guañacos | 37°16′52.10″ | 70°41′26.40″ | Limay | River | 1 | A |
| RHERMOINF | Lower Hermoso River | 40°20′23.60″ | 71°20′24.70″ | Limay | River | 2 | P |
| RHERMOSUP | Hermoso River, headwaters | 40°21′17.70″ | 71°28′26.00″ | Limay | River | 1 | A |
| RHUAHUPTE | Hua Hum River, bridge | 40°07′10.60″ | 71°39′43.50″ | Hua Hum | River | 3 | P |
| RLILEOPTE | Lileo River | 37°13′28.40″ | 70°45′36.60″ | Neuquén | River | 2 | A |
| RLIMAYARR | Limay River, near Arroyito | 39°5′31.10″ | 68°34′30.00″ | Limay | River | 5 | A |
| RLIMAYLLA | Limay River, Llanquin village | 40°53′40.00″ | 71°02′23.10″ | Limay | River | 4 | P |
| RLIMAYPRE | Limay River, before confluence with Neuquén River | 38°58′43.30″ | 68°02′25.20″ | Neuquén | River | 5 | A |
| RLIMAYSUP | Limay River, headwaters | 41°03′23.90″ | 70°21′43.00″ | Limay | River | 4 | P |
| RLIMAYTAU | Middle Limay River, downwaters Picun Leufu | 39°48′42.90″ | 69°36′54.20″ | Limay | River | 2 | P |
| RMALLECON | Malleo River, confluence with Alumine River | 39°49′42.20″ | 70°53′00,50″ | Limay | River | 2 | P |
| RMALLEINF | Malleo River, Route 23 | 39°46′52.40″ | 71°02′10.90″ | Limay | River | 2 | P |
| RMALLESUP | Upper Malleo River | 39°36′47.40″ | 71°20′49.50″ | Limay | River | 1 | P |
| RMELIQPTE | Meliquina River, headwaters | 40°22′58.90″ | 71°15′56.80″ | Limay | River | 2 | A |
| RMINERINF | Minero River, outflow | 40°40′08.50″ | 71°15′26.00″ | Limay | River | 1 | A |
| RNAHUENAC | Headwaters Nahueve River | 36°50′58.03″ | 71°0′49.09″ | Neuquén | River | 1 | A |

(continued)

**Table 12.1** (continued)

| Sampling site code | Sampling site name | Latitude | Longitude | Basin | Type of environment | Order | DG |
|---|---|---|---|---|---|---|---|
| RNAHUESUP | Upper Nahueve River | 36°59′14.30″ | 70°51′34.40″ | Neuquén | River | 3 | P |
| RNEUANDAC | Neuquén River, Andacollo | 37°10′40.02″ | 70°39′27.06″ | Neuquén | River | 4 | A |
| RNEUBALLE | Neuquén River, Vista Alegre | 38°43′49.60″ | 68°10′6.50″ | Neuquén | River | 2 | A |
| RNEUBANDE | Neuquén River, Banderita plain | 38°33′30.69″ | 68°28′10.89″ | Neuquén | River | 2 | A |
| RNEUBARTU | Neuquén River, El Sauce downwaters Añelo | 38°24′3.42″ | 68°42′8.09″ | Neuquén | River | 2 | A |
| RNEUCHINF | Neuquén River, downwaters Chañar dam | 38°36′1.80″ | 68°23′3.19″ | Neuquén | River | 2 | A |
| RNEUCHOSM | Neuquén River, Chos Malal | 37°24′24.00″ | 70°13′45.00″ | Neuquén | River | 4 | A |
| RNEUCHSUP | Neuquén River, upwaters Chañar dam | 38°33′41.30″ | 68°26′21.00″ | Neuquén | River | 2 | A |
| RNEUESQUI | Neuquén River, Cuatro Esquinas place | 38°54′16.47″ | 68°4′2.05″ | Neuquén | River | 5 | A |
| RNEUFIGUE | Neuquén River, Figueroa street Neuquén City | 38°57′33.80″ | 68°00′28.00″ | Neuquén | River | 5 | A |
| RNEUHUITR | Neuquén River, Huitrin balsa | 37°39′53.27″ | 69°58′58.82″ | Neuquén | River | 5 | A |
| RNEUINDIO | Neuquén River, Paso de los Indios | 38°32′43.1″ | 69°23′49.8″ | Neuquén | River | 5 | A |
| RNEUMANZA | Neuquén River, Manzana Amargo place | 36°45′01.00″ | 70°45′38.00″ | Neuquén | River | 3 | P |
| RNEUQUPRE | Neuquén River before confluence with Limay River | 38°59′28.60″ | 68°00′05.90″ | Neuquén | River | 5 | A |
| RNEUSAUZA | Neuquén River, Sauzal Bonito | 38°36′48.70″ | 69°08′44.40″ | Neuquén | River | 5 | P |
| RNEUTOMAÑ | Neuquén River, Añelo EPAS | 38°22′41.10″ | 68°48′10.79″ | Neuquén | River | 2 | A |
| RNEUTRATA | Neuquén River, Tratayen bridge | 38°28′07.00″ | 68°35′34.00″ | Neuquén | River | 2 | A |
| RPULMAINF | Lower Pulmari River | 39°05′42.90″ | 70°57′52.70″ | Limay | River | 3 | P |
| RPULMASUP | Upper Pulmari | 39°08′38.40″ | 71°13′47.40″ | Limay | River | 3 | P |
| RQUILLINF | Lower Quillen River | 39°22′42.80″ | 70°56′39.30″ | Limay | River | 2 | P |
| RQUILLSUP | Upper Quillen River | 39°21′47.70″ | 71°13′11.70″ | Limay | River | 2 | A |

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| RQUILQAER | Quilquihue River, airport | 40°04′47.30″ | 71°09′19.40″ | Limay | River | 3 | A |
| RQUILQGOL | Quilquihue River, Golf field | 40°05′37.30″ | 71°12′22.50″ | Limay | River | 3 | P |
| RQUILQINF | Quilquihue River, Chimehuin outflow | 40°03′33.00″ | 71°04′58.50″ | Limay | River | 3 | P |
| RQUILQPTE | Quilquihue River, Route 40 bridge | 40°03′27.00″ | 71°04′39.40″ | Limay | River | 3 | P |
| RQUILQSUP | Upper Quilquihue River (headwaters) | 40°04′12.00″ | 71°18′57.00″ | Limay | River | 3 | A |
| RQUILQTOMA | Quilquihue River, EPAS | 40°05′37.40″ | 71°15′01.60″ | Limay | River | 3 | A |
| RRCHORINF | Lower Ruca Choroi River (outflow) | 39°12′23.00″ | 70°55′21.50″ | Limay | River | 1 | P |
| RRCHORSUP | Upper Ruca Choroi | 39°13′30.20″ | 71°10′10.10″ | Limay | River | 1 | A |
| RREÑILERP | Reñileuvu River | 37°20′07.60″ | 70°43′17.60″ | NeuquénNeuquén | River | 2 | A |
| RTRAFUINF | Traful River, downwaters Cuyin Manzano River | 40°42′53.80″ | 71°06′48.20″ | Limay | River | 3 | P |
| RTROCOMRP | Trocoman River | 37°22′13.60″ | 70°42′43.00″ | NeuquénNeuquén | River | 3 | P |
| RVARVVARV | Varvarco River, upwaters Varvarco village | 36°50′33.00″ | 70°40′38.00″ | NeuquénNeuquén | River | 3 | A |
| RVILLAPTE | Villarino River, Falkner | 40°26′44.70″ | 71°32′33.00″ | Limay | River | 1 | A |

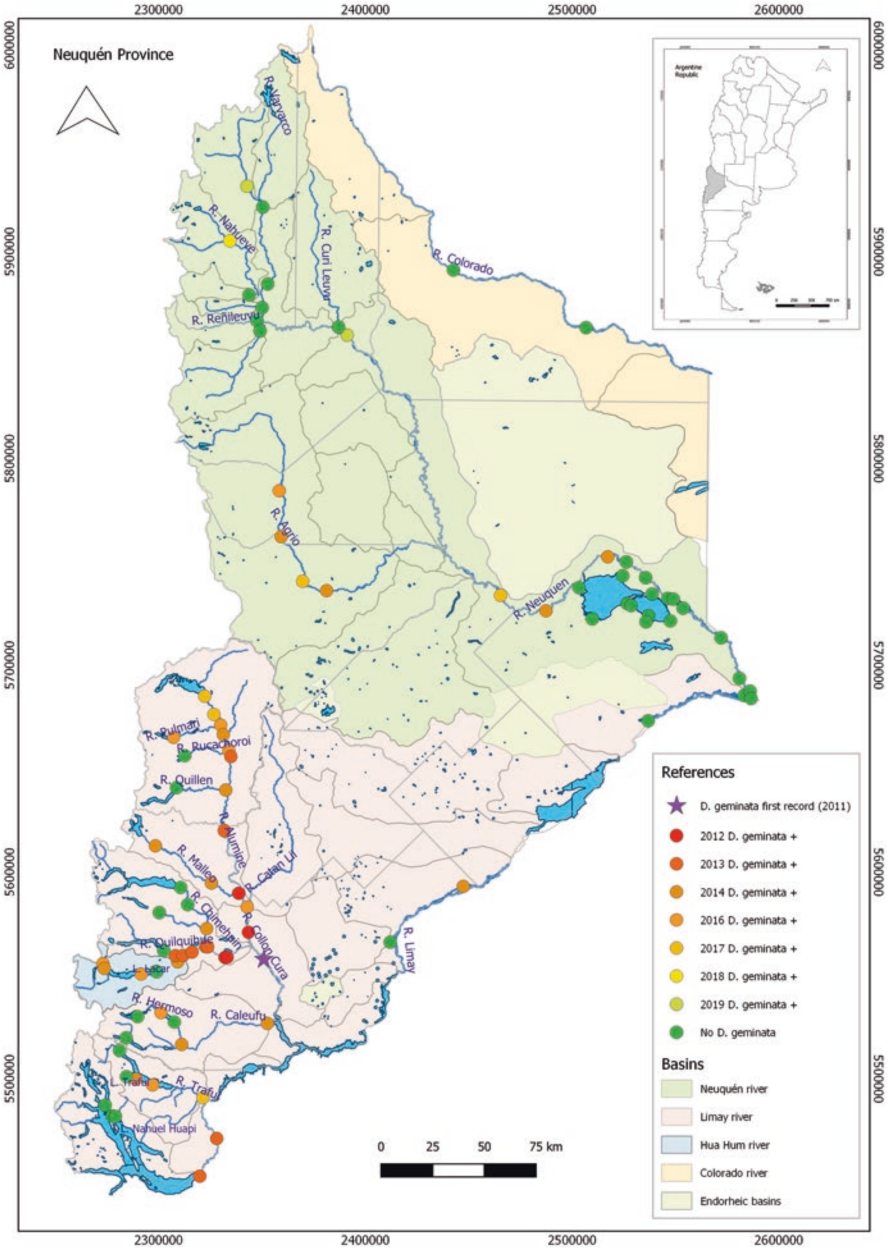

**Fig. 12.3** Sampling sites in Neuquén Province (Argentina) surveyed by the Surveillance and Monitoring Program (11 sampling campaigns). The colors of the circles show the dispersion of *D. geminata* between years 2012 and 2018. (Modified from Beamud et al. (2019))

## 3 Dispersion of *D. geminata* in the Province

*D. geminata* was detected at 40 out of the 93 sites sampled within the Neuquén Province (Table 12.1 and Fig. 12.3), including streams, lakes, and rivers. The average relative abundance (%, between 2012 and 2019) of cells at sampling sites in the province was 1.5%, ranging between 0.01% and 9.2%. Although it is not a high abundance, these values coincide with those observed in a previous study in the Restigouche River basin (Quebec, Canada), where abundance of *D. geminata* cells was <3%, even at sites with extensive *D. geminata* stalk material (Gillis and Lavoie 2014; Sivarajah et al. 2015).

The pattern of invasion of the alga from its first appearance in 2011 is shown in Fig. 12.3. The first site where *D. geminata* was detected was the Collon Cura River (Balsa Vieja) in October 2011, and in April 2012, it appeared in the Chimehuin River. In November 2013, the alga was detected in the Quilquihue and Alumine Rivers (near the City of Alumine) and in the headwaters of the Limay River. Between April and June 2014, its presence was determined in the Malleo River and the Chapelco Stream and, at the end of 2014, in the Traful Lake and the Meliquina and Caleufu Rivers. In 2015, there was no new detection of the species in the province. Between March and May 2016, it appeared in the Hua Hum River, in the Grande stream (tributary of Lake Lacar), and in the Hermoso River. In 2017, it was detected at a single new site, in the headwaters of the Alumine River; and in 2018, it was detected in the Nahueve River and the outlet of the Pocahullo stream in Lake Lacar. In 2019, the alga appeared in the Neuquén and Trocoman Rivers in the Neuquén basin. At most of these sampling sites except for the Upper Malleo River, the Chapelco Stream, the Meliquina River, the Caleufu River, and the Grande and Pocahullo streams, *D. geminata* presence is noticeable at first sight. Most sites with *D. geminata* are located in western Neuquén, the so-called Andean zone, belonging to the basins of Limay and Hua Hum Rivers (Fig. 12.3 and Table 12.1). Most environments with *D. geminata* are related to recreational activities carried out in the rivers and lakes in the area (fly-fishing, rafting, kayaking, extreme competitions, etc.).

There are few sites with *D. geminata* in northern and eastern Neuquén, wherein it appeared 3 years after the first detection in the Collon Cura River in 2011. While the northern area of the province has a more arid climate and higher altitude, the central-east zone is the most affected by land farming activities, and the one with higher population density, since the largest cities of the province are in this region (Colorado and Neuquén river basins). Also, river orders are higher (contrarily to mountain rivers in the Andean zone) (Table 12.1). Therefore, these factors result in waterbodies wherein nutrients, conductivity, and temperature are higher (Table 12.2) and *D. geminata* was absent or its presence was delayed.

**Table 12.2** Main chemical and physical characteristics, habitat quality, and *D. geminata* abundance in sampling basins

| Variables | Units | Colorado | Hua Hum | Limay | Nahuel Huapi | Neuquén |
|---|---|---|---|---|---|---|
| SST | mg L$^{-1}$ | 164.1 (± 80.3) | 6.6 (± 5.9) | 16.5 (± 49.1) | 35.1 (± 0.7) | 60.3 (± 102.9) |
| Alk | mg L$^{-1}$ | 101.8 (± 29.3) | 42.3 (± 17.9) | 26.2 (± 15.0) | 24.9 (± 15.5) | 42.6 (± 19.7) |
| Ca | mg L$^{-1}$ | 100.5 (± 25.7) | 11.1 (± 4.6) | 9.2 (± 16.6) | 5.1 (± 2.2) | 22.1 (± 15.7) |
| Mg | mg L$^{-1}$ | 9.6 (± 1.9) | 2.8 (± 1.3) | 1.8 (± 1.4) | 1.9 (± 1.4) | 3.1 (± 1.7) |
| TN | μg L$^{-1}$ | 158.4 (± 36) | 206.9 (± 321.4) | 92.1 (± 153.6) | 77.3 (± 37.1) | 103.8 (±52.6) |
| SRP | μg L$^{-1}$ | 3.2 (± 2.3) | 21.0 (± 37.0) | 5.1 (± 8.9) | 3.0 (± 3.4) | 8.2 (± 7.5) |
| TP | μg L$^{-1}$ | 190.9 (± 121.8) | 44.1 (± 54.9) | 32.3 (± 48.3) | 21.4 (± 3.6) | 84.8 (± 110.7) |
| SO4 | mg L$^{-1}$ | 132.3 (± 70.1) | 0.3 (± 0.9) | 5.8 (± 26.9) | 0.7 (± 0.9) | 24.7 (± 23.9) |
| Fe | mg L$^{-1}$ | 0.06 (± 0.05) | 0.07 (± 0.07) | 0.05 (± 0.08) | 0.07 (± 0.07) | 0.07 (± 0.07) |
| Si | mg L$^{-1}$ | 18.3 (± 0.3) | 18.4 (± 5.0) | 14.7 (± 3.5) | 14.9 (± 10.5) | 15.7 (± 7.7) |
| EC | μS cm$^{-1}$ | 1106.5 (± 27.2) | 92.4 (± 31.6) | 74.9 (± 89.3) | 68.1 (± 41.7) | 204.4 (± 143.5) |
| DO | mg L$^{-1}$ | 8.9 (± 0.2) | 9.7 (± 0.9) | 9.6 (± 0.4) | 9.7 (± 0.4) | 9.7 (± 0.7) |
| Veloc | m s$^{-1}$ | 1.3 (± 0.02) | 0.5 (± 0.33) | 0.8 (± 0.33) | 0.5 (± 0.00) | 0.9 (± 0.41) |
| Temp | °C | 14.8 (± 1.6) | 11.3 (± 2.8) | 12.4 (± 1.5) | 10.9 (± 2.4) | 12.9 (± 2.9) |
| pH | | 8.1 (± 0.1) | 7.7 (± 0.3) | 7.6 (± 0.3) | 7.3 (± 0.0) | 7.9 (± 0.4) |
| PEH | | 95 (± 19.1) | 123.1 (± 34.7) | 126.5 (± 26.1) | 70.9 (± 0.0) | 121.3 (± 21.7) |
| DG | % | 0 (± 0.00) | 0.04 (± 0.08) | 0.98 (± 2.35) | 0 (± 0.0) | 0.37 (± 1.37) |

The values represent averages (± SD) of sites within basins from 2012 to 2018

*SST* total suspended solids, *Alk* alkalinity, *Ca* calcium, *Mg* magnesium, *TN* total nitrogen, *SRP* soluble reactive phosphorus, *TP* total phosphorus, *SO4* sulfates, *Fe* total iron, *Si* silica, *EC* electrical conductivity, *DO* dissolved oxygen, *Veloc* current velocity, *Temp* temperature, *PEH* habitat quality index, *DG D. geminata* relative abundance

## 4 Ecological Preferences of *D. geminata*

The factors that control the occurrence and abundance of *D. geminata* in the Neuquén Province were evaluated in previous studies between the years 2012 and 2017 (Beamud et al. 2019). It was found that the occurrence (presence/absence) and abundance of the species were controlled by different environmental variables, the order of the environments being important for both algal parameters. Thus, the presence or absence (P/A) of *D. geminata* was controlled by order, Mg and SO4 concentrations, alkalinity, DO content, and water temperature. On the other hand, for the abundance of the species, the only significantly important variable was the order of the environments. So, in Neuquén Province, the distribution and growth of *D. geminata* are not associated with the factors typically described in previous studies, including water velocity, Fe, and dissolved and total P and N (Bothwell et al. 2012, 2014; James et al. 2014; Cullis et al. 2015; Bray et al. 2016b; Sanmiguel et al. 2016). Jackson et al. (2015) also found that the abundance of *D. geminata* in systems of British Columbia (Canada) was negatively related to stream order. This relationship is not surprising since the order of the systems is a variable that summarizes a lot of information regarding environment water quality (Wetzel 2001): the lower the order, the higher the system quality is.

In this new instance of analysis, we intend to explore the characteristics of the basins sampled within the Monitoring Program. With this aim, the sites sampled were ordered using the multivariate method Principal Component Analysis (PCA) to find a configuration to observe similarities among basins by means of the environmental data (variables) measured during the samplings. PCA was performed using CANOCO 5.0 for Windows (ter Braak and Smilauer 2012). From the results, we could observe that the basins in which *D. geminata* was present (stars) or absent (circles) separate in the ordination diagram according to their physicochemical characteristics (Fig. 12.4). A gradient was observed from the basins that have better habitat quality (PEH index) such as the Limay, the Hua-Hum, and the Nahuel Huapi basins to the Neuquén and Colorado river basins with higher values of nutrients, ions, conductivity (EC), pH, and alkalinity (alk) (Fig. 12.3). Within this gradient,

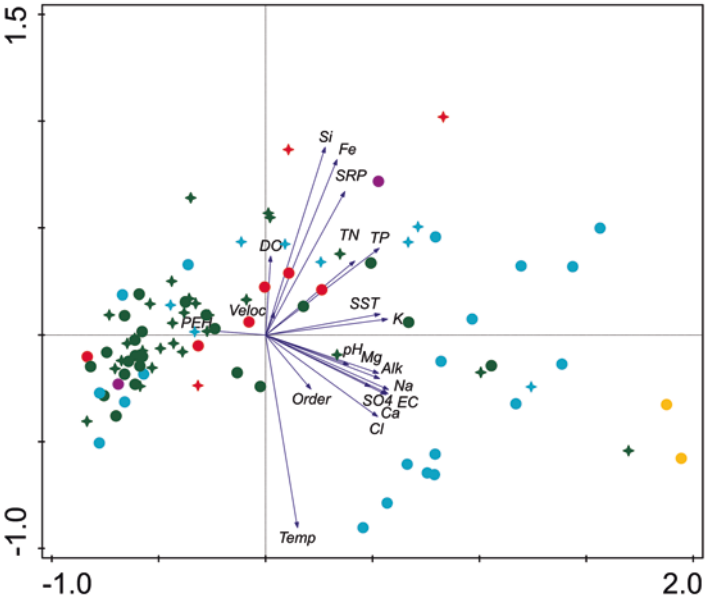

**Fig. 12.4** Principal Components Analysis (PCA) biplot with environmental variables. Each point corresponds to a sampling site in rivers and lakes from 2012 to 2018. Circles: sites without *D. geminata*; stars: sites with *D. geminata*. Red, Hua Hum River basin; blue light, Neuquén River basin; yellow, Colorado River basin; dark green, Limay River basin; and violet, Nahuel Huapi Lake basin. The environmental variation explained by the first two axes of the PCA was 63.9%

*D. geminata* is present mainly in the most Andean basins since the beginning of its invasion, in systems that fall within the environmental tolerance range of the species (Kilroy et al. 2005; Spaulding and Elwell 2007), as they were previously classified as oligotrophic to mesotrophic, with circumneutral and well-oxygenated waters (Diaz et al. 2007). Also, the basins wherein *D. geminata* was present were low-order and high current velocity (Veloc) systems, with the lowest values of most variables in the basins (Table 12.2 and Fig. 12.4).

Once the diatom invades an environment, other factors come to be important for it to become an abundant or predominant species or even form massive blooms. There is an overlap of sites and basins with and without *D. geminata* (see Fig. 12.4) located in the Andean region. This overlap means that those sites and basins share the same environmental characteristics which favor the spread of the alga. Hence, the sites still free from *D. geminata* could be considered as more susceptible to future invasion by this alga.

Kilroy and Bothwell (2011) have described a mechanism for bloom formation linked to dissolved P: cell division is limited by P but not stalk production. This can lead to massive growths occurring at dissolved P values less than 2 μg L$^{-1}$ (Bothwell et al. 2014). According to Bray et al. (2017), above that P concentration, not only *D. geminata* would not grow excessively but would persist only as a minor component of periphyton. Kunza et al. (2018) maintained this threshold value for SRP and *D. geminata* success, but they also discussed the possibility of a modification since several studies have dated algal blooms to higher SRP values.

Although sites where *D. geminata* was present in the Neuquén Province, SRP ranged between <2 and 118 μg P L$^{-1}$ (Fig. 12.5); most of *D. geminata* blooms were

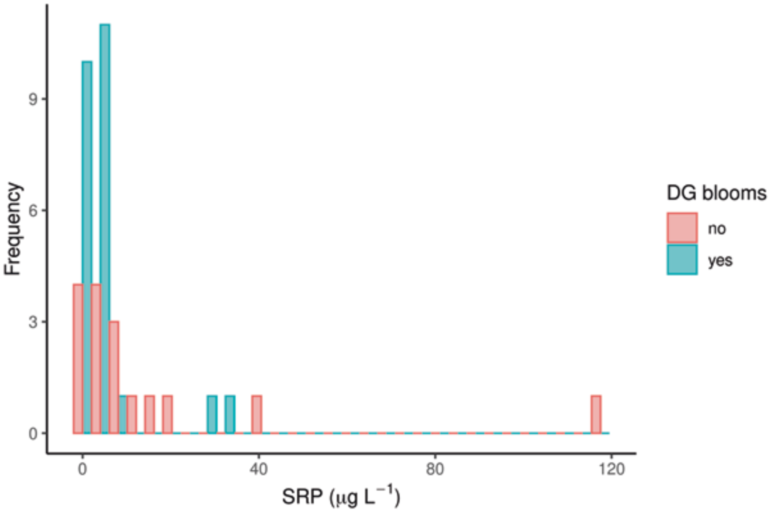

**Fig. 12.5** Histogram of frequency of *D. geminata* (forming blooms or not) and phosphorus values (SRP, μg P L$^{-1}$) at sampling sites

recorded approximately between 0.5 and 5 μg P $L^{-1}$ (Fig. 12.5). Nevertheless, there were rivers such as the Collon Cura, the Agrio, and the Malleo, where *D. geminata* cells were present and blooms occurred at SRP concentrations >5, >10, and >30 μg P $L^{-1}$ (Fig. 12.5), respectively, exceeding the limits previously described for the blooms by West et al. (2020). For the rest of Neuquén environments, the rule seems to be fulfilled because, although the alga has been found at values higher than 2 μg P $L^{-1}$, it does not form massive blooms. The Patagonian environment that also seems to escape from all ranges is Lake Nahuel Huapi (on the Río Negro Province shore), wherein massive growths of *D. geminata* have been previously observed at dissolved P values of 24.6 μg P $L^{-1}$ (Beamud et al. 2016) well above the threshold values cited by Bothwell and Kilroy (2011). These results confirm that *D. geminata* is expanding its ecological spectrum, showing a wider tolerance to environmental conditions, such as P concentrations, and that there are other environmental factors critical for the occurrence and growth of this species. Nevertheless, it has been reported that invasive species show high phenotypic plasticity at new environmental conditions offering them an extra advantage over native species (Davidson et al. 2011; Sassenhagen et al. 2015). Therefore, this expansion of its environmental preference ranges could modify the importance of certain previously studied variables (such as P, temperature, and current velocity, among others) subsequently modifying the environments susceptible to being invaded by *D. geminata*.

## 5 Algal Diversity Related to *D. geminata*

The diversity of periphytic communities including *D. geminata* as an invasive species shows contrasting situations worldwide. Spaulding et al. (2010) pointed out that the presence of *D. geminata* stalks would lead to more homogeneous algal associations which would reduce community diversity. This situation was observed in Chile (Salvo Pereira and Oyane del Pérez 2019) and in New Zealand (Bray et al. 2020), where *D. geminata* influenced algal composition by decreasing diversity and homogenizing communities. By contrast, Kilroy et al. (2009), Gillis and Lavoie (2014), and Sanmiguel et al. (2016) found a greater diversity of diatoms and an increase in periphyton biomass associated with *D. geminata* massive mat formations. Ladrera et al. (2014, 2018) disagree with those authors because they postulate that following massive *D. geminata* growth, a community dominated by small organisms is created, causing a significant reduction in the number of diatom species with similar ecological requirements that is clearly different from the previous river community structure. Meanwhile, in an Andean basin in Chile, Figueroa et al. (2018) found that when *D. geminata* coverage was greater than 50%, the specific richness increased with respect to the non-invaded site, while when the coverage was less than 30%, the richness was similar between both sites. In addition, these authors found a positive relationship between *D. geminata* biomass and the relative abundance of other small diatoms.

In the case of the Neuquén Province, the algal diversity and cell abundance at sites with or without *D. geminata* were also studied from different approaches during previous research projects. The algal community was described in detail by Beamud et al. (2019) and was represented by diatom species, chlorophytes, and cyanobacteria (see Table S3 in online supplementary material from Beamud et al. (2019) and Table 12.3 of this chapter). At sites wherein *D. geminata* was not abundant, both chlorophytes and cyanobacteria were present with a greater number of

**Table 12.3** Names, taxonomic affiliation, and abbreviations of periphytic species

| Species name | Taxonomic group | Species abbreviation in NMDS biplot |
|---|---|---|
| *Achnanthes* sp. | Bacillariophyceae | *Achnanth* |
| *Achnanthidium minutissimum* | Bacillariophyceae | *AchnanthMin* |
| *Aphanothece* sp. | Cyanophyceae | *Aphanoth* |
| *Asterionella* sp. | Bacillariophyceae | *Asterion* |
| *Audouinella* sp. | Florideophyceae | *Audouinl* |
| *Aulacoseira* sp. | Bacillariophyceae | *Aulacose* |
| *Calothrix* sp. | Cyanophyceae | *Calothrix* |
| *Chaetophora* sp. | Chlorophyceae | *Chaetoph* |
| *Chroococcal* | Cyanophyceae | *Chroococ* |
| *Closterium* sp. | Conjugatophyceae | *Closteri* |
| *Cocconeis* sp. | Bacillariophyceae | *Cocconei* |
| *Cosmarium* sp. | Conjugatophyceae | *Cosmariu* |
| *Cyclotella stelligera* | Bacillariophyceae | *Cyclotel* |
| *Cymatopleura* sp. | Bacillariophyceae | *Cymatopl* |
| *Cymbella* sp. | Bacillariophyceae | *Cymbella* |
| *Diatoma* sp. | Bacillariophyceae | *DiatmSp* |
| *Didymosphenia geminata* | Bacillariophyceae | *Didymosp* |
| *Diploneis* sp. | Bacillariophyceae | *Diplonei* |
| *Dolichospermum* sp. | Cyanophyceae | *Dolichosp* |
| *Epithemia* sp. | Bacillariophyceae | *Epithemi* |
| *Euastrum* sp. | Conjugatophyceae | *Euastrum* |
| *Eunotia* sp. | Bacillariophyceae | *Eunotia* |
| *Fragilaria* sp. | Bacillariophyceae | *Fragilar* |
| *Frustulia* sp. | Bacillariophyceae | *Frustuli* |
| *Gloeocapsa* sp. | Cyanophyceae | *Gloeocap* |
| *Gloeothece* sp. | Cyanophyceae | *Gloeothc* |
| *Gomphoneis herculeana* | Bacillariophyceae | *Gomphone* |
| *Gomphonema* sp. | Bacillariophyceae | *Gomphonm* |
| *Gyrosigma* sp. | Bacillariophyceae | *Gyrosigm* |
| *Hannaea arcus* | Bacillariophyceae | *Hannaea* |
| *Lyngbya* sp. | Cyanophyceae | *Lyngbya* |
| *Melosira* sp. | Bacillariophyceae | *Melosira* |
| *Meridion* sp. | Bacillariophyceae | *Meridion* |
| *Merismopedia* sp. | Cyanophyceae | *Merismop* |

(continued)

**Table 12.3** (continued)

| Species name | Taxonomic group | Species abbreviation in NMDS biplot |
|---|---|---|
| *Micrasterias* sp. | Conjugatophyceae | *Micrastr* |
| *Microcoleus* sp. | Cyanophyceae | *Microcol* |
| *Microcystis* sp. | Cyanophyceae | *Microcys* |
| *Monoraphidium* sp. | Chlorophyceae | *Monoraph* |
| *Mougeotia* sp. | Conjugatophyceae | *Mougeoti* |
| *Navicula* sp. | Bacillariophyceae | *Navicula* |
| *Nitzschia* sp. | Bacillariophyceae | *Nitzschi* |
| *Nostoc* sp. | Cyanophyceae | *Nostoc* |
| *Oedogonium* sp. | Chlorophyceae | *Oedogoni* |
| *Oocystis* sp. | Trebouxiophyceae | *Oocystis* |
| *Oscillatoria* sp. | Cyanophyceae | *Oscillat* |
| *Pandorina* sp. | Chlorophyceae | *Pandorin* |
| *Pediastrum* sp. | Chlorophyceae | *Pediastr* |
| *Phormidium* sp. | Cyanophyceae | *Phormidi* |
| *Pinnularia* sp. | Bacillariophyceae | *Pinnular* |
| *Rhoicosphenia abbreviata* | Bacillariophyceae | *Rhoicosp* |
| *Rhopalodia* sp. | Bacillariophyceae | *Rhopalod* |
| *Selenastrum* sp. | Chlorophyceae | *Selenast* |
| *Sphaerocystis* sp. | Cyanophyceae | *Sphaeroc* |
| *Spirogyra* sp. | Conjugatophyceae | *Spirogyr* |
| *Spirulina* sp. | Cyanophyceae | *Spirulin* |
| *Staurastrum* sp. | Conjugatophyceae | *Staurast* |
| *Surirell* | Bacillariophyceae | *Surirella* |
| *Synedra* sp. | Bacillariophyceae | *Synedra* |
| *Tabellaria* sp. | Bacillariophyceae | *Tabellar* |
| *Ulothrix* sp. | Ulvophyceae | *Ulothrix* |
| *Zygnema* sp. | Conjugatophyceae | *Zygnema* |

species. According to these authors, the Shannon's species diversity (H) did not show a clear trend or pattern, since H appeared as a poorly represented variable in the PCA ordination diagram. Figure 12.6 shows an ordination (nonmetric multidimensional scaling (NMDS)) of the sampling sites and basins, with and without *D. geminata*, in relation to the similarity in periphyton species composition. NMDS was performed using CANOCO 5.0 for Windows (ter Braak and Smilauer 2012). It could be observed that some basins such as the Limay, the Nahuel Huapi, and the Colorado have sites that are mixed and overlap, showing a similarity in algal diversity, despite the presence or absence of *D. geminata*. On the other hand, in the Hua Hum and the Neuquén basins, the sites wherein *D. geminata* is present are more scattered and show similarities with the sites of the Limay River, where the species is also present. When the diversity of the periphyton community is evaluated over the elapsed time since *D. geminata* invasion, no clear pattern of change is observed after t-test comparisons at each sampled site (Fig. 12.7). Sites where *D. geminata*

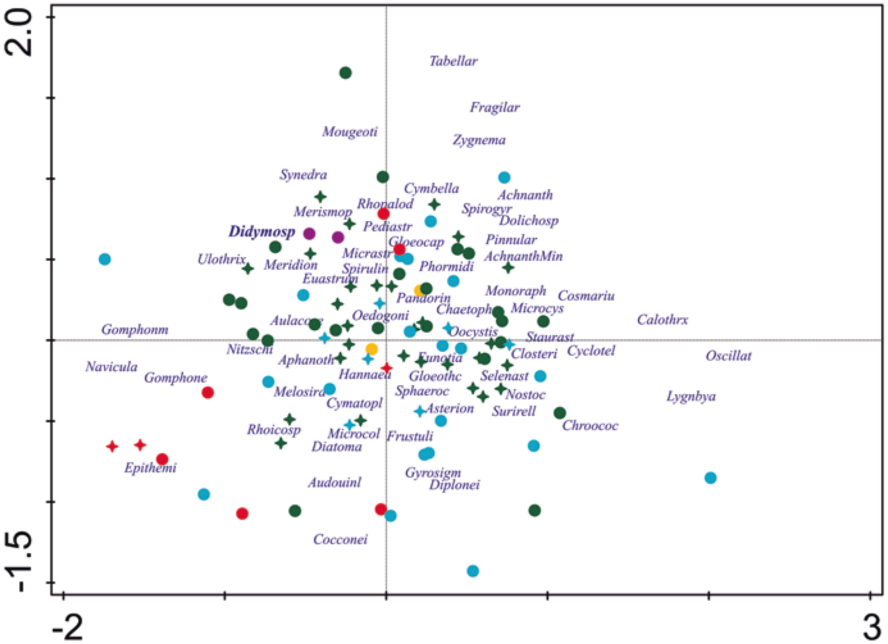

**Fig. 12.6** Nonmetric multidimensional scaling (NMDS) biplot based on periphyton composition. Each point corresponds to a sampling site. Circles, sites without DG; stars, sites with DG. Red, Hua Hum River basin; blue light, Neuquén River basin; yellow, Colorado River basin; dark green, Limay River basin; and violet, Nahuel Huapi Lake basin. Stress value: 0.184. Species names' abbreviations are shown in Table 12.3

had been present for less than a year, and for over 6 years, showed no statistical differences in diversity pre- and post-detection. From this result, we could think that after eight years of monitoring the invasive species at some sites, the presence of *D. geminata* did not cause differences in algal diversity. t-Test comparisons were performed within the R statistical environment (R Core Team 2020).

## 6 Conclusion

The data collected so far allowed the study of the spatial and temporal components of algal dispersal, in particular the information on the environmental factors that affect the occurrence and abundance of this invasive exotic diatom in Patagonia. The Monitoring Program was successful in the selection of environmental variables (both those measured in situ and those determined in the laboratory) analyzed to assess *D. geminata* dynamics. The program also enabled us to observe that this species can grow in a wider range than expected of some environmental variables such

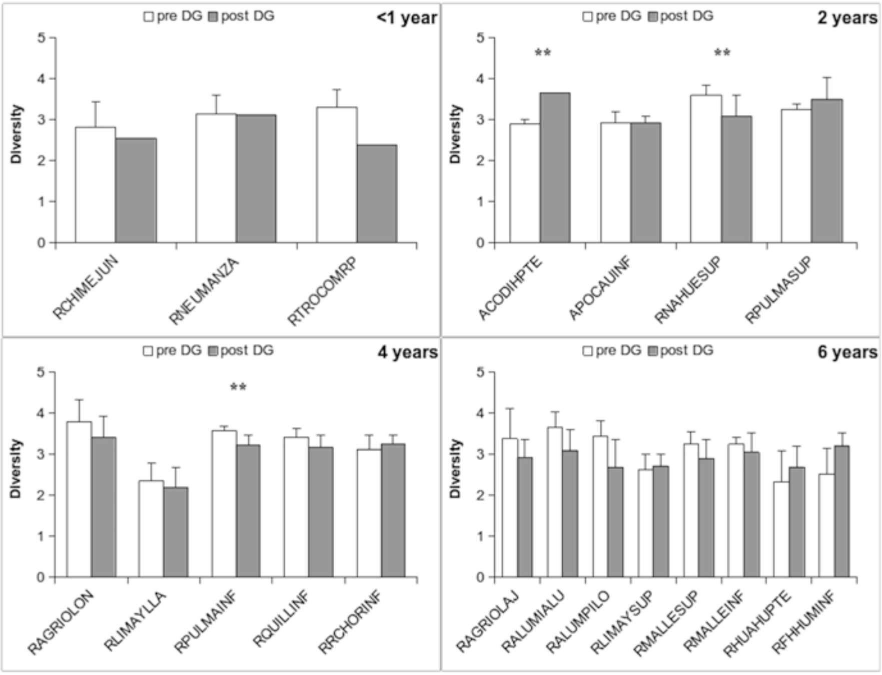

**Fig. 12.7** Average diversity of periphyton (Shannon index) before (pre-DG) and after (post-DG) the appearance of *D. geminata*. The sites were grouped according to the time elapsed since their first detection (until 2019 sampling). Asterisks over the bars indicate statistical differences for t-tests (**$p < 0.05$)

as SRP. These are the reasons driving the continuation of the Monitoring and Surveillance Program for *D. geminata* by the Neuquén Province.

Between 2012 and 2014, the program had a frequency of two samplings per year, autumn and spring, which did not show a marked seasonality due to the variables measured (Beamud et al. 2019). Frequency has been changed and only one sampling per year is currently done. Along with this improvement, measuring light irradiation, daylight hours, and inorganic forms of N (nitrates and ammonium) should also be considered since these variables are important for the ecology of the alga (Bray et al. 2017; Kilroy and Larned 2016). In Chile, it was observed that between 2010 and 2015, the alga continued its colonization at a much slower rate without yet reaching a complete biogeographic equilibrium (Montecino et al. 2016). In the case of Neuquén, the expansion of *D. geminata* was similar to that observed in Chile: abundance increased continuously from its detection until 2019, with maximum values between 2014 and 2016, and subsequently decreased to values similar to those registered in 2012 (Beamud et al. 2019).

Despite the efforts by state agencies and early provincial response with training, regulations, and exhaustive monitoring, the problems regarding coordination between jurisdictions, and the lack of sustained control and prevention actions over time, the spread of *D. geminata* has been impossible to stop. Its expansion to new environments in northeastern Neuquén and other Argentinean provinces could very likely be due to the plasticity of this invasive species. We believe that for invasive species management and control actions to be truly effective, there must be previous protocols and agreements between the different government agencies (scientific, financial, and control). This would allow a faster response to alerts, avoiding administrative delays that only hinder actions, with possible negative results in the face of dispersal of new invasive species.

Although the various contributions of nature to people enrich us as a species, we must take into account that man continually moves from one ecological system to another, enjoying its contributions and services. For this reason, it is essential that the State, at all its organizational levels, understands that nature does not acknowledge legal jurisdictions and that we must learn to prevent invasions as a consequence of our continuous movement and not to facilitate them. It is a great advantage for Neuquén Province to already have regulations to prevent new invasions. Therefore, it is vital to demand their compliance.

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# Chapter 13
# Fish and Fisheries of the Patagonian Steppe

**Claudio C. R. Baigún, Miguel Casalinuovo, Pamela Quiroga, Carla Riva-Rossi, Darío Colautti, Patricio Solimano, Nelson Bovcon, Tomas Maiztegui, and Facundo Llompart**

## 1 Introduction

Patagonia, a vast territory at the southern tip of South America, has a diverse array of water bodies, which have been heavily influenced by Andean range orogeny during the Miocene (approximately 23 My BP) and the glacial cycles of the Quaternary (2.5 My–10,000 years BP) (Clapperton 1993; McCulloch et al. 2000; Corbella and Lara 2008; Rabassa et al. 2011). The influence of these past geological and climatic events in shaping the fish biodiversity of the Patagonian steppe has only recently been recognized (Zemlack et al. 2008, 2011; Muñoz Ramírez et al. 2014). In the Andean area, the last glacial retreat allowed fish species to colonize lakes and rivers

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(Ruzzante et al. 1998, 2006); however, glaciation had a much lower impact on the steppe, where fish species similarity across disjunct drainages has been explained by past mixing in the extensive paleolakes formed during the glacial retreat (Clapperton 1993; Zemlack et al. 2011). Postglacial lakes east of the Andes are relatively young environments where species have not had enough time for speciation or colonization (Cussac et al. 2009). The presence and overriding dominance of exotic salmonids as part of the Patagonian fish assemblages is the result of introduction practices carried out since the beginning of the twentieth century (Baigún and Quirós 1985; Pascual et al. 2002; Macchi and Vigliano 2014).

The Patagonian plateau is a complex landscape mainly characterized by a basaltic plateau and tectonically uplifted pebble fans (Iriondo 1989), crossed by several large transversal basins that have their headwaters in the Andes mountains (Fig. 13.1). Most of the basins originated from watercourses produced by snowmelt. Typically, the Patagonian plateau is crossed by wide meandering rivers, sometimes with secondary channels but lacking main tributaries. Unlike the Andean zone where most lakes were formed by glaciers, large lakes of eolic and tectonic origin coexist in the arid steppe (Quirós and Drago 1999; Baigún and Marinone 1995; Baigún 2001), along with reservoirs created by the construction of large hydroelectric dams (INA 2020) (see also Chap. 9). Plateau lakes generally range from mesotrophic to eutrophic, due to favorable morphoedaphic conditions (Baigún and Marinone 1995; Diaz et al. 2000).

## 2 General Background

Although in the past most research on Patagonian fish focused on populations inhabiting Andean sites, several studies on the plateau have emerged from specific cases describing the life history of migratory species (Riva-Rossi et al. 2007; O'Neal and Stanford 2011), exotic fish invasions (Ciancio et al. 2005; Riva-Rossi et al. 2012; Quiroga et al. 2015; Casalinuovo et al. 2018b), species distribution (Soricetti et al. 2020), trophic characteristics (Bello et al. 1991; Ferriz 1987; Ruiz 2002), growth (Dománico and Freyre 2008; López Cazorla and Sidorkewicj 2008, 2011),

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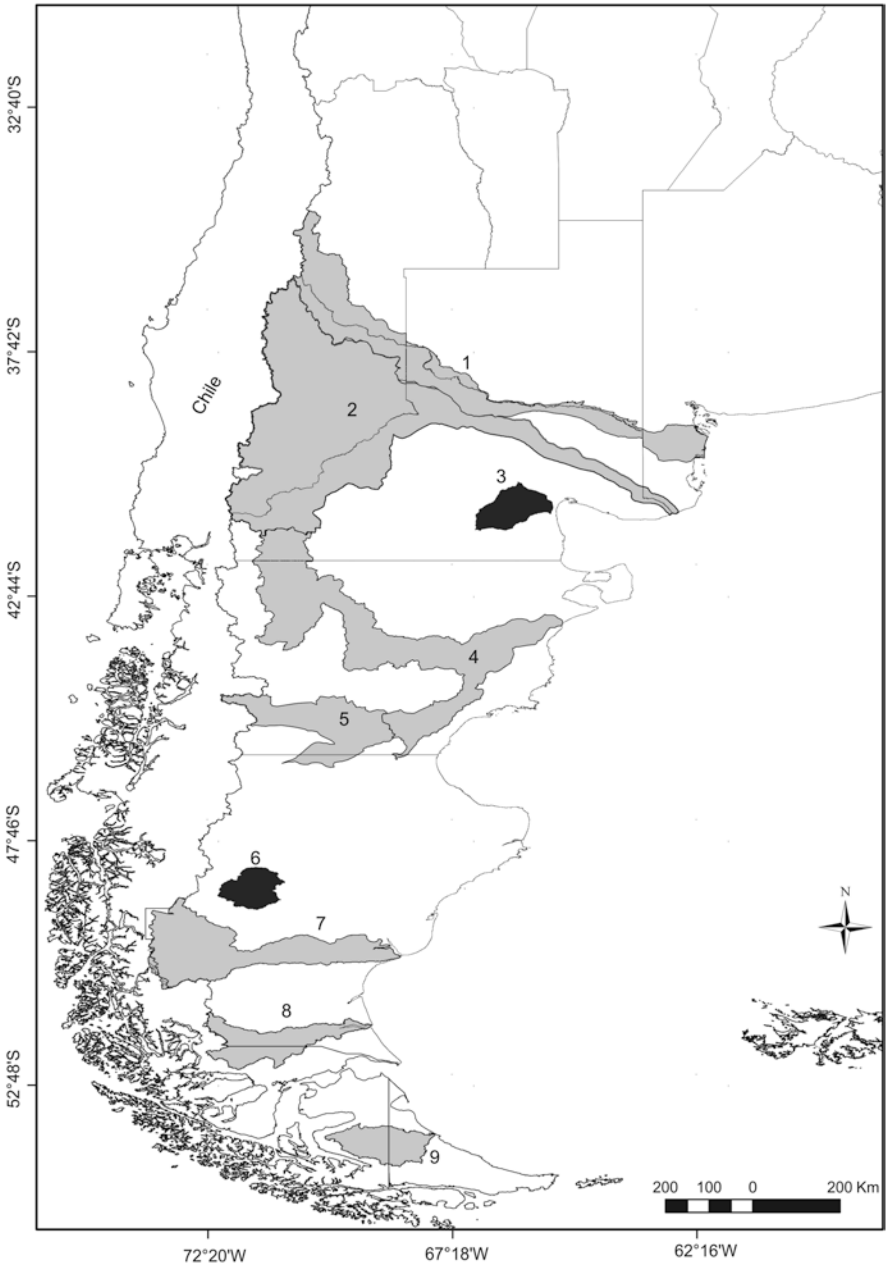

**Fig. 13.1** Location of the Patagonian basins assessed. (1) Colorado River, (2) Negro River, (3) Valcheta River, (4) Chubut River, (5) Senguerr River, (6) Strobel-Cardiel lakes, (7) Santa Cruz River, (8) Gallegos River, and (9) Grande River. Black areas represent endorheic basins considered in this study

reproduction (Ruiz 2002; López Cazorla and Sidorkewicj 2011), extreme endemism (Ortubay et al. 1997; Quiroga et al. 2015), and cases of isolation (Colautti et al. 2020). Large-scale studies have also obtained valuable information on distribution patterns (e.g., Pascual et al. (2002, 2007), Baigún and Ferriz (2003), Alvear et al. (2007), Aigo et al. (2008), and Carrea et al. (2012)).

Information about fisheries associated with water bodies and rivers of the Patagonian steppe is still scarce. The introduction of salmonids into some rivers and lakes in this area has promoted valuable recreational fisheries (e.g., Pascual et al. (2009), Garcia de Leaniz et al. (2010), Casalinuovo et al. (2018b), and Llompart et al. (2019)), which are based on the capture of trophy-sized anadromous fish (Casalinuovo et al. 2018b). Anadromous species such as *Salmo trutta* (brown trout) have led to the development of world-class fisheries on the Gallegos and Grande rivers, attracting fishermen from all over the world (Vigliano and Alonso 2007), whereas recreational fisheries of the anadromous ecotype of *Oncorhynchus mykiss* (rainbow trout) and *Oncorhynchus tshawytscha* (Chinook salmon) on the Santa Cruz River have gained regional importance. A distinctive feature of some plateau lakes is the existence of small-scale commercial fisheries, such as those extracting *O. mykiss* from the Cardiel (Pascual et al. 2001a), Musters, and Colhue Huapi lakes (Baigún and De Nichilo 1985; Baigún et al. 1988), and several shallow lakes on the Strobel Lake plateau (Pascual et al. 2009). Special mention should be made of the Strobel Lake world-class recreational fisheries, where *O. mykiss* have high growth rates and reach sizes similar to those of anadromous specimens. Some of these environments are exploited commercially and restocked periodically. The target species in Laguna Blanca Lake is *Percichthys trucha* (Creole perch) (Ortubay et al. 2006), whereas in Pellegrini Lake, the target species are *Odontesthes hatcheri* (Patagonian pejerrey) and *Odontesthes bonariensis* (pejerrey) (Amalfi and Verniere 2009). In the past (2009), the yield of the commercial fisheries in this lake was up to 300 tons annually for both species. The fish yield of small lakes located in the ecotonal area between the plateau and the Andean region has been estimated to be as high as 20–40 kg $ha^{-1}$ $year^{-1}$, while in large lakes and reservoirs, production could reach 20–30 kg $ha^{-1}$ $year^{-1}$ (Baigún 2001; Baigún et al. 2007).

Recreational fisheries in Patagonia are subject to regulations set by a licensing board integrated by the provinces (except Tierra del Fuego, Antártida e Islas del Atlántico Sur) and National Park Administration. However, each province has specific regulations for certain environments. The license includes catching regulations by species, quota, gear, environment, and season. All salmonids have recreational fishing value and are therefore subject to specific catch restrictions. The fishing of most native species is prohibited, with the exception of *O. hatcheri* and *P. trucha*, for which daily catch quotas are prescribed. The exotic *Cyprinus carpio* (common carp) is the only species in Patagonia with no catch limitations. Information on fishery assessment and management is still scarce and is mostly found as gray literature, but time and effort have been devoted to producing a general overview of some general fishery characteristics of lakes and reservoirs located in the Patagonian steppe (e.g., Baigún and De Nichilo (1985), Baigún (2001), and Casalinuovo et al.

(2018a, b)). Nevertheless, there is still lacking a comprehensive and integrated long-term perspective on the fish and fishery resources of the Patagonian steppe.

This chapter reviews current knowledge of the ichthyofauna that inhabits the main basins and aquatic environments of the Patagonian plateau. We also describe the artisanal and sport/recreational fisheries and give an overview of the main threats and impacts that affect native fish species. We also discuss past and current management policies on non-native species introduction as a basis for fishery development and the conservation of native fish species.

## 3 Main Characteristics of the Ichthyofauna

Although the Patagonian region covers a territory from 37° S to 56° S, from an ichthyogeographic point of view, the upper boundary of Patagonia extends north of Colorado River and includes the southern area of Mendoza, La Pampa, and Buenos Aires provinces (López et al. 2008). The fish richness of the Patagonian ecoregion has varied between 25 and 35 species, depending on whether the assessment criteria considered the estuarine species and species of doubtful presence (e.g., Baigún and Ferriz (2003), Pascual et al. (2007), Aigo et al. (2008), and Becker et al. (2018)). In this chapter, we have differentiated fish species according to the type of biogeographical region, migratory strategy, and origin, identifying a total of 29 valid species in the steppe region, belonging to 11 orders and 16 families (Table 13.1). To classify the species by origin, we followed the criteria of Copp et al. (2005). We considered a native species (subspecies, ecotype, or race) to be a taxon occurring naturally in a geographical area, with dispersal taking place independently of human intervention. Conversely, a non-native or exotic species was considered a species, subspecies, race, or variety that does not occur naturally in a geographical area and that has been, intentionally or not, introduced by man to establish self-sustaining populations. A translocated (transplanted) species is an organism that, although it belongs to a political entity (country) where it is native, is present in basins in which it did not exist naturally.

## 4 Fish Species Composition

Species of the Patagonia steppe include elements of the Patagonian, Andean Cuyean, and Pampean ichthyological provinces (López et al. 2008), which share several species, such as *O. hatcheri*, *Hatcheria macraei* (torrent catfish), *Diplomystes viedmensis* (velvet catfish), and *P. trucha* (Creole perch) (Arratia et al. 1983). The large basins of Patagonia differ from one another, but differences can also be found between the Andean and plateau regions (Cussac et al. 2016). Several authors have

**Table 13.1** Species recorded in Patagonia

| Order | Family | Species | Common name | Biogeographic region and province | | | Distribution in Patagonian basins | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | *O* | MS | CR | RR | Ch R | SR | SCR | GaR | GR | EB |
| Petromyzontiformes | Geotriidae | *Geotria macrostoma* | Patagonian lamprey | Neotropical (Patagonia) | N | A | | M, L | L | | U, M, L | U, M, L | U, M, L | |
| Characiformes | Characidae | *Psalidodon pampa* | Characin | Neotropical (Pampean) | T | F | L | U, M, L | | | | | | |
| | | *Cheirodon interruptus* | Characin | Neotropical (Pampean) | T | F | M, L | U, M, L | | | | | | Yes |
| | | *Gymnocharacinus bergii* | Naked characin | Neotropical (Patagonia) | N | F | | | | | | | | Yes |
| | | *Oligosarcus jenynsii* | | Neotropical (Pampean) | T | F | M, L | | | | | | | |
| Siluriformes | Diplomystidae | *Diplomystes cuyanus* | Velvet catfish | Neotropical (Patagonia) | N | F | U, M, L | | | | | | | Yes |
| | | *Diplomystes mesembrinus* | Velvet catfish | Neotropical (Patagonia) | N | F | | | M, L | M | | | | |
| | | *Diplomystes viedmensis* | Velvet catfish | Neotropical (Patagonia) | N | F | M, L | U, M, L | | | | | | |
| | Callichthyidae | *Corydoras paleatus* | Peppered corydoras | Neotropical (Pampean) | E | F | | U, M, L | | | | | | |
| | Trichomycteridae | *Hatcheria macraei* | Torrent catfish | Neotropical (Patagonia) | N | F | U, M, L | U, M | U, M, L | U, M | | | | |
| | | *Trichomycterus areolatus* | Torrent catfish | Neotropical (Patagonia) | N | F | U, M, L | | U | | | | | |

| | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | *Trichomycterus borelli* | Torrent catfish | Neotropical (Patagonia) | N | F | U, M, L | | | | | | | Yes |
| | Ariidae | *Genidens Barbus* | White sea catfish | Neotropical | N | M | | L | | | | | | |
| Osmeriformes | Galaxiidae | *Galaxias platei* | Big puyen | Neotropical (Patagonia) | N | F | | U | | U, M, L* | U, M | O | O | |
| | | *Galaxias maculatus* | Small puyen | Neotropical (Patagonia) | N | M/F | | U, M, L | | | U, M, L | U, M, L | U, M, L | |
| | | *Aplochiton zebra* | Peladilla | Neotropical (Patagonia) | N | A | | * | | | | | | |
| | | *Aplochiton taeniatus* | Peladilla | Neotropical (Patagonia) | N | F | | * | | | | * | | |
| Atheriniformes | Atherinopsidae | *Odontesthes hatcheri* | Patagonian pejerrey | Neotropical (Brasilic) | N | F | U, M, L (rf) | U, M, L (rf, af) | U, M, L | U, M, L (F) | | | | Yes |
| | | *Odontesthes bonariensis* | Pejerrey | Neotropical (Brasilic) | T | F | U, M, L (rf) | U, M, L (rf,af) | | | | | | |
| | | *Odontesthes argentinensis* | Marine pejerrey | Neotropical (Brasilic) | N | B | L (rf) | | | | | | | |
| Perciformes | Percichthyidae | *Percichthys trucha* | Creole perch | Neotropical | N | F | U, M, L (rf) | U, M, L (rf, af) | U, M, L (rf) | U, M, L (af) | U, M, L (rf) | * | * | Yes |

(continued)

**Table 13.1** (continued)

| Order | Family | Species | Common name | Biogeographic region and province |  |  | Distribution in Patagonian basins |  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
|  |  |  |  |  | *O* | MS | CR | RR | Ch R | SR | SCR | GaR | GR | EB |
|  | Eleginopsidae | *Eleginops maclovinus* | Patagonian blenny | Neotropical (Patagonia) | N | B |  |  | L (rf) |  | L (rf) | L (rf; af) | L (rf) |  |
| Cypriniformes | Cyprinidae | *Cyprinus carpio* | Common carp | Palearctic | E | F | U, M, L (af) | U, M, L (rf) |  |  |  |  |  |  |
| Salmoniformes | Salmonidae | *Salvelinus fontinalis* | Brook trout | Nearctic | E | F | U | U | U (rf) | U, M (rf) | O | U, M (rf) | * | Yes |
|  |  | *Salvelinus namaycush* | Lake trout | Nearctic | E | F |  |  |  |  | U, M, L |  |  |  |
|  |  | *Salmo salar* | Atlantic salmon | Nearctic | E | F |  | U |  |  |  |  |  | Yes |
|  |  | *Salmo trutta* | Brown trout | Palearctic | E | F | U | U, M (rf) | U, M, L (rf) |  | U, M, L (rf) | U, M, L (F) | U, M, L (F) |  |
|  |  |  | Sea trout | Palearctic | E | A |  |  |  |  | O | U, M, L (F) | U, M, L (F) |  |
|  |  | *Oncorhynchus mykiss* | Rainbow trout | Nearctic | E | F | U, M, L | U, M, L (rf) | U, M, L (rf) | U, M, L (rf, a) | U, M, L (rf) | U, M, L | U, M, L | Yes |

| | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | Steelhead | Nearctic | E | A | | | | | U, M, L (F) | R | R | |
| | | *Oncorhynchus tshawytscha* | Chinook salmon | Nearctic | E | A | | R | R | | U, M, L (F) | R | U, M, L (F) | |
| | | *Oncorhynchus kisutch* | Coho salmon | Nearctic | E | A | | | | | R | | | |
| Cyprinodontiformes | Poeciliidae | *Cnesterodon decemmaculatus* | Ten spotted livebearer | Neotropical (Pampean) | T | F | | U, M, L | | | | | | Yes |
| | Anablepidae | *Jenynsia lineata* | One sided livebearer | Neotropical (Brasilic) | N | F | M, L | U, M, L | | | | | | Yes |
| Mugiliformes | Mugilidae | *Mugil liza* | Lebranche mullet | Neotropical | N | B / M | L | L (rf) | | | | | | |
| Pleuronectiformes | Paralichthyidae | *Paralichthys orbignyanus* | Flounder | Neotropical | N | B | L | L (rf) | | | | | | |

*O* origin, *N* native, *E* exotic, *T* translocated species, *MS* migratory strategy, *F* freshwater, *A* anadromous, *B* brackish, and *M* amphidromous. Distribution in Patagonian basins: *CR* Colorado River basin, *NR* Río Negro River basin, *ChR* Chubut River basin, *SR* Senguerr River basin, *SCR* Santa Cruz River basin, *GaR* Gallegos River basin, *GR* Grande River basin, and *EB* endorheic basis. *U* upper basin, *M* middle basin, *L* lower basin, *O* species/ecotype with occasional records, *R* species reported intermittently that do not have self-sustaining populations and are possibly the result of straying individuals, * species recorded in the literature but of doubtful presence today, (f) species of international recreational fishing importance in one or more sectors of the basin, (rf) species of local recreational/sprot fishing importance in one or more sectors of the basin, and (af) species of local artisanal fishing importance in one or more sectors of the basin. Ichthyological province classification followed López et al. (2008) criteria, whereas species origin was classified according to Copp et al. (2005)

identified a zoogeographic transition zone recognized as an ecotone, formed by the Negro and Colorado rivers, where several species of the biogeographic Brazilic subregion (Ringuelet 1975) have been detected (e.g., Cazzaniga (1978), Ferriz and López (1987), Almirón et al. (1997), Ortubay et al. (1997), Baigún et al. (2002), and Soricetti et al. (2020)). According to López et al. (2008), the Colorado and Negro basins have more affinity with fish assemblages from rivers located in the Andean-Cuyean ichthyological province. The presence of this ecotonal area enabled to divide Patagonia in two: a northern part where species of the Brazilic subregion coexist with species of the Austral subregion and a southern region inhabited only by strictly Patagonian species that belong to the Austral subregion (Ringuelet 1975). This distribution pattern also allows dividing the Patagonian steppe into two main groups, mainly due to the presence of some species of Characiformes, Mugiliformes, and Pleuronectiformes order exclusively in the Colorado and Negro basins (Fig. 13.2). Moreover, more than 50% of the total richness (15 species) is present only in these two basins. These basins share some Siluriformes and Atheriniformes with the Senguerr and Chubut rivers, but these fish orders are totally absent in the second group which is made up of only the southern basins.

## *4.1 Main Family Characteristics*

### Geotriidae

The Patagonian lamprey, *Geotria macrostoma*, is the only native anadromous species that inhabits all the large Atlantic basins of Patagonia, including coastal streams flowing into the Beagle Channel (from 35°S to 55°S), except the Colorado River. The taxonomic status of this species has recently been amended by several authors (Nardi et al. (2020), Riva-Rossi et al. (2020a, 2022), and Baker et al. (2021)), who provide molecular, morphological, and morphometric evidence to differentiate *G. macrostoma*, endemic to Patagonia, from the more widespread *Geotria australis*, distributed throughout Australasia and Chile.

### Characidae

The taxonomic status of some Characids from Patagonian basins is still under debate, and several records of *Astyanax eigenmaniorum* correspond to *Psalidodon pampa* (characin). Other characins identified in the Patagonian steppe are *Cheirodon interruptus* and *Oligosarcus jenynsii*, both transplanted, and the native *Gymnocharacinus bergi* (naked characin). This native species has been assessed as critically endangered (Cussac et al. 2019), representing a unique case within the Patagonian ichthyofauna.

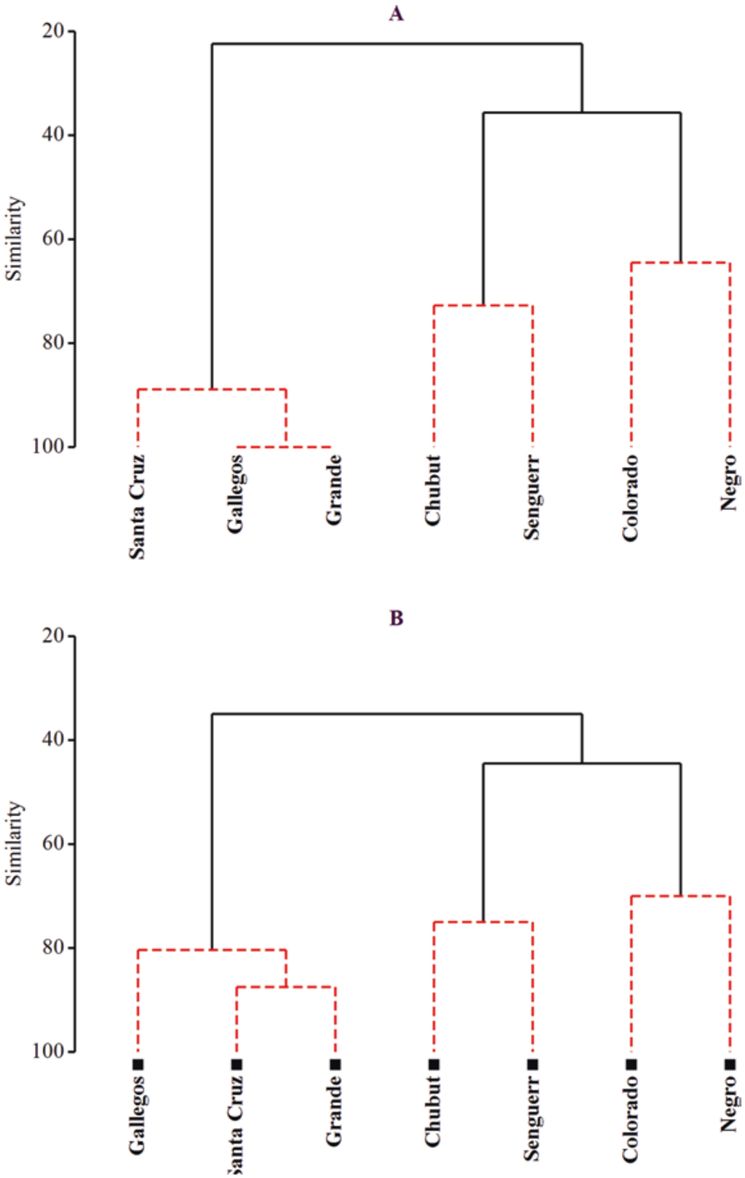

**Fig. 13.2** Cluster analysis of Patagonian steppe rivers according to the presence/absence of their native fish (**a**) and to the total species richness (native and exotic) (**b**). Sorensen index was calculated as a similarity measure for the species list presented in Table 13.1 but disregarded doubtful species and those reported only as stray individuals (species symbolized as * and R in Table 13.1). The red dashed line indicates groupings that are not significantly differentiated by the SIMPROF (similarity profile) test

## Diplomystidae

The Diplomystidae family is represented in Argentina by the velvet catfish *Diplomystes cuyanus*, *D. mesembrinus*, and *D. viedmensis*, each of which is endemic to a different large river basin (Azpelicueta 1994). These species were included within a new genus, *Olivaichthys* (Arratia 1987), but following Sullivan et al. (2006) and Muñoz Ramírez et al. (2014), we have accepted the presence of *D. viedemensis* and *D. mesembrinus* as the only valid species. These species develop small local populations (Azpelicueta 1994), and it has been suggested that maximum conservation priority should be allocated to *D. mesembrinus* (Bello and Úbeda 1998; López et al. 2002).

## Callychthydae

This family is represented in Patagonia only by *Corydoras paleatus* but has a relatively wide geographic distribution that covers almost all southern South America (Tencatt et al. 2016). This species is widely used in aquariums, mainly because of its feeding behavior. This is one of the main reasons for its introduction and invasion of several ecosystems around the world, including north Patagonian water bodies (Baigún et al. 2002; Gertzen et al. 2008; Maceda-Veiga et al. 2013).

## Trichomycteridae

The Trichomycteridae family is widely distributed throughout Argentina. In particular, the torrent catfish *Trichomycterus areolatus* and *H. macraei* have been recorded in Patagonia (Baigún and Ferriz 2003). *Trichomycterus areolatus* inhabits several rivers from north to central Patagonia, whereas *H. macraei* is found up to the north of Santa Cruz Province. These species inhabit fast, clear, cold, and well-oxygenated currents, exhibit negative phototaxis (Menni 2004), and show ontogenetic changes in their diet (Barriga and Battini 2009).

## Ariidae

The catfish of the Ariidae family are common in estuarine environments in the southwestern Atlantic Ocean (Gomes et al. 2001). Ariids can exhibit seasonal movements between environments with different salinities; the movements are related to biological aspects such as reproduction, feeding, and ontogeny (Avigliano et al. 2017). The marine catfish, *Genidens barbus*, is an anadromous species considered important historically for commercial and recreational fisheries (Reis 1986; Velasco et al. 2007). It is distributed along the southwestern Atlantic Ocean from Bahía in

Brazil to the Negro River in Argentina (Avigliano et al. 2021), and it is caught mainly in estuarine or coastal lagoon systems. Adults migrate to estuaries to breed (Reis 1986; Avigliano et al. 2017). Their reproductive strategy involves the production of a few large eggs and male gastric incubation (Reis 1986). This species has recently been included in the Red List of endangered species in Brazil (Di Dario et al. 2015); for Argentina, Baigún et al. (2012) classified the species as vulnerable.

### Galaxiidae

These species exhibit a wide variety of life history patterns and have played a key role in the postglacial colonization of Andean lakes and streams, representing a key component of the aquatic food webs (Cussac et al. 2020). Their eurythermal nature enables them to occupy a wide latitudinal range (Carrea et al. 2012; Barrantes et al. 2017). *Galaxias platei* (big puyen) is endemic to the Patagonian ichthyogeographic province (Dyer 2000; Habit et al. 2010). It inhabits the deep benthic zones of lakes and rivers throughout the region and is the only fish species that occurs naturally in several high-altitude Patagonian lakes (Ruzzante et al. 2006; Zemlack et al. 2008). Adults are even found below the thermocline and euphotic zone (Cussac et al. 2020), while the larval and juvenile stages are coastal epipelagic, associated with vegetation (Ferriz 2003). In turn, *Galaxias maculatus* (small puyen) is characterized by resident and diadromous populations (Carrea et al. 2012), this species having facultative diadromy. Although most *G. maculatus* populations in the Patagonian steppe exhibit resident life history (Cussac et al. 2004; Zattara and Premoli 2004), in southern Patagonia, diadromous populations have been reported (Boy et al. 2007, 2009). In this case, reproduction takes place in estuarine environments, and juveniles (crystalline) breed in estuaries or in the brackish waters of the lower course of the rivers (Cussac et al. 1992; Barriga et al. 2002; Boy et al. 2007). Other species such as *Aplochiton zebra* and *Aplochiton taeniatus*, with a narrower distribution range (Baigún and Ferriz 2003), have not been reported in the Patagonian steppe for a long time.

### Atherinopsidae

*Odontesthes hatcheri* inhabits lakes and rivers of the steppe (Bello et al. 1994; Liotta 2006), showing high plasticity and growth variability in different environments (Grosman 1993; Bello et al. 1994; Ruiz 2002; Conte-Grand 2012). This species has also shown spatial overlap and hybridization with *O. bonariensis* (Rueda et al. 2017), which was introduced by stocking (Cussac et al. 2009). A rare case of a landlocked marine pejerrey is found in the La Salada de Pedro Luro Lake, at the northern limit of Patagonia; this populations represents an alternative ecophenothype of the marine *Odontesthes argentinensis* (Colautti et al. 2020) and has developed a local self-sustaining population displaying adaptive/phenotypic traits to cope with the lake's environmental and trophic conditions.

## Percichthydae

*Percichthys trucha* is endemic to southern South America and in Argentina is distributed from 38°S to 55°S (Arratia et al. 1983). The genus *Percichthys* inhabits several rivers and lakes in Patagonia, including the steppe (Amalfi and Verniere 2009). *Percichthys* is of moderate recreational value but in some environments of the steppe represents a target species for commercial fishing. The taxonomic status of this species remains controversial (López-Albarello 2004; Ruzzante et al. 2006; Arratia and Quezada-Romegialli 2019); however, at present, there is consensus that *P. trucha* is the only valid species inhabiting Patagonia, while the other recognized species represent cases of intra-specific trophic polymorphism and display significant phenotypic variation (Ruzzante et al. 1998, 2003; Crichigno et al. 2014).

## Eleginopsidae

Eleginopsidae is a monospecific family comprising only *Eleginops maclovinus* (Figueroa 2019). This species is endemic to the temperate sub-Antarctic waters of southern South America. In the Pacific Ocean, it is distributed from Valparaiso, Chile (33°S) (Pequeño 1989), to the southern tip of the Fuegian archipelago (56°S), including the Malvinas Islands archipelago, and in the Atlantic Ocean, it is found from Tierra del Fuego to Golfo San Matías, Río Negro, Argentina (40°S) (Cousseau and Perrotta 2013). It is a demersal-benthic, strictly coastal species that spends most of its life in waters shallower than 50 m, although it has occasionally been caught at depths of up to 250 m. It is a protandric hermaphrodite species (Calvo et al. 1992), and the presence of adults and juveniles is also very common in estuaries and at river mouths, which is why it is considered a euryhaline fish (Ceballos 2011).

## Cyprinidae

The common carp is considered one of the most ecologically harmful freshwater fish (Koehn 2004; Britton et al. 2010; Kloskowski 2011), and worldwide it accounts for most of the records of successful establishments (Kulhanek et al. 2011). However, most Patagonian rivers are still free of *C. carpio*, with the exception of the northern area, where the species was introduced in the mid-1980s and is currently a common component of the fish assemblages of the Colorado and Negro rivers (Maiztegui et al. 2016). Large dams located in the Patagonian steppe lacking fishways represent major barriers to the dispersal of this species. The absence of *C.*

*carpio* in the middle and southern catchments could be linked to the isolation of basins and probably to the lack of stocking events in the area.

### Salmonidae

Salmonids have represented a distinctive feature of Patagonia since their introduction at the beginning of the twentieth century from North America and Europe (Baigún and Quirós 1985; Pascual et al. 2007; Macchi and Vigliano 2014). From a total of ten species stocked between 1904 and 1930, only *O. mykiss*, *Salvelinus fontinalis* (brook trout), *S. trutta*, *Salmo salar* (Atlantic salmon), and *Salvelinus namaycush* (lake trout) were able to establish self-sustainable populations. All established salmonids in Patagonia display resident life histories, with the exception of *O. tshawytscha*, and the anadromous ecotypes of *O. mykiss* (steelhead) and *S. trutta* (sea-run brown trout) (Pascual and Ciancio 2007; Casalinuovo et al. 2018b). More than a century after the first introduction, *O. mykiss* can be considered the most successful and widely distributed salmonid (Pascual et al. 2002) and along with *S. trutta*, it is widespread throughout Patagonia. Introductions of *O. mykiss* into the Patagonian plateau began around 1904 and accelerated over the following decades due to stocking practices associated with recreational fisheries (Lancelotti et al. 2010).

Macchi and Vigliano (2014) mentioned that *S. trutta* and *O. mykiss* occur in 85% and 97% of the basins, respectively, where they were stocked, whereas *S. fontinalis* is present in only 42%. Of all stocked salmonids, *S. salar* as a landlocked ecotype is found only in a few lakes and rivers of the Andes range, being absent in the steppe. Other stocking attempts based on *Oncorhynchus nerka* (sockeye salmon), *Oncorhynchus masou* (masou salmon), *O. tshawytscha*, *Oncorhynchus kisutch* (coho salmon), and *Coregonus clupeaformis* (lake whitefish) failed. Over the last decades, however, *O. tshawytscha* has colonized several Patagonian basins after escaping from Chilean fish farms and net pens (Ciancio et al. 2005; Di Prinzio and Pascual 2008; Riva-Rossi et al. 2012; Ciancio et al. 2015; Di Prinzio et al. 2015; Gomez-Uchida et al. 2018); stray individuals of *O. kisutch* have also been reported in south Patagonia (Casalinuovo et al. 2016; Chalde and Fernandez 2017; Chalde and Llompart 2021).

### Poeciliidae

This family is represented in the Patagonian steppe by the species *Cnesterodon decemmaculatus*, a small viviparous fish. The species was first recorded in 1994 by Ortubay in the Curicó Lake (40°36′54″S, 65°55′48″W) (Ortubay et al. 1994) and then in the Negro River in 2020 (Soricetti et al. 2020).

## Anablepidae

The Anablepidae family is represented in northern Patagonia by *Jenynsia lineata*, a small viviparous fish that inhabits a wide range of habitats. This species can be found from lowlands up to 2300 m above sea level and from montane rivers in the Andes to big rivers in La Plata Basin, in Argentina, Brazil, and Uruguay (Aguilera and Mirande 2005; Frota et al. 2019; Briñoccoli et al. 2021). The species was first reported in Patagonia in the Negro River (MacDonagh 1936) and is considered native to the Colorado and Negro rivers.

## Mugilidae

The Mugilidae family consists of medium-sized fishes distributed in tropical, subtropical, and temperate regions of all continents (Whitfield et al. 2012). These species are mostly euryhaline and can inhabit marine coasts, brackish lagoons, and estuaries and even enter freshwater streams (González-Castro and Minos 2015). They use estuarine habitats for feeding, growth, and gonadal maturation (González-Castro et al. 2009). In Argentina, the only species present in this family is *Mugil liza* (Lebranche mullet) (González-Castro et al. 2008). This species is distributed from Florida (USA) through the Caribbean Sea to the north of Argentine Patagonia and is the most southerly distributed mugilid in the southwestern Atlantic Ocean (Cousseau et al. 2005).

## Paralichthyidae

The Paralichthyidae are a non-monophyletic and cosmopolitan family of sinistral (usually) flatfishes (Munroe 2015). Most *Paralichthys* are distributed in temperate and tropical seas along the coasts and on the continental shelves of the American continent (Munroe 2003). In the southwestern Atlantic Ocean, *Paralichthys* species are distributed from northern Brazil, just south of the Amazon River (Walsh et al. 2015), to central Patagonia, Argentina, at about 47°S (Díaz de Astarloa and Munroe 1998). Throughout central Patagonia, this species is found in diverse habitats, including shallow waters in areas containing muddy and silty substrates, such as turbid estuaries, and also in sandy substrates at moderate depths on the continental shelf; a very small number of species are found in deepwater substrates located on the outer continental shelf. *P. orbignyanus* is a shallow-water fish found from Rio de Janeiro to the San Matías Gulf in Argentina. Within its distribution area, it inhabits estuarine zones and depths from the coastline to about 30 m (Díaz de Astarloa et al. 2018).

# 5 Species Distribution Patterns

Species distribution by basin and origin is displayed in Fig. 13.3. Species richness decreases toward higher latitudes, whereas the proportion of salmonids increases from northern to southern Patagonia (Aigo et al. 2008). Although the number of salmonid species is quite similar among basins, the proportion of anadromous species or species with anadromous ecotypes increases with latitude.

## *5.1 Species of Restricted Distribution*

Endemism on a local scale is not common in the Patagonian steppe, except for *G. bergii* (naked characin) from the Valcheta River basin on the Somuncura plateau (Figs. 13.4 and 13.5). This species has been identified as the southernmost characid in the world (Menni and Gómez 1995; Quiroga et al. 2017). The basin has a cold branch where the water temperature is between 20.5 and 22.5 °C and a hot branch where it reaches 22–26 °C (Ortubay et al. 1997; Quiroga et al. 2017). Individuals are most abundant in the headwaters, decreasing in abundance downstream (Quiroga et al. 2017). The basin is also inhabited by three neotropical transplanted species, *Cnesterodon decemmaculatus* (Poeciliidae), *J. lineata*, and *Ch. interruptus* (Pérez et al. 2015). Pérez et al. (2015) suggested that *Ch. interruptus* could compete strongly with *G. bergii* for food and space, but little information has been gathered to evaluate the impact of non-salmonid species. The vulnerability of this unique species can also be related to watershed degradation due to land use modifying Valcheta basin conditions (Velasco et al. 2016; Arellano et al. 2017).

Two salmonid species, *O. mykiss* (Macchi et al. 2008; Macchi and Vigliano 2014) and *S. fontinalis*, were introduced into Valcheta River basin (Ortubay et al. 1997; Escalante and Menni 1999; Ortubay et al. 2002). The distribution of *S. fontinalis* is widespread throughout the basin. In the cold branch, *O. mykiss* has been detected all year round, whereas in the hot branch, the species has been found only occasionally during winter (Ortubay et al. 1997). Cussac et al. (2019) reported a decline in the distribution of *G. bergii* to less than half its known historic distribution; this decline coincided with the increasing expansion of *O. mykiss* toward the headwaters (Macchi and Vigliano 2014). The construction of waterfalls and control of the species by extractive fishing have been proposed to prevent exotic species from moving upstream. Direct predation of *O. mykiss* on *G. bergii* has been documented (Ortubay and Cussac 2000; Quiroga et al. 2017), partly because in winter months, the trout can ascend the warm branch.

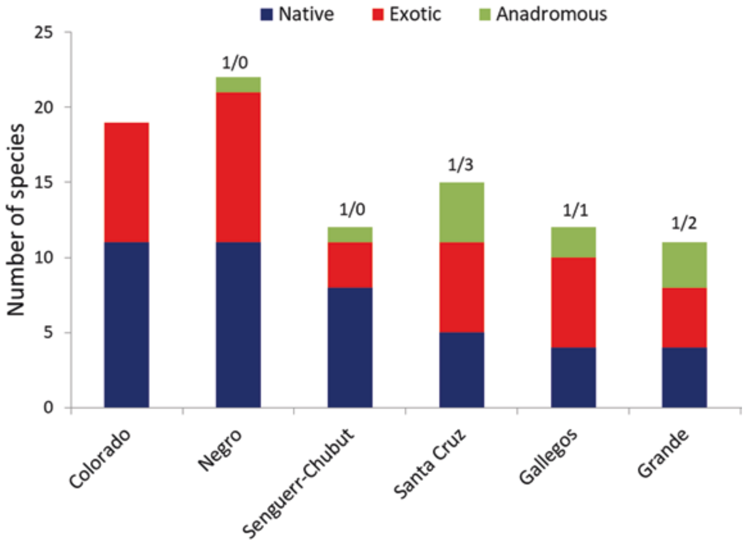

**Fig. 13.3** Number of native, exotic, and anadromous species (exotic and native) by basin (excluding occasional or stray individuals and those recorded in the literature but currently of doubtful presence; see Table 13.1). First number above the bar indicates number of native anadromous species; second number above the bar indicates number of exotic anadromous species

## 6 Fish Assemblages and Fisheries of the Main Basins

### *6.1 Colorado River*

#### Ichthyofauna Characteristics

Of the 19 fish species found in the Colorado basin, 15 are native (Table 13.1). This basin is part of a zoogeographic transition zone, recognized as an ecotone, between the Brazilian and Austral subregions (Arratia et al. 1983; Almirón et al. 1997). Indeed, native fish richness is higher than in the other Patagonian rivers because of the climatic conditions and occasional connections of this system with northern watercourses, resulting in the presence of Characiformes and Cyprinodontiformes. Four exotic species have developed self-sustaining populations in the basin. The salmonids *S. truta* and *S. fontinalis* are restricted to the upper sector, whereas *O. mykiss* and *C. carpio* have spread throughout the entire basin. Also, two estuarine species, *M. liza* and *P. orbignyanus*, migrate in considerable numbers into the lower sector of the Colorado River, increasing diversity even more (Almirón et al. 1997; Fernandez et al. 1998). As in other regions of the country, *C. carpio* has undergone an exponential expansion process over recent decades representing at present a common species in the middle and lower course (Maiztegui et al. 2016).

**Fig. 13.4** Adult individuals of *Gymnocharacinus bergii* (naked characin). (Photo by H. Povedano)

**Fig. 13.5** Headwaters of the Valcheta stream. (Photo by P. Solimano)

### Fisheries

The fisheries are only recreational and more common in the middle basin. They have certain local importance for people living in the towns near the watercourse. Some users pay for access to riverbanks, where they either fish from the shore or launch boats to fish from. In the middle basin, target species include two species of *Diplomystes* and *H. macraei*, which reach up to 30–35 cm in length, and small specimens of *O. hatcheri*, *P. trucha*, and *O. mykiss* of no more than 1 kg. However, the most valued target species of this sector is *C. carpio*, caught with a diverse array of fishing modalities. In the "Casa de Piedra" Reservoir, where fishing is carried out from the shore or in boats, the main target species is *O. bonariensis* and to a lesser extent *C. carpio*, *P. trucha*, and *O. mykiss*. On the lower portion of the river, due to changes in water quality, the fishery experiences diversification of target species and fishing modalities. The best places for fishing *C. carpio* (3–8 kg), *O. mykiss* (up to 1 kg), *O. bonariensis*, and *O. hatcheri* have been identified downstream of Casa de Piedra Dam. In this area, fly casting is the preferred technique of experienced anglers, but most people use line and hook devices. Historical records indicate that a well-established *Odontesthes* sp. fishery also existed at La Salada Lake from the end of the nineteenth century to 1936 (MacDonagh and Thormahlen 1945); this lake currently has an attractive fishery only for local anglers.

## *6.2 Negro River*

### Ichthyofauna Characteristics

The total richness of fish species in the entire basin is 21 (Table 13.1). This is higher than the other Patagonian rivers due to the historical connections with northern basins that resulted in the presence of Characiformes and Cyprinodontiformes and the introduction of several species (Cione and Barla 1997; Casciotta et al. 1999; Bruno et al. 2013). This river exhibits anadromous species such as *G. macrostoma* and *G. barbus* and amphidromous species such as *M. liza* and *P. orbignyanus*. Most of the ichthyological information on the Negro River basin is related to its upper sector (Baigún and Ferriz 2003; Pascual et al. 2007; Aigo et al. 2008), and several fish studies in ecology, distribution, and fisheries have been performed in the Limay River subbasin (e.g., Barriga et al. (2002), Rechencq et al. (2017), and Fernández et al. (2018)). Ichthyological information on the Neuquén and Negro subbasins is scarce and is mainly focused on certain fish species (Ruzzante et al. 2011; Unmack et al. 2012; Muñoz Ramírez et al. 2014) or associated with pollutants of fish fauna (Ondarza et al. 2012, 2014, Miglioranza et al. 2013). Only a few studies performed along the Negro River described its fish assemblage (Alvear et al. 2007; Solimano et al. 2019; Soricetti et al. 2020). Alvear et al. (2007) documented that the most abundant species is *O. hatcheri*, followed by *P. trucha*. Solimano et al. (2019) obtained similar results, highlighting also the abundance of *J. lineata*. These studies have reported a greater abundance of native species on the upper course of the Negro River subbasin, which decreases slightly toward the lower course of the river; this tendency is the inverse of that observed in the groups of exotic species. The fish fauna of the Limay and Neuquen Rivers, and of the upper portion of the Negro River, corresponds markedly to the Austral subregion, characterized by the dominance of *P. trucha*, *G. maculatus*, *O. mykiss*, and *O. hatcheri* (Solimano et al. 2019). In this portion of the Negro River basin, the presence of *D. viedmensis* is also usual; however, this species is restricted to shallow areas of the river. It is important to highlight the presence of *C. paleatus* and *C. carpio* in the Limay and Neuquén river subbasins up to Arroyito Dam (Baigún et al. 2002; Alvear et al. 2007; Solimano et al. 2019), while *S. trutta* and species of the Cyprinodontiformes and Characiformes are rare in this sector. In some parts of the basin, the abundance of *O. mykiss* and *S. trutta* can be accounted for by the activity of the hatcheries that rear and release juveniles of these species for recreational fishing. Some of these aquaculture facilities also carry out captive breeding of native species, such as *O. hatcheri* and *P. trucha*. In the middle portion of the Negro River, Alvear et al. (2007) indicated the dominance of *P. trucha*, *C. carpio*, and *O hatcheri* and a decrease in abundance of *O. mykiss*. Also, Solimano et al. (2019) remarked on the presence of *P. trucha*, *O. bonariensis*, and *O. hatcheri*. In this sector, *G. macrostoma* was a common species, but its presence seems to be diminishing in association with the increment of *C. carpio* abundance in this zone of the river. In the lower sector of the Negro River, *O. bonariensis*, *O. hatcheri*, and *C. carpio* are common and *P. trucha* is also found but to a lesser extent (Soricetti et al. 2020). In this sector, the river is under a marked marine influence, shown by the upstream migrations of *M. liza*, *P. orbignyanus*, and *G. barbus* (Soricetti et al. 2020). *M. liza* is common in the irrigation channels of Viedma City 50 km upstream, whereas *P. orbignyanus* can be found as far as 70 km upstream. *G. barbus* is abundant from September till March, entering from the Atlantic Ocean and reaching as far as 100 km upstream (Solimano et al. 2019, Soricetti et al. 2020).

## Fisheries

The only artisanal fishery in the basin is located in Pellegrini Lake. It is a small gill net fishery that catches exclusively *Odontesthes* spp. and *P. trucha*. Commercialization is carried out locally, representing an income for the villages near the lake. Recreational fishery throughout the entire basin is based on three groups of fishers that differ in their target species. In the Limay and Neuquén rivers as well as the upper portion of the Negro River, there are some recreational fishing spots of international value that focus on the fly-fishing of salmonids. The upper sector of the Limay River is considered the main spot for recreational fishing in this area. Fishers focus mainly on large specimens of *S. trutta*; however, large adults of *O. mykiss* are common and *S. fontinalis* are also captured (Casalinuovo et al. 2017). In recent years, several authors have observed a decline in the quality and size of the salmonid species caught in the basin (Pascual et al. 2007; Vigliano and Alonso 2007). In the middle and lower courses of the Negro River, the main target species are *C. carpio*, *O. hatcheri*, and *O. bonariensis*, which are caught with baited bottom or floating lines. *O. bonariensis* was stocked for recreational purposes (Rueda et al. 2017), and in Pellegrini Lake and some parts of the Negro River, these species sustain a regional-scale fishery. In this portion of the basin, the importance of the *C. carpio* fishery has grown during the last decade, mainly due to an increase in the abundance of this species (Alvear et al. 2007; Soricetti et al. 2020). Fishing tournaments targeting *C. carpio* have become progressively popular in the lower section of the Negro River. Finally, at the limit with the Negro River estuary, the fishery is focused on marine/brackish species, such as *P. orbignyanus* and *G. barbus*.

Recreational fishing in the Negro River basin is regulated by mandatory permits and compliance with the Patagonian continental sport fishing regulations. A group called "Fishing Committee," made up of private stakeholders and governmental fishing authorities, is in charge of enforcing the fishing regulations. However, no fish yield or catch and effort data are available for the basin, precluding managers from estimating long-term fish yields or detecting fishery trends, despite the importance of this recreational fishery (Cussac et al. 2016).

## *6.3 Chubut and Senguerr Rivers*

### Ichthyofauna Characteristics

In the Senguerr and Chubut River basins, 12 fish species are found, 8 native and 4 introduced (Di Persia et al. 1991; Baigún and Ferriz 2003; Ruiz and Fondacaro 2020) (Table 13.1). Present in the Senguerr River basin are the native *O. hatcheri*, *P. trucha*, *H. macraei*, *D. mesembrinus*, and *G. platei*, along with the exotic *O. mykiss* and *S. fontinalis* (Di Persia et al. 1991). All these species are also present in the Chubut River, together with *G. macrostoma*, *T. aerolatus*, and *S. trutta* (Baigún and Ferriz 2003; Ruiz and Fondacaro 2020). According to Arratia et al. (1983), *A. taeniatus* has been recorded in the Chico River, but this could be considered doubtful since several Patagonian rivers have the same name. There are also sporadic records of *O. tshawytscha* in the lower Chubut River basin. Fish assemblages differ according to the environment and their location in the basin. In the Musters and Colhué Huapí lakes and the Florentino Ameghino Reservoir, *P. trucha* and *O. hatcheri* are dominant, while in the lakes of the upper basin (located outside the steppe), salmonids dominate the assemblages. These differences are associated with changes in the trophic gradient of the water bodies in the basin (Baigún 2001).

In the Chubut-Senguerr basins, *O. hatcheri* exhibits great morphometric variability as a consequence of its plasticity to environmental conditions (Baigún 2001; Ruiz 2007). According to Baigún (2001), these differences are correlated with variations in environmental variables such as transparency, total phosphorus, and mean depth. This species can reach a maximum age of 7 years in the Chubut River (Ruiz 2007), but its longevity is variable and differs between the lakes and dams located across the basins (Grosman 1993; Bello et al. 1994; Conte-Grand 2012). The reproductive season of *O. hatcheri* in Florentino Ameghino Reservoir occurs between late winter and spring, with several partial spawning (Ruiz 2007). Hybrids of *O. bonariensis* and *O. hatcheri* were detected in this reservoir (Rueda et al. 2017).

*Percichthys trucha* is widely distributed in the Senguerr and Chubut basins (Quirós and Baigún 1986; Gosztonyi 1988; Di Persia et al. 1991; Ruzzante et al. 2006; Aigo et al. 2008; Ruiz and Fondacaro 2020). The life history of this species varies widely throughout its distribution in the basins due to the influence of abiotic factors (Guerrero 1991, Baigún 2001). This species is also present in low abundance in the Fontana and La Plata lakes (Di Persia et al. 1991; Ruzzante et al. 2006). In the Muster and Colhué Huapí lakes and the Florentino Ameghino Reservoir, very variable average and maximum lengths were recorded (Baigún 2001). Populations inhabiting environments located at higher altitudes or in colder areas attain larger sizes (Baigún 2001). The reproductive season may also differ between environments but typically occurs from late winter to spring (Baigún and De Nichilo 1985). Baigún (2001) suggested that the spawning of this species is linked to fluctuations in water level since an increase in volume increases the surface of vegetated areas suitable as refuges for the first life stages.

*Galaxias platei* is present in La Plata Lake and Senguerr River and in the past was also recorded in Colhué Huapí and Musters Lakes (Di Persia et al. 1991; Milano and Vigliano 1997; Cussac et al. 2004; Aigo et al. 2008). In the Florentino Ameghino Reservoir, this species was registered by Cussac et al. (2004) but was never found in the fishery surveys (Baigún 2001; Ruiz and Fondacaro 2020). In contrast, *G. maculatus* was reported only for the Chubut River basin by Pozzi (1945, in Ringuelet et al. (1967)) and Arratia et al. (1983), who probably only replicated old citations, since the species has not been recorded again in the last 40 years; this may indicate an erroneous record for the basin or local extinction of the species. *H. macraei* was recorded in the lower basin of the Chubut River (Ruiz and Fondacaro 2020). These authors observed gonads in preparation throughout the year, with a higher percentage in autumn and winter, and spawning was recorded at the end of the cold season.

The only native anadromous species reported in the Chubut basin is *G. macrostoma*, which enters the Chubut River every autumn as a subadult from the sea. Its distribution in the Chubut River is limited by a diversion weir located downstream of the Florentino Ameghino Dam, which prevents upstream migrations.

Salmonid introduction into the Chubut River began in 1910 with the release of *S. fontinalis* into the upper basin of the river. Between 1942 and 1943, *O. mykiss*, *S. salar*, and *S. fontinalis* were also introduced into Fontana Lake and Senguerr River, while the Mayo River was stocked with *O. mykiss* and *S. fontinalis* (Baigún and Quirós 1985). The last species introduced into the upper basin was *S. trutta* in the 1980s. In 1975, a fish hatchery began operating in the upper Chubut basin (Bagillt hatchery), releasing *O. mykiss*, *S. trutta*, and *S. fontinalis* juveniles into closed lakes and rivers.

## Fisheries

A gill net small-scale fishery has developed in the Musters Lake and occasionally in the Colhue Huapí Lake when the water level was appropriate. The main target species in Musters Lake are *P. trucha*, *O. hatcheri*, and *O. mykiss*. The catch was variable and the historical value was around 40 tons per year. This fishery has been considered underexploited based on the potential yield estimated by Baigún (2001), who suggested that plateau lakes can provide between 20 and 40 kg ha$^{-1}$. Fishermen suggest that this variability in captures could be attributed to an increase in water turbidity, mainly in seasons 2018 and 2019, thus hindering the *O. mykiss* catch (Fig. 13.6). Acoustic estimates carried out by the Provincial Fisheries Directorate determined a fish density of 22 kg ha$^{-1}$, which could be even higher as *O. hatcheri* biomass may have been underestimated as this species inhabits shore areas (< 4 m). In turn, the fishery of the Colhue Huapi Lake was based on only *P. trucha* (Quirós and Baigún 1986). The entire catch is processed in local fishing plants (Sarmiento City) and sold to major provincial cities (Bonacci 2013).

According to Castro Briz (2021), recreational fishing in the Chubut River basin is concentrated in the upper sector (El Maiten), where fly-fishing predominates and *O. mykiss* represents the main target species. In this section, *S. trutta* is also caught, but in low numbers, while all the specimens of *S. fontinalis* captured were of small size. In the middle basin, spinning and trolling take place in the Florentino Ameghino Reservoir. Target species are *O. my*kiss and *S. trutta*, and trophy weights of 10 kg and 6 kg can be reached, respectively. Less effort is devoted to the capture of *O. hatcheri* and *P. trucha*. People living along the river also catch *P. trucha*. Fly-fishing is the main fishing activity downstream of the dam being *O. mykiss* and *S. trutta* the target species (Castro Briz 2021). Very experienced fishermen catch between 150 and 200 trout per season in this section of the river.

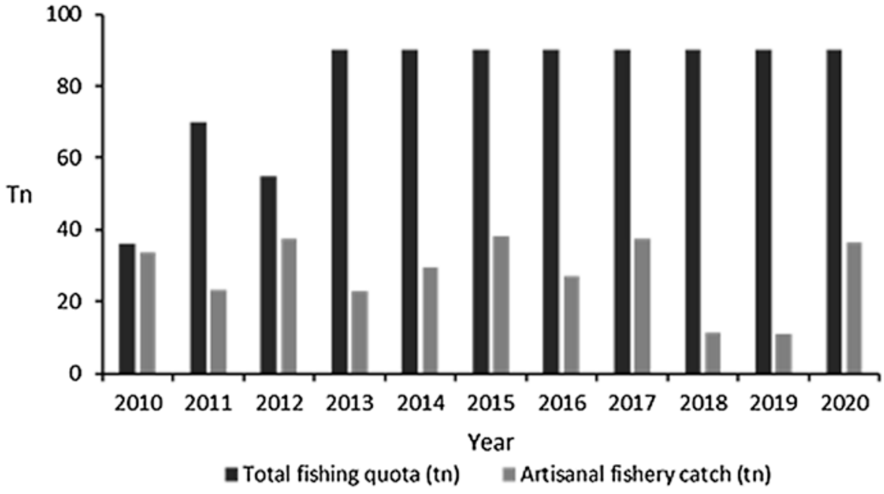

**Fig. 13.6** Artisanal fishery catches and permitted fishing quota in tons (t) per year in Musters Lake

## *6.4 Santa Cruz River*

### Ichthyofauna Characteristics

The fish fauna of Santa Cruz River is composed of four native species, *P. trucha*, *G. maculatus*, *G. platei*, and *G. macrostoma*, and four exotic salmonid species: *O. mykiss*, *S. namaycush*, *S. trutta*, and *O. tshawytscha* (Table 13.1). The marine/brackish species *E. maclovinus* can also be found in the inner area of the estuary and up to 45 km upstream of the outflow of the river into the Atlantic Ocean.

Over recent years, information has been gathered on the distribution and abundance of the fish fauna of Santa Cruz River because of the imminent construction of Condor Cliff and La Barrancosa hydroelectric dams on the main river channel (Riva-Rossi et al. 2018; Casalinuovo et al. 2018a, 2019). According to these studies, the native *G. maculatus* is the most abundant species throughout the main course of the river, particularly in the medium and lower sections, followed by *G. macrostoma* and *O. mykiss* in the upper and middle sections. Anadromous *O. mykiss*, once very abundant especially in the middle and upper courses, has dramatically declined in the last years, probably displaced by the highly invasive *O. tshawytscha* and the expanding *S. trutta*. *Eleginops maclovinus* is found exclusively in the lower course and the estuary of the river (Fig. 13.7). As noted by Tagliaferro et al. (2014) and Quiroga et al. (2015), the middle basin has the highest density of *G. maculatus* and juveniles of *O. mykiss.*

In the Santa Cruz River, information on anadromous and exotic species is more abundant compared to other large Patagonian basins. For example, the anadromous and semelparous *G. macrostoma* begins its life cycle in freshwater where the larval stage (ammocoetes) feed and grow buried in the mud for at least 4 years (Fig. 13.8). At the end of this period, the larvae metamorphose into parasitic juveniles and migrate to the ocean during the winter months, where they feed on fish blood and body tissues for more than a year. When juveniles are fully grown at sea, they cease to feed, and as subadults, they return to freshwater from summer through early fall (December to April), where they become sexually mature, spawn, and then die (Riva-Rossi et al. 2020a). Ammocoetes of *G. macrostoma* occur throughout the entire mainstem river, the southeastern shore of Lake Argentino, and La Leona and Chico rivers, whereas spawning adults have only been found in the upper course and La Leona River (Riva-Rossi et al. 2020b).

In Santa Cruz River, *G. maculatus* also exhibits two life history forms: resident individuals remaining in freshwater throughout their entire lives and diadromous individuals that are more abundant close to the estuary and whose juveniles develop and feed in the lower or middle course of the river (Carrea et al. 2012; Tagliaferro et al. 2014). The highest abundance of *G. maculatus* in the Santa Cruz River occurs in the middle and lower course (Tagliaferro et al. 2013, Riva-Rossi et al. 2018; Casalinuovo et al. 2019) and is associated with the increasing availability of habitats with fine sediments and calm, shallow waters (Tagliaferro et al. 2013; Riva-Rossi et al. 2018). This species has also been recorded along the shores of Lake Argentino

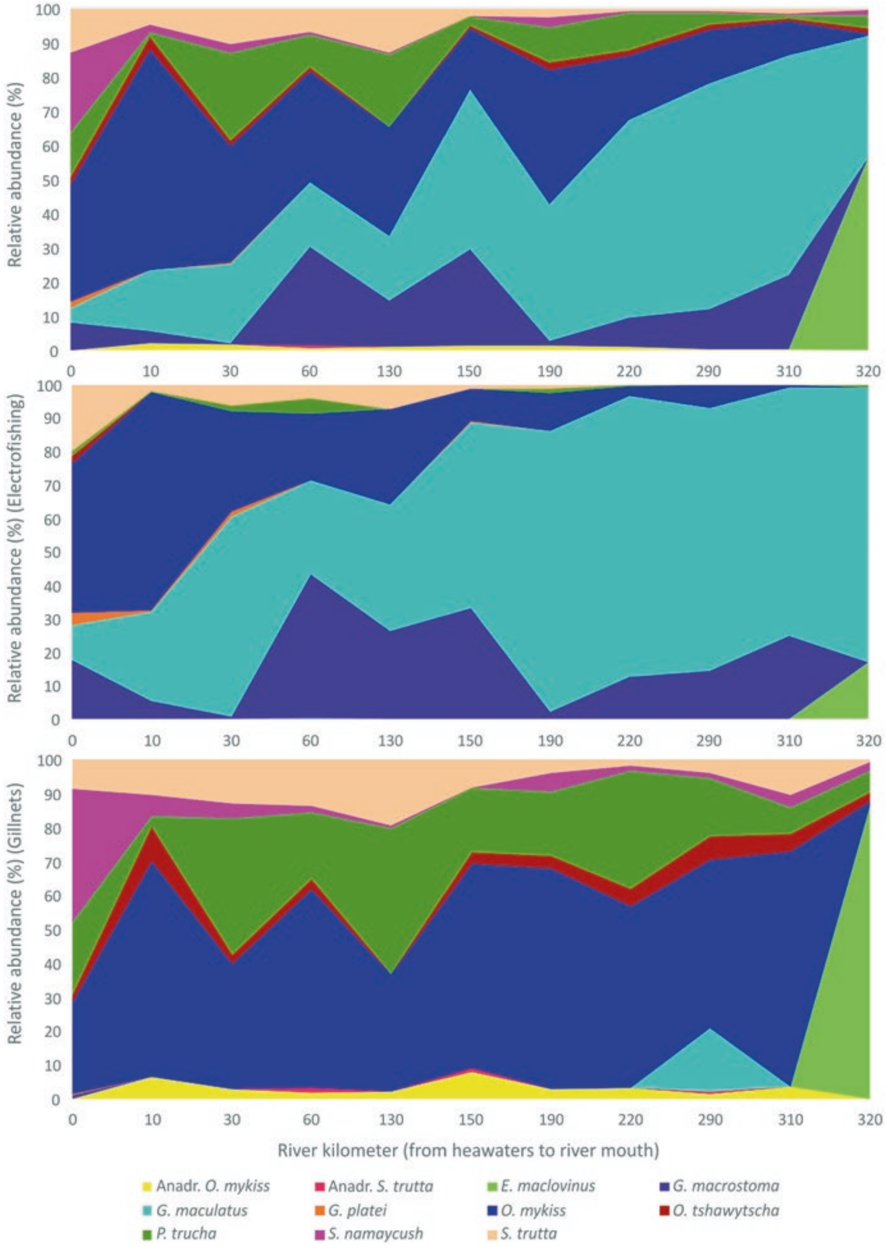

**Fig. 13.7** Stacked area chart representing the relative abundance of fish species throughout the Santa Cruz River, estimated from electrofishing and gillnetting surveys (2017–2019; Riva-Rossi et al. 2018, Casalinuovo et al. 2018c, 2019). The river kilometer (rkm) from the river origin headwaters to the river mouth is presented on the x axis. Zero rkm includes headwater locations, such as Argentino Lake and La Leona River; rkm 0–130 correspond to the upper basin, rkm 150–220 to the middle basin, and rkm 290–320 to the lower basin

and La Leona River but seems to be absent in the Viedma Lake and tributaries. In turn, the abundance of *G. platei* is relatively low compared to *G. maculatus* (Riva-Rossi et al. 2018, Casalinuovo et al. 2019), being more common in the upper mainstem of Santa Cruz River, in Argentino Lake, in its tributaries from the northern coastline and the southern tributaries of Viedma Lake, and in La Leona River (Casalinuovo et al. 2019).

*Percichthys trucha* has been detected in La Leona River, in Argentino Lake, throughout the mainstem of the Santa Cruz River, and in Chico River (Riva-Rossi et al. 2018; Casalinuovo et al. 2019; Riva-Rossi and Quiroga 2020). The abundance of this species increases toward the middle and upper reaches of the mainstem of Santa Cruz River, associated with low-velocity, deep pool habitats (Riva-Rossi et al. 2018).

The marine *E. maclovinus* is found exclusively in the lower course of Santa Cruz River, close to its outflow into the Atlantic Ocean, where pre-spawning adult individuals are caught in high abundance from late spring through late fall (October–June). Juveniles exhibit the highest abundance during summer (January) (Riva-Rossi et al. 2018; Casalinuovo et al. 2019).

The first introductions of exotic salmonids into Santa Cruz River took place from 1906 to 1910 based on eggs and embryos of *O. mykiss*, *O. tshawytscha*, *O. kisutch*, *O. nerka*, *S. trutta*, *S. salar*, *S. fontinalis*, and *S. namaycush* (Tulian 1908; Marini 1936; Riva-Rossi et al. 2004). At present *O. mykiss* is the predominant salmonid in Santa Cruz River; this species exhibits partial migration of anadromous and stream-resident individuals coexisting within the same population (Pascual et al. 2001b; Riva-Rossi et al. 2007). The migratory form, unique to Santa Cruz River in Argentina, is commonly known as “steelhead.” This ecotype spawns in freshwater and migrates to the ocean during spring, after 2–3 years of growing up to the juvenile stage (Pascual et al. 2001b). After migrating as smolts, *O. mykiss* feeds for less than a year in the ocean before returning to the river to spawn (Fig. 13.8). Once maturity is reached (at 3–4 years old in both forms), resident *O. mykiss* may repeat spawning up to four times annually, whereas anadromous *O. mykiss* may spawn annually up to eight times (Pascual et al. 2001b; Riva-Rossi et al. 2003, 2007). On the other hand, stream-resident *O. mykiss* spends its entire life feeding, growing, and spawning in freshwater. Quiroga et al. (2015) showed that throughout Santa Cruz River, the juveniles of anadromous and resident *O. mykiss* spawn primarily in the upper Santa Cruz River section.

*Geotria macrostoma*, *O. mykiss*, and ocean-type *O. tshawytscha* subadults enter the river from the ocean during high flow, while stream-type *O. tshawytscha* enters during low flow discharge. Spawning takes place at low flow in all the species and ecotypes (Ciancio et al. 2016; Riva-Rossi and Quiroga 2020; Riva-Rossi et al. 2020b) (Fig. 13.9). Smolts of *O. tshawytscha* and *O. mykiss*, and also adults of the latter, move downstream toward the ocean when discharge increases, while macrophthalmia stage of *G. macrostoma* moves downstream at high flow.

Confirmed records of *O. tshawytscha* in Santa Cruz River date back about 30 years, when escaped individuals from sea-ranching experiments in Chile strayed into Santa Cruz River and established feral populations (Ciancio et al. 2005;

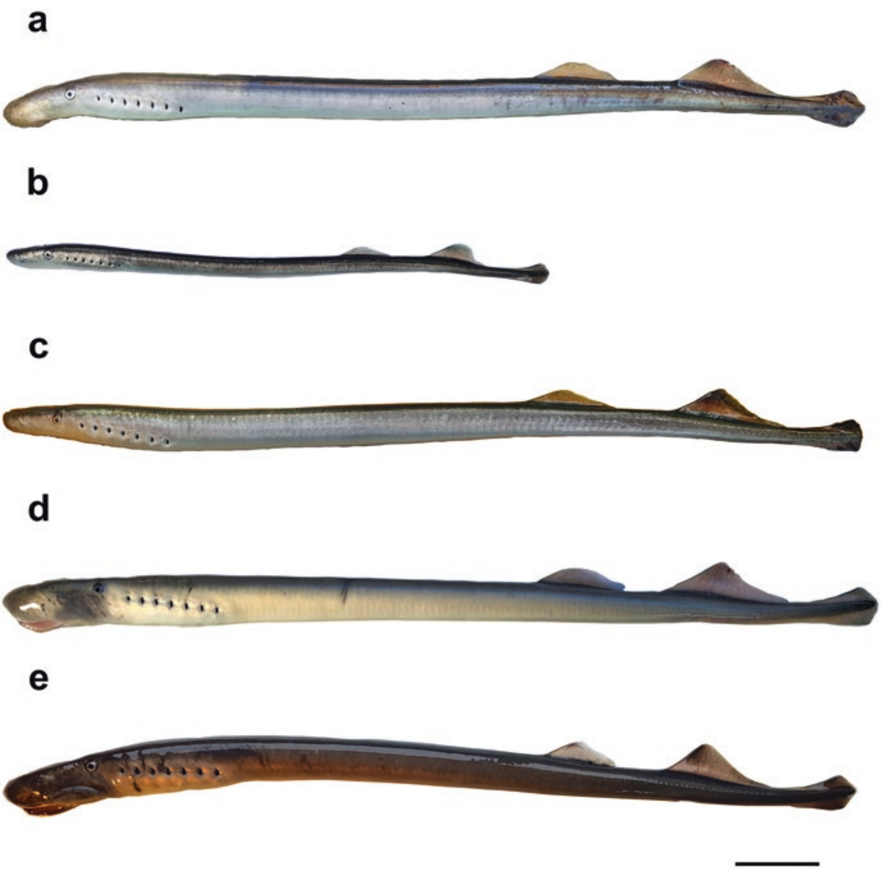

**Fig. 13.8** Immature and maturing *G. macrostoma*: (**a**) and (**b**) freshwater run *G. macrostoma* of different sizes, (**c**) after 1 week in freshwater, (**d**) sexually maturing female, and (**e**) sexually maturing male. Scale bars = 5 cm

Riva-Rossi et al. 2012; Ciancio et al. 2015). In the Santa Cruz River, there is a main spawning run of this species with a stream-type life history, normally entering the river during spring, from September to December, but there is also a very small run of salmon during the fall, from March to April (Fig. 13.9) (Riva-Rossi and Quiroga 2020). Spawning takes place within a short time window from late summer to early fall, from January through May (Ciancio et al. 2005).

*Salmo trutta* has established self-sustaining resident populations throughout the Santa Cruz River catchment, although in recent years, anadromous individuals or "sea-run brown trout" have become increasingly common in fish surveys (Riva-Rossi and Quiroga 2020). Stream-resident *S. trutta* occurs throughout the entire catchment of Santa Cruz River (Del Desierto Lake, De Las Vueltas River, and its tributaries, Viedma and Argentino lakes and their northern and southern tributaries, La Leona River, and the mainstem of the Santa Cruz River), whereas anadromous

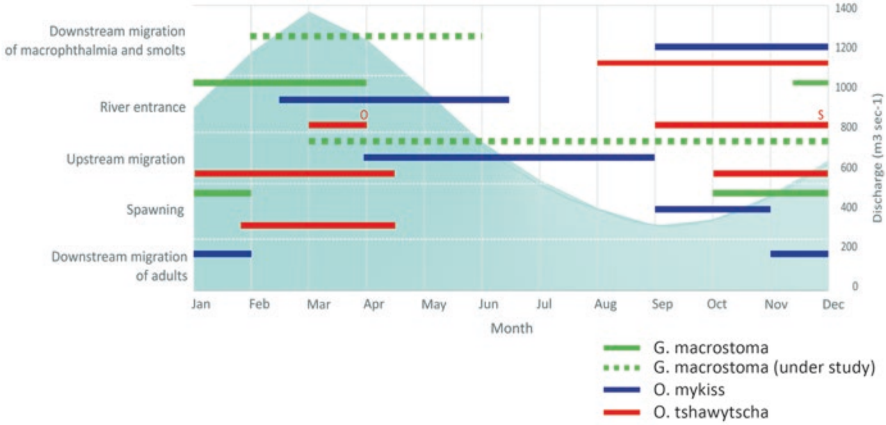

**Fig. 13.9** Cycle of anadromous species and its relation to flow rate

individuals have been recorded only in the mainstem river (Riva-Rossi and Quiroga 2020).

*Salvelinus namaycush* has established self-sustaining populations in the basin, mainly in the Argentine Lake and nearby streams, and to a lesser extent, they have spread into the Santa Cruz and La Leona rivers and Viedma Lake. This species is a top piscivorous predator and has little or no value for the recreational fishery or as a food source for local people. The spawning of lake trout occurs during late fall and winter (May–August) along lake shorelines. Juvenile has been documented exclusively on lake shores, whereas feeding adults are found both in the lake and in rivers.

## Fisheries

The recreational fisheries in Santa Cruz River are unique in Patagonia since in this almost pristine riverine habitat, and the main target species are two large anadromous salmonids. The most iconic game fish is the migratory form of a self-sustaining population of *O. mykiss* (Pascual et al. 2001b; Tagliaferro et al. 2013); however, in recent years *O. tshawytscha* has become the second most valued species (Llompart et al. 2019). Recreational fishing is carried out mostly in the lower basin near Piedra Buena City which is 40 km from the Atlantic Ocean, while occasional fishing activity also takes place in some lakes and rivers in the upper basin.

The *O. mykiss* fishing season starts in November and finishes in June when the steelhead migrates and passes through the main fishing sector. In contrast, the *O. tshawytscha* fishing season runs from mid-September to the end of October, in synchrony with the beginning of the upstream migration toward the upper basin (Fig. 13.10). During these months, angler participation increases notably and then decreases during the summer. This species is also captured in the Caterina and De Las Vueltas rivers located in the upper basin where species spawns (Ciancio et al. 2005, 2019). In addition, during April and May, the *E. maclovinus* fishery reaches

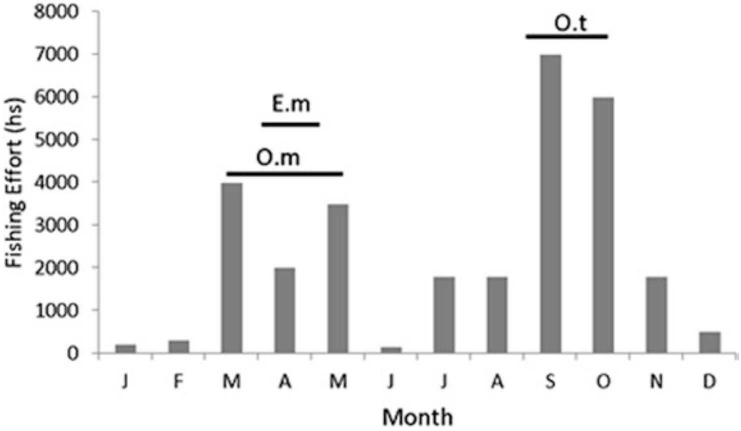

**Fig. 13.10** Seasonal pattern of the Santa Cruz River fishery. O.m *Oncorhynchus mykiss*, E.m *Eleginops maclovinus*, and O.t *Oncorhynchus tshawytscha*

**Fig. 13.11** Steelhead tournament on Santa Cruz River. (Photo by Claudio Baigún)

its maximum fishing effort, although the season is restricted to a short time between October and November. For more than two decades, local governments and other authorities have organized the "National Steelhead Trout Festival" in April (Fig. 13.11), which concentrates around 200 anglers from different Patagonian provinces.

Recreational fisheries raise great local interest, resident anglers accounting for half the participation, and 35% of nonresident/tourist anglers coming from villages and cities close to Santa Cruz River. According to angler motivations, the attractiveness of Santa Cruz River is more related to its natural characteristics providing human well-being for residents and visitors than for its fishing value. The annual monetary benefits for the community associated with the fishery have been estimated at around 25,000 USD, which could have some impact on the economic structure of Piedra Buena City (Llompart et al. 2019).

## *6.5 Gallegos River*

### Ichthyofauna Characteristics

In the Gallegos River and its tributaries, there are seven species, four of which are native: *G. platei*, *G. maculatus*, *G. macrostoma*, and *E. maclovinus*, a marine/brackish species whose juveniles can be found near the estuary in freshwater (Casalinuovo et al. 2016). The exotic species are *S. trutta*, *O. mykiss*, and *S. fontinalis*. A few individuals of *O. tshawytscha* have also been registered, although at present, it is considered that there are no self-sustaining populations (Casalinuovo et al. 2018b). *Salmo trutta* developed partially migratory populations composed of anadromous and resident individuals that belong to the same stock (Casalinuovo et al. 2018b), while *O. mykiss* and *S. fontinalis* populations are strictly resident in freshwater. Salmonids were introduced in the early nineteenth century (Tulian 1908; Marini 1936), and large numbers of eggs and embryos were released into the system from 1915 to 1937 (Colihueque et al. 2003; Valiente et al. 2007, 2010). Following this, no regular intensive stockings were performed.

*Salmo trutta*, *G. maculatus*, and *G. macrostoma* are the most widely distributed species in the basin, while *S. fontinalis* is mainly found in minor tributaries. *O. mykiss* is caught sporadically throughout the basin, while *G. platei* is very scarce (Casalinuovo et al. 2018a). Fish assemblage composition varies according to the environments: *S. trutta* is the dominant species in the mainstem, while *G. maculatus* is the most abundant in secondary channels, particularly in the lower reaches.

Information on species biology in the Gallegos River is still scarce except for *S. trutta* and even in this case only partially, since the species has been studied only intermittently, despite its economic importance (Amaya Santi and Pascual 2006; Casalinuovo et al. 2014, 2018a). It is assumed that *S. trutta* biology is similar to that of specimens from the Grande River in many aspects, particularly those associated with migration of the different stages. The size of the run has not been estimated, although it is presumed that the relative number is lower than for the Grande River. Fish reach sexual maturity between the third and sixth years of age, mostly during the fourth year. Sex ratios are biased in favor of females, both resident and anadromous. Smoltification is verified between the first and fourth years, with a large proportion in the second year. Some differences are nevertheless notable, such as a lower maximum age and higher mortality rate than the Grande River (Casalinuovo et al. 2014). Anadromous and resident fish show clear differences in growth, defined by where they feed (Casalinuovo et al. 2018a, b).

## Fisheries

Catch records from recreational anglers show *S. trutta* as an almost unique species, and the anadromous ecotype slightly exceeds 30% of the catch in the medium and lower basins but represents less than 15% in the upper basin (Casalinuovo et al. 2018a). This situation is similar in the upper basin of the Grande River, where this ecotype is reported to account for 20% of the catch in the upper zone. The fishing season extends from November to April and the regulations contemplate sectors and periods of catch and release; however, control is perceived by anglers as deficient (Casalinuovo et al. 2014). The size of anadromous *S. trutta* has meant that *estancias* (ranches) offer expensive accommodation and guiding services, usually for foreign anglers (fishing lodges). Three fishing lodges currently operate at a weekly cost per angler of USD 4000–6500. Lodges implemented a Fisheries Information System (FIS) similar to that developed in Tierra del Fuego, Antártida e Islas del Atlántico Sur Province, based on collecting data on fishing effort, catch per unit effort, and the size and weight of the fish (Casalinuovo et al. 2018b). The total catch records per season (January–March) show catches of around 1600 fish, mainly resident *S. trutta*. A guided angler captures four fish (one anadromous) per 8 hours of fishing, with an average length of approximately 654 mm for the anadromous trout and 389 mm for the residents (Fig. 13.12). Unfortunately, FIS was discontinued in 2016, and management is currently based on outdated information. In the estuary, the authorized fisheries are recreational (rods) and artisanal (nets), whose target species is *E. maclovinus*, and some recreational salmonid fishermen point this out as one of the main factors responsible for the decrease in fishing quality, in conjunction with poaching (Casalinuovo 2013; Casalinuovo et al. 2014).

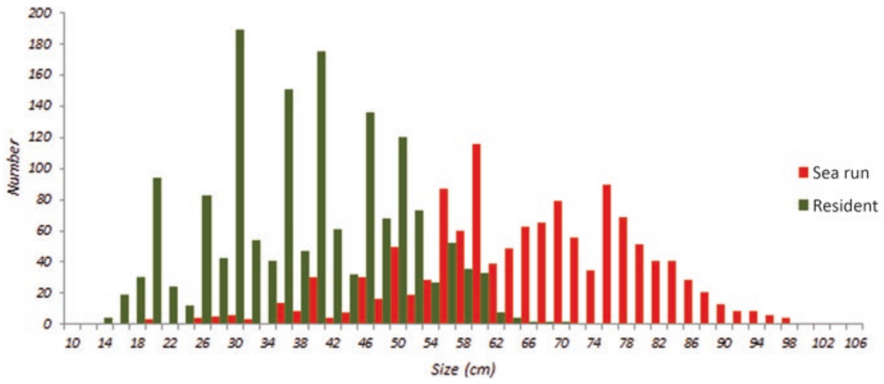

**Fig. 13.12** Size frequency distribution of *S. trutta* caught in the Gallegos River

## *6.6 Grande River*

### Ichthyofauna Characteristics

In the Argentine sector of the Grande River, there are seven species, of which four are native: *G. platei*, *G. maculatus*, *G. macrostoma*, and *E. maclovinus.* All exotic species are salmonids including *S. trutta*, *O. mykiss*, and *O. tshawytscha.* The only exotic species that appears to have strictly freshwater resident populations is *O. mykiss* whereas *S. trutta* developed partially migratory populations, as in the Gallegos River (Casalinuovo et al. 2014; Casalinuovo et al. 2018b). Salmonids were introduced in 1931 (Bruno Videla 1978), but after the year 2000, stocking was greatly reduced (Casalinuovo et al. 2002). *Oncorhynchus tshawytscha* enters the Grande River from Blanco River in the upper basin, where it spawns, but no records of spawning grounds have been kept in the Argentinean sector (Casalinuovo et al. 2016, 2018b). Records of adults that escaped from Chilean farms started in 2000 on the Argentinean side (Casalinuovo et al. 2002), and juvenile records (all fluvial ecotype) have been reported in the entire basin since 2016.

*Geotria macrostoma*, *S. trutta*, *O. tshawytscha*, and *O. mykiss* are found throughout the entire basin, while *G. platei* is very scarce, and *G. maculatus* and *E. maclovinus* have been caught mainly in the area near the estuary. The ichthyofauna is largely dominated in almost all environments by *S. trutta* (Casalinuovo et al. 2016, 2018b). The second most important species is *O. mykiss* in both the mainstem and tributaries. *Galaxias maculatus* inhabits refuge areas of vegetated secondary channels, where it can avoid predation and, consequently, reach high densities. Restricted basin-scale distribution and habitat-level spatial segregation of galaxiids are a strong indication of the impact of salmonid introductions. Species of juvenile salmonids have a sympatric distribution, occupying both the secondary channels and the shallow sites of the main channel. The abundance of these fishes decreases toward the estuary (O'Neal et al. 2007; Casalinuovo and Castro 2014). Catch records from recreational anglers show *S. trutta* as an almost unique species, and the anadromous ecotype exceeds 98% of catch records (Casalinuovo et al. 2018a). The catch data of the upper basin on the Chilean side shows that 80% of the records belong to the resident ecotype, similar to the Gallegos River.

Information on species biology in the Grande River is still scarce, except for *S. trutta*, the only species of abundance and recreational fishing importance. Movements are associated with the ontogeny of individuals. Immature specimens make lateral movements between the emergency sites and those for refuge and feeding. In addition, longitudinal migration occurs, such as during smoltification or reproduction. Adult movements are predominantly longitudinal and involve mainly the reproductive and trophic phases of their life cycle. Reproduction takes place from July to August and juveniles emerge from the gravel around the beginning of November (Casalinuovo et al. 2016), and as in Gallegos River, the sex ratio is biased in favor of females in both ecotypes. Once emerged, the juveniles are located in secondary channels or shallow sectors of the main channel, where they have abundant shelter and food. As they increase in size, they can move toward the main channel and to greater depths. It is suggested that smolts appear in spring, taking advantage of the seasonal floods after remaining in freshwater for between 2 (the majority) and 4 years (Giese 2011). Some still immature specimens return to wintering, as indicated by the capture of small anadromous fish by anglers at the end of the summer (Giese 2011; Casalinuovo et al. 2018a). The entry of anadromous *S. trutta* to freshwater is presumed to occur all year round, although there are two peaks in catches by anglers, one in mid-January and the other in February/March (O'Neal et al. 2007; Giese 2011). This peak occurs within 1 week after the maximum tidal amplitudes (Giese 2011). The relationship between catches and tides has been verified in other environments of Tierra del Fuego, such as the Irigoyen River (Casalinuovo et al. 2012). The run is age-structured, with the youngest and smallest individuals entering the system late (Giese 2011). Studies of migratory movements showed that more than 60% of the fish were located in the last 25 km of the river with a strong tendency to use a single site. Adults enter the river early, at the beginning of summer, traveling greater distances, while the juveniles are located mostly near the estuary (Casalinuovo et al. 2018b). Most specimens return to the sea from late spring, in conjunction with the increment in ocean productivity. Anadromous specimens do not feed in freshwater, except for certain prey items taken immediately after spawning, generally aquatic insects (Casalinuovo et al. 2016, 2018b); in contrast, resident and juvenile specimens prey all year-round on invertebrates, mainly of the genus *Hyalella* and *Chilina* (O'Neal 2008; Casalinuovo et al. 2016).

### Fisheries

Annually, 53,000–85,000 trophy size anadromous *S. trutta* uses the Grande River to spawn. This makes the river the best fishery in the world for this species, with memorable sizes and catch rates (an average size of 644 mm for anadromous and 296 mm for resident fish). The last 25 km of the river section concentrates 75% of anglers' catches, although the fish sizes show no significant differences from the rest of the river (Casalinuovo and Castro 2014). Fishing is mostly channeled through six fishing lodges, as in the Gallegos River, and about 5000 specimens are captured between January and March each year (O'Neal and Stanford 2011; Casalinuovo and Castro 2014; Casalinuovo et al. 2018b). These characteristics led to the establishment of an exploitation system of high economic and social value (O'Neal 2008; O'Neal and Stanford 2011; Casalinuovo et al. 2018b). A week of fishing costs USD 2500–7500, depending on the month and lodge (Casalinuovo et al. 2018b). Lodge operation records show that a guided angler captures four fish per 8 hours of fishing, with an average length of approximately 644 mm for the anadromous trout and 296 mm for the residents (Figs. 13.13 and 13.14); these values are among the highest reported worldwide (O'Neal and Stanford 2011). Remarkable marine growth rates, iteroparity, longevity, and maximum sizes reached by these specimens have been reported (O'Neal and Stanford 2011).

**Fig. 13.13** Grande River brown trouts can reach world-class trophy sizes. (Photo by M. Zanglá)

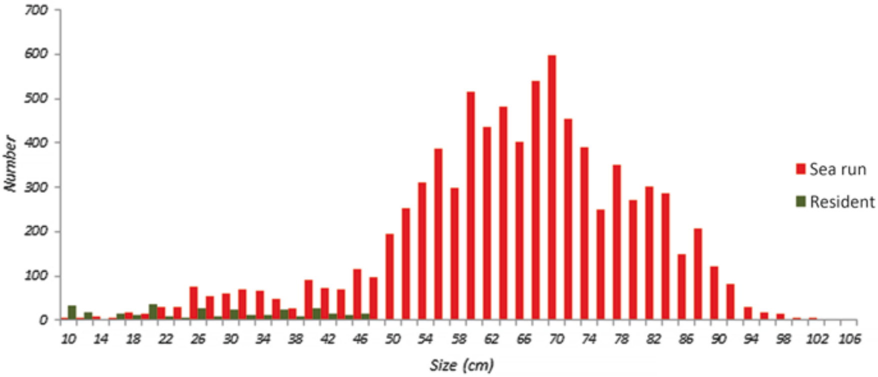

**Fig. 13.14** Size frequency distribution of *Salmo trutta* in the Grande River

The fishing season starts each year in November and finishes in May, although the main fishing pressure is concentrated in 15 weeks between January and March. In order to regulate this pressure and protect the fishery resource, the daily maximum number of anglers is regulated (the exact number slightly varies with fishing area and month). Fly-fishing and spinning are the approved fishing methods, the former being most commonly used. Regulatory norms eliminated open fishing in almost the entire river, and access to the system is through strictly controlled sites with a quota of rods, this being the only river in the country with this characteristic. The rod quota is distributed 70/30 between commercial (used by lodges) and no-cost fishing (only anglers resident in Tierra del Fuego). No-cost access is through an online reservation system for some sites that include tributaries. There is strict control of mortality, the regulation of fishing effort, and a catch and release policy being the only management tools. Management has been oriented through a participatory council composed of representatives of fishing lodges, the academic sector, provincial government, several NGOs, etc. This has complemented fishery studies and the catch data reported by guides, and all this input was used to draw up the Grande River Fisheries Information System that enabled modeling the fishery management based on different harvesting levels (García Asorey 2011). This pioneering system in Argentina, however, was discontinued in 2018, and fishery management is currently carried out in the absence of technical criteria and reliable data.

## 7 Impacts and Threats

The Patagonian steppe has suffered different types of human threat and impact, particularly after European immigration, and these are mainly associated with land use, exotic species introduction, and overgrazing (see Chaps. 10, 14, and 15). Freshwater communities in Patagonia have been influenced by the development of hydroelectric projects and alternative uses of natural stream flows, such as human consumption and crop irrigation (Pascual et al. 2007).

### *7.1 Species Introduction*

Of all fishes introduced into Patagonia, salmonids have been the most studied (Anderson and Valenzuela 2014), but their impact on native species has been considered variable, affecting native species abundance, distribution, habitat use, use, survival, growth, reproduction, behavior, and feeding (Vigliano et al. 2009; Casalinuovo et al. 2018a, b). The coexistence of salmonids and native fishes has mainly benefited from the spatial and temporal segregation of their breeding habitats and reproduction periods (Cussac et al. 1992; Barriga et al. 2002; Buria et al. 2007). In lakes, for example, the interaction between salmonids and native species takes place mostly in the littoral zone of the water bodies (e.g., Macchi et al. (1999), Ruzzante et al. (1998, 2003), and Milano et al. (2002, 2006)). Evidence of direct predation has been observed for *O. mykiss* on *G. maculatus* (e.g., Macchi et al. (1999, 2008), Milano et al. (2002, 2006), and Vigliano et al. (2009)) and *D. viedmensis* (Macchi et al. 1999), but a similar effect was also reported for *G. platei* in several Patagonian basins (Milano et al. 2002, 2006; Habit et al. 2010; Habit and Victoriano 2012). However, trophic interference is probably the mechanism that affects the native species most negatively, due to the presence of salmonids (Garcia de Leaniz et al. 2010; Elgueta et al. 2013). It is important to note that the impact of salmonids on native species can also occur indirectly by generating modifications in limnological conditions (Ortubay et al. 2006) and other biotic communities such as zooplankton (Reissig et al. 2006) and benthos (Buria et al. 2007).

The increasing presence of salmonids that escaped from Chilean farms (Becker et al. 2018) such as *O. tshawytscha* may generate new threats and conflicts in aquatic ecosystems and native and exotic species. This species has been targeted as a new successful invader of South American rivers (Correa and Gross 2008) and is also expanding across south Patagonia. One distinctive impact of this anadromous semelparous species is carcasses of dead individuals that provide low-productivity Patagonian rivers with a source of nutrients, stimulating the growth of algal biofilms and the autotrophic production of biomass. This subsidization of algae would enhance primary productivity, increasing invertebrate biomass and in turn food availability for salmonid juveniles (Benjamin et al. 2020; Kaylor et al. 2020). Although marine-derived nutrients from salmon carcasses have multiple benefits in ecosystems where *O. tshawytscha* is native, this process in Patagonian ecosystems remains unclear (Arismendi and Soto 2012).

Among non-salmonid exotic species, *C. carpio* represents a major threat as one of the most pervasive species on a global scale and one of the world's worst invasive alien species (Koehn 2004; Matsuzaki et al. 2009; Kloskowski 2011). This species spread notably throughout the Colorado River catchment during the 1980s (Fernandez et al. 1998), provoking changes in water quality in the irrigation channel located in the lower basin (Sidorkewicj et al. 1998). The spread of this species has also been registered in the middle and lower portions of Negro River (Alvear et al. 2007; Maiztegui et al. 2016; Solimano et al. 2019; Soricetti et al. 2020), where its increasing abundance seems to be related to a decrease in *G. macrostoma* larvae (Arakawa and Lampman 2020). Aigo et al. (2008) also suggested that the expansion of this species in the Limay River basin and toward the south and west, from the Colorado River basin to the Negro River basin, was due to the use of live bait (Aigo et al. 2008). Although *C. carpio* has never been recorded in southern Patagonian basins (Maiztegui et al. 2016), this species has been able to colonize cold-temperate environments located in a similar latitudinal range, as observed in the northern hemisphere (McCrimmon 1968; Scott and Crossman 1973; Chow-Fraser 1998). Another example of non-salmonid introductions that had a broad environmental impact was the stocking of *P. trucha* in Laguna Blanca Lake (Ortubay et al. 2006), even though the species is native to Patagonian waters.

### *7.2 Land Use*

The land use appears as one of the common and most pervasive impacts that affect the Patagonian steppe (see Chap. 10). Agriculture, afforestation, cattle, roads, and bridge construction represent potential impact sources not yet fully evaluated. A paradigmatic example is found in the Valcheta basin, where land use for cattle ranching (cows, horses, sheep, goats) has played a critical role in modifying the environmental conditions of the stream (Arellano et al. 2017; Velasco et al. 2016).

As cattle can impact riparian vegetation through trampling of the sediment, destruction of the river bed and banks, deterioration of water quality, reduction in water loss through evaporation, soil erosion, and reduction in deep drainage (Aguiar et al. 1996), these effects could affect Valcheta instream flow. Therefore, suggested protection measures include prevention of livestock entry, the raising of public awareness to prevent the stocking or release of new fish species, and changing the legal framework to prevent new exotic species introductions. Similar effects of livestock have been found in the middle and upper sectors of the Grande River, where the presence of cattle and sheep has affected the water quality and influenced salmonid distribution and abundance. Other impacts such as deforestation are increasing in northwest Patagonia (Laclau et al. 2002). This practice has the potential to reduce streamflow in drylands (Smith and Scott 1992), reducing microhabitat availability for small native fish species such as *H. macraei* and *Diplomystes* spp.

## *7.3 Pollution*

Land use in the different basins of Patagonia has also produced pollutants that can reach the rivers and are common in some populated and industrial areas such as the upper part of the Negro River. Evidence of anthropogenic activities in this river was found due to the detection of metals (Cu, As) and different types of hydrocarbon and pesticide (Miglioranza et al. 2013; Migueles et al. 2019). In turn, Macchi et al. (2018) recorded values of chlorpyrifos that exceeded the acute hazard quotient for invertebrates in the irrigation system of the upper valley of the Negro River (see Chap. 10), whereas other studies found high concentrations of pollutants in *C. carpio*, *O hatcheri*, and *O. mykiss* (Ondarza et al. 2012, 2014).

In the Grande River basin, the expansion of Rio Grande City is affecting the water quality through pollution and contamination due to sewage discharge and solid urban waste (Lofiego et al. 2009). Also, in the Lower Chubut River, Sastre et al. (1998) found evidence of eutrophication due to nutrient inputs, as this area has intensive agriculture activity, whereas hydrocarbons were detected in the fluvial sediments due to vessel activity in the ocean. In this river, urban expansion has also degraded the extensive floodplains of the lower sector (Ares and Serra 2008). Irrigation practices and artificial channel networks have facilitated the loading of nutrients from croplands (Liberoff et al. 2019).

## *7.4 Water Use and Hydraulic Infrastructure*

Dams are mostly located in the western sector of the Patagonian steppe, and their impact on most native fish populations is still poorly understood. Dams in the upper part of the Negro River basin have affected *G. macrostoma* migration, causing the disappearance of this species from the upper and middle parts of the Limay River upstream the dams (Pascual et al. 2007; Cussac et al. 2016). In the case of the Santa Cruz basin, two planned hydroelectric dams are expected to generate fragmentation of salmonid populations and block upstream migration of anadromous species, reducing spawning and breeding areas (Quiroga et al. 2015). Both dams will have far-reaching consequences for the ichthyofauna throughout the entire mainstem of the Santa Cruz River. The most severely affected species will be the migratory *G. macrostoma*, *O. mykiss*, and *O. tshawytscha*, as the dams will constitute migration barriers in the middle and upper courses to their spawning habitat and convert their riverine habitat to reservoir habitat (Riva-Rossi et al. 2020b). Furthermore, the dams will result in the population fragmentation and/or isolation of almost every species inhabiting the Santa Cruz River catchment. For example, the spawning and rearing habitats of *G. platei*, *P. trucha*, stream-resident *G. maculatus*, anadromous *O. mykiss*, *S. trutta*, and *O. tshawytscha* will become isolated or eliminated upstream the dams. In addition, the spawning and rearing habitats of diadromous *G. maculatus* will be restricted downstream of the dams while the stream-resident *O. mykiss* and *G. macrostoma* populations will be fragmented or extirpated.

The formation of reservoirs in the Santa Cruz River will generate additional shore habitats for *P. trucha*, whose feeding habits make use of both litoral and benthic areas (Ruzzante et al. 1998; Buria et al. 2007). Unlike other dams already installed in Patagonia, at least one of them will have fish passages to allow the ascent of anadromous salmonids and Patagonian lamprey. In the steppe, other dams lacking fishways have also blocked the migration of *G. macrostoma.* A similar problem has been detected with this species in the lower Chubut River due to a 1 m height weir located in the lower valley of the Chubut River to divert water for drinking and irrigation representing a major barrier to lamprey upstream migration.

However, the lack of fishways on the Limay and Neuquén Rivers had positive effects as they prevent the upstream displacement of *C. carpio* (Maiztegui et al. 2016). In addition, the reservoirs derived from these dams provided suitable lentic habitats for native species. Extensive shallow littoral areas are appropriate habitats for *G. maculatus* reproduction, and deep limnetic zones provide refuge for the larval stage (Cussac et al. 1992; Barriga et al. 2002; Rechencq et al. 2017).

Some of the other basins exhibit specific problems related to small-scale hydric infrastructure and water use. In the Gallegos River, water diversion and consumption have increased in the headwaters located in Chile (Diaz et al. 2016), thus modifying the downstream flow. Water diversion is also an important cause of drainage modification in the lower Senguerr River and affects fish species abundance. In this sector, there is no control over a large amount of water used for agriculture practices, cattle, and consumption which, in addition to the water deficit in the basin, has contributed to the drying of Colhue Huapi Lake (Scordo 2018). The consequence was the depletion of the *P. trucha* fishery. Of the annual discharge of the Senguerr River, 85% is used for agricultural livestock activity, farming, petroleum activity, and water supply to the coastal cities in the south of Chubut province and north of Santa Cruz province. In the Valcheta River, channeling and water storage may have accelerated the degradation of environmental conditions in the basin. Several of the problems with water use are related to its management and governance, which is closely linked to the legal/institutional frameworks of the provinces. In agreement with other cases in Argentina, there is still not a robust coordinated interprovincial framework that allows for a balanced use of water to meet the social but also ecosystemic demands (Trimble et al. 2020).

## *7.5 Poaching and Illegal and Bycatch Fishing*

Salmonids are subject to illegal practices in isolated areas due to poaching in different sectors of the basins or captured with illegal gears. In the estuaries, commercial fisheries may also have an impact due to bycatch, but there is no reliable information on this effect (Casalinuovo 2013).

## *7.6 Climate Change*

Climate change is another factor that can have a strong influence on fish assemblage composition and fishery performance. Climate change will strongly affect Patagonia by reducing river flows and impacting biota. This will be aggravated by an increase in water demand for different productive activities (e.g., agriculture and electricity generation). An increase in water temperature will have a severe effect on salmonid distribution in the steppe shallow lakes and the reservoir sectors used by salmonids (Aigo et al. 2008, 2014). Increased stream temperatures, coupled with the decreasing snowpack and precipitation predicted for Patagonia (Barros et al. 2014) (see Chaps. 1 and 4), will result in reduced streamflow and river discharge, influencing spawning habitat availability for mature fish and the rearing habitats of larvae and juveniles (Winfield et al. 2016). This effect may be very important in the small shallow lakes distributed across the steppe (Scordo 2018; Lancelotti et al. 2020), as these environments are strongly dependent on annual precipitation and snowfall (see also Chap. 14). This will also cause mismatches between the timing of physiological processes and life history events and the local environment, such as spawn timing, egg incubation, fry and larvae emergence dates, developmental rates, and upstream and downstream migration (Crozier et al. 2012). However, native Patagonian species such as *H. macrei*, *P. trucha*, and *O. hatcheri* appear to have a better physiological capacity to adapt to higher temperatures (Aigo et al. 2008, 2014; Cussac et al. 2009) as these species exhibit a higher preferred temperature (Becker et al. 2018). An increase in temperature would also lead to better conditions for the invasion of new Brazilic species into the north of Patagonia (Baigún et al. 2002). These authors pointed out that the expansion of Brazilic species from the Pampean region has been restricted by thermal conditions, but the warmer climatic scenarios predicted could generate more changes in north Patagonian fish assemblages.

## 8 Final Remarks

Knowledge of the fish and fisheries of the Patagonian steppe is still poor compared to that of the Andean region. This bias is rooted in different issues associated with scientific interests, human development, social demands, cultural values, and the distribution of protected areas. This review of information at the steppe scale based on fish assemblages and fishery knowledge shows important differences among basins, possibly associated with the amount of research carried out in the past. An example can be seen in the Santa Cruz River basin, for which there is a considerably better knowledge than for other basins, whereas in the Colorado River basin, the biological and ecological information of fish population is still very poor.

Specific fish richness in the steppe is slightly higher than that found in the Andean region, which can be associated with past geological events, modifications of fluvial landscapes due to anthropic impact, and introductions of exotic species. Fish assemblages have been greatly modified and shaped by non-native species introductions. The steppe has a lower proportion of salmonids than the Andean region but stands out for having anadromous ecotypes. The main difference from their Andean counterparts is that the plateau environments, due to their morphology and origin, have higher potential fish production and potential artisanal fisheries development. However, water quality, related to edaphic and climatic characteristics, can limit the distribution and abundance of certain species. Species differences are also found to follow a latitude pattern, which can be associated with a temperature range. The northern basins, which include the Colorado and Negro systems, present quite different conditions; this explains the successful introduction of Brazilic species, increasing species richness in north Patagonia. The expansion of *C. carpio* was favored by its naturally high invasive capacity, but also by the reckless and indiscriminate stocking practices of fishing associations and authorities, which represent a worrying threat. This trend could be exacerbated in the short term by the predicted climate change, as cold water inflow is modified and a shift in the steppe assemblage composition is favored.

Policies regarding exotic species introduction have differed between provinces and have been strongly influenced by stakeholders' attitudes and provincial agencies regarding stocking strategies. Historically, the strongest initiatives to introduce salmonids were concentrated in the Andean region and by developing practices to satisfy recreational fishing demands which are considered as a valuable ecosystem service in Patagonia. However, due to the hydrological configuration of the Patagonian basins, several of these species, particularly those with higher thermal tolerance, have expanded toward the plateau. The management of exotic species represents a critical aspect that merits an in-depth examination of the benefits and impacts observed after more than 100 years of introductions. Unfortunately, the integrated endeavor of the provinces and National Park Administration to develop a common regulatory framework, while at the same time facilitating management of recreational fishing, has not contributed to the conservation of native species. Because not all environments have similar ecological characteristics, it is important to carry out specific studies at the small watershed scale or even at a water body level in order to define specific management strategies that will preserve or restore native fish species. Possible trophic interference or direct predation on native species and the use of habitats for feeding, reproduction, and breeding are decisive elements of the possible impact of exotic species introduction. In this context, the uniqueness of some environments requires consideration of a much smaller scale of analysis than that applied by fishing regulations and considered by the provinces for species introduction practices. This perspective can provide more solid indications and recommendations for defining when salmonid introductions should be considered detrimental. Examples of these incongruities have occurred historically in many small high-altitude Andean lakes, plateau lakes, and closed basins, where salmonids never reached even acceptable sizes. The iconic case of the Valcheta basin, where salmonid fishing is regulated instead of being used as a tool to reduce exotic species populations or even bring about their extermination, reflects the shortsightedness of the state regarding the conservation of native species with extreme endemism. The main lesson from more than one century of introductions is that exotic species management should be reevaluated in terms of the social and economic benefits versus ecological impact.

On the other hand, the management of recreational or commercial fisheries in the steppe has been developed erratically and with little interest in monitoring their progress. Basic information on recreational fisheries is lacking in all the basins, with the exception of the Grande River. A characteristic feature of the Patagonian steppe rivers is the possibility of finding freshwater and estuarine species in their lower course, generating fisheries that are not found in the other sectors of the Patagonian basins. Such fisheries exhibit a unique complexity associated with tide cycles and magnitude that should be carefully evaluated. On the other hand, almost no data are available from the few artisanal fisheries that occur in some lakes. Steppe large lakes and reservoirs have a high potential for exploitation of species such as *O. mykiss* and *P. trucha*. In addition, they offer attractive trophy sizes for the recreational fishing sector. Undoubtedly, management of the fish populations in the Patagonian steppe will demand a better understanding of the ecological functioning of the systems. Their limnological and morphological conditions must be assessed and the impact of exotic species on aquatic ecosystems and their natural communities must be considered.

Finally, human activities (species introduction, pollution, overfishing, hydroelectric dams, flow diversions, etc.) are having an increasing impact on the ecological integrity of steppe aquatic ecosystems. Dams planned or already installed in Patagonia have generated an interesting debate, not only because they are high-head dams but also because they can block the migratory routes of native and exotic anadromous species and can extirpate them from the upper basin sectors. The ecological role of dams in the Patagonian steppe has been overlooked, as the formation of reservoirs has changed the way in which primary and secondary production is being channeled in trophic networks through processes that are absent in free-flowing rivers. Under a climate change scenario based on a reduction in precipitation in the steppe and a decreasing snowpack in the Andean region, the construction of additional dams to satisfy energy demands will require exhaustive evaluation of the environmental flows necessary to conserve the native biota of Patagonian plateau environments. Water management, on the other hand, represents a critical aspect that has negatively affected steppe water bodies and their associated biodiversity in some watersheds where irrigation demands are significant. No effective governance mechanisms or processes have been detected that take into account other stakeholders involved in the conservation or use of biological resources in the face of changes in the hydrological regime or flow reductions. More work is needed to gather better information on the array of lotic and lentic environments of the steppe basins and their fish assemblages and environmental drivers if we are to achieve integrated management policies on a watershed scale.

**In Memoriam** This chapter is dedicated to the memory of our colleague and friend Miguel Casalinuovo, who passed away in July 2022. Miguel was an enthusiastic researcher and a prestigious reference for recreational and sport fishing in Patagonia. The ichthyologists of Argentina will miss his presence, his incredible knowledge of the rivers, and his lucid vision and experience on the management of freshwater fisheries to which he dedicated

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# Chapter 14
# Influence of the Fish Introduction in Lakes of the Arid Southern Patagonia

**Sol Porcel, Juan Francisco Saad, María Cristina Marinone, Irina Izaguirre, and Julio Lancelotti**

## 1 Introduction

Patagonia is a vast region that covers 790,000 km$^2$ in Argentine continental territory, from the wet Andes to the dry Atlantic coast and from the temperate Colorado River to the north to the cold Tierra del Fuego to the south. This area can be divided into two subregions: Patagonian forest and Patagonian steppe (Morello et al. 2012) (see Chap. 1).

The Patagonian steppe encompasses approximately 20% of the Argentinean territory and is characterised by regular reliefs and by its arid to semiarid climate (Cabrera 1976), with a mean annual precipitation below 300 mm year$^{-1}$ (Jobbágy et al. 1995; Paruelo et al. 1998). An extensive system of elevated basaltic plateaus holds most of the freshwater of this subregion, including lakes larger than 300 km$^2$

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and thousands of shallow lakes (see Chap. 10). These plateaus were formed by tectonic activity during the Cenozoic (Panza and Franchi 2002) and include natural depressions that collect both rainfall and snowfall (Pereyra et al. 2002). These aquatic systems, which are naturally fishless, are characterised by a high diversity of morphometric and limnological characteristics (Lancelotti et al. 2009a), playing a crucial ecological role, since they constitute one of the few sources of water available to numerous species of animals, particularly for a diverse aquatic bird community. For this reason, Patagonian basaltic plateaus were declared relevant areas for bird conservation in Argentina (Di Giacomo 2005; Scott and Carbonell 1986). Particularly, this system constitutes the breeding habitat for *Podiceps gallardoi* Rumboll 1974, an endemic and critically endangered diving bird (Fjeldså 1986; Roesler et al. 2012; BirdLife International 2021). Moreover, other endemic organisms (e.g. vertebrates, invertebrates and algae) were recorded in these aquatic environments (Canevari et al. 1999; Menu-Marque et al. 2000; Ortubay et al. 2006; Coesel et al. 2017; Manzo et al. 2019) (see Chaps. 10, 11 and 15).

Exotic flora and fauna introductions have been common practices in Patagonia for more than a century. In particular, the introduction of salmonids and also the translocation of native fish species have been reported for many water bodies in this region (Perotti et al. 2005) (see Chap. 13). The introduction of *Oncorhynchus mykiss* (Walbaum 1792) in naturally fishless lakes of the Patagonian plateaus began around 1940, but over the last 20 years, the number of stocked lakes increased substantially, mainly for developing aquaculture and sport fisheries (Lancelotti et al. 2009a). These introductions have generated concerns about their potential consequences on the receiving communities in general and on the hooded grebe in particular (Lancelotti et al. 2009b, 2010b; Roesler et al. 2012).

The impact of climate variability on water systems is especially critical for the Patagonian region. Regional climate models project around 10–30% less precipitation over northern Patagonia by the end of the century, being the Patagonian forest subregion the most threatened one by this decline (Barros et al. 2014; Pessacg et al. 2020). Both the decrease of snowfall and the rise of temperature produce glacier recession and regional hydrological changes (Masiokas et al. 2008; Pasquini et al. 2008; Barros et al. 2014) (see Chap. 4). Moreover, the surface area of some lakes of the extra-forest Patagonian plains underwent a dramatic reduction since 2000 (Tejedo 2003; Llanos et al. 2016; Scordo et al. 2018a), thus increasing the desertification processes (Scordo et al. 2018b). Particularly, the lakes of the highland plateaus in the Patagonian steppe have shown a decrease in the water level of many shallow lakes and even the complete evaporation of some others (Roesler et al. 2012; Lancelotti et al. 2020).

The aim of this chapter is to summarise the current knowledge and ecological impacts associated with non-native fish introductions in lakes of the arid southern Patagonia. We provide an overview of the available information on planktonic and macroinvertebrate communities and on food webs in lakes of the two main Patagonian plateaus, the Buenos Aires Plateau and the Strobel Plateau, also assessing the influence of *O. mykiss* introductions and climate variability on these water bodies.

# 2 Characterisation of the Lakes of the Patagonian Steppe

## 2.1 *Topography*

The studied plateaus are located in the central and northwestern area of Santa Cruz Province, in the arid Patagonian steppe of Argentina (Fig. 14.1).

Topographically, the Strobel Plateau is relatively uniform with a mild upwards slope towards the northwest, varying in altitude from 1200 m a.s.l. westwards to 700 m a.s.l. eastwards (Lancelotti 2009). It has an extension of 3000 km$^2$ holding >1500 natural depressions fed by rainfall and snowmelt (Lancelotti et al. 2009a). It represents an extensive lacustrine system, including small temporary ponds as well as a few large lakes (~100 km$^2$). Most of these lakes are closed depressions, but a few of them are connected by temporary watercourses (Lancelotti 2009). On the other hand, the Buenos Aires Plateau also reaches its maximum altitude westwards

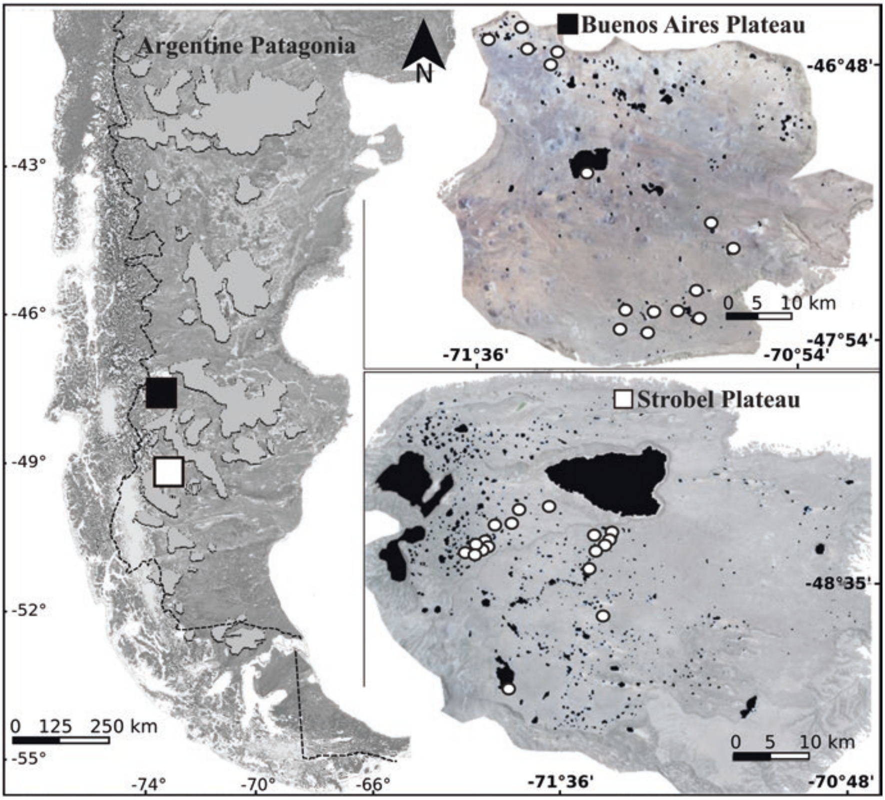

**Fig. 14.1** Basaltic plateau system from Patagonia (grey areas, adapted from D'Orazio et al. (2000)) and geographic location of the studied lakes (white circles) in the Buenos Aires Plateau and in the Strobel Plateau, in Santa Cruz Province, Argentina

(~1800 m a.s.l. with Mount Zeballos peaking at 2700 m a.s.l.) to decrease eastwards (700 m a.s.l.) (Roesler 2016). This plateau covers a total area of 3650 $km^2$ holding more than 150 endorheic shallow lakes.

## 2.2 Climate

The region is characterised by its extremely cold, windy and dry climate (Cabrera 1976). It presents a great variation in the photoperiod over the year, with temperatures varying between −22.4 and 38.5 °C (Lancelotti et al. 2020). The water regime is mostly determined by the balance between precipitation and evaporation, favoured by the high irradiance and strong winds that can reach up to 150 km $h^{-1}$ of speed (Correa 1998; Paruelo et al. 1998). Due to the frequent and strong westerly wind storms year round, even the deeper lakes are continuously mixed, preventing thermocline formation (Lancelotti 2009). All small lakes (<30 $km^2$) remain frozen from early autumn throughout late spring. This very dynamic lacustrine system is controlled by climatic variables, mostly precipitation (Lancelotti 2009; Lancelotti et al. 2010a, b). The scarce annual precipitation is the only source of water for these lakes, and thus it has a direct effect on their dynamics and limnological characteristics (Lancelotti et al. 2010a).

## 2.3 Limnological Characterisation

The lakes of this region are naturally fishless and range from oligotrophic to eutrophic (Izaguirre et al. 2018; Porcel 2020). *Myriophyllum quitense* Kunth is the dominant macrophyte, occurring in most low salinity lakes of this region (Lancelotti 2009). From a limnological point of view, the lakes of the Strobel Plateau have been classified into four general categories (Lancelotti et al. 2009a): turbid lakes, large vegetated lakes, large unvegetated lakes and ponds. In large vegetated lakes, macrophytes cover from 30% to 80% of the lake surface area, whereas in large unvegetated lakes, macrophytes cover less than 5% of the lake surface area. Small ponds (<9 ha) are usually completely covered by plants (~90%) (see Chap. 10). The turbid environments can be separated into two categories: organic and inorganic turbid lakes. The first group includes phytoplankton turbid systems, which could have small patches of macrophytes, and the second one presents environments with higher values of conductivity (>2 mS $cm^{-1}$) and is devoid of macrophytes (Izaguirre et al. 2018; Saad et al. 2019; Porcel 2020). This lake classification also applies to the water bodies of the Buenos Aires Plateau (Izaguirre et al. 2018; Porcel 2020). In addition, in this plateau, another category was described: clear mesohaline lake ("Honda"), which is a hard water lake because of its high concentration of calcium (bi)carbonate (Izaguirre et al. 2018; Porcel 2020), although these lake types are

**Table 14.1** Mean values and standard deviation (between brackets) of physical and chemical variables registered in the different types of lakes in Strobel Plateau and Buenos Aires Plateau

| Plateau | Regime | Fish | Conductivity (μS cm$^{-1}$) | Chl*a*(μg L$^{-1}$) | DIN (mg L$^{-1}$) | P-PO$_4$ (mg L$^{-1}$) | DOC (mg L$^{-1}$) | K$_d$ (m$^{-1}$) |
|---|---|---|---|---|---|---|---|---|
| STROBEL | Clear vegetated | Yes | 434 (292) | 10.24 (13.93) | 0.08 (0.07) | 0.21 (0.27) | 35.84 (26.67) | 2.04 (1.82) |
| | | No | 383 (433) | 5.50 (6.75) | 0.10 (0.18) | 0.32 (0.42) | 30.68 (15.15) | 1.98 (1.30) |
| | Organic turbid | Yes | 570 (354) | 7.95 (4.62) | 0.06 (0.03) | 0.17 (0.06) | 23.38 (9.59) | 2.29 (1.23) |
| | | No | 637 (315) | 18.52 (12.66) | 0.06 (0.06) | 0.21 (0.16) | 33.14 (20.80) | 2.95 (–) |
| | Inorganic turbid | No | 5670 (14) | 43.55 (18.47) | 0.11 (0.01) | 3.06 (3.82) | 136.80 (105.78) | 10.99 (14.13) |
| BUENOS AIRES | Clear mesohaline | No | 14,805 (121) | 2.72 (0.63) | 0.03 (0.02) | 0.08 (0.06) | 1442 (39) | 0.99 (0.10) |
| | Clear oligotrophic vegetated | | 67 (7) | 1.58 (0.61) | 0.06 (0.09) | 0.28 (0.20) | 1.41 (0.88) | 1.00 (0.71) |
| | Clear vegetated | | 121 (68) | 9.95 (10.99) | 0.03 (0.03) | 0.01 (0.01) | 9.52 (7.67) | 2.11 (1.66) |
| | Organic turbid | | 237 (143) | 11.71 (12.22) | 0.11 (0.10) | 0.30 (0.44) | 45.42 (22.35) | 5.45 (3.58) |
| | Inorganic turbid | | 522 (777) | 59.67 (57.66) | 0.02 (0.01) | 0.29 (0.17) | 43.77 (64.81) | 28.11 (–) |

*Chla* chlorophyll a, *DIN* dissolved organic nitrogen, *P-PO*$_4$ soluble reactive phosphorus, *DOC* dissolved organic carbon, $k_d$ vertical photosynthetically active radiation (PAR) attenuation coefficient

poorly represented on these aquatic systems. The main limnological features of the lake regimes of these plateaus are summarised in Table 14.1.

## 3 Fish Introduction

Biological invasions are recognised as a threat to biodiversity, generating alterations at multiple ecological levels (Mack et al. 2000; Simon and Townsend 2003). Fish introduction is a common and often successful tool for managing commercial or recreational fisheries (Wiley 2006; Gozlan 2008). However, this practice is among the main causes of impact in aquatic environments (Bahls 1992; McNaught et al. 1999; Sala et al. 2000; Parker and Schindler 2006; Schabetsberger et al. 2009). Particularly, the introduction of top predators into fish-lacking habitats has triggered trophic cascades and ecosystem alterations: changes in the abundance and composition of species, bioavailability and recirculation of nutrients, changes in primary production and shifts in the behaviour of organisms in the receiving lake (Carpenter and Kitchell 1993; Mittelbach et al. 1995; Scheffer 1998; Schindler et al. 2001; Schindler and Parker 2002). For example, the introduction of strictly planktivorous fish can lead to a reduction in the grazing pressure of zooplankton on phytoplankton, triggering an increase in algal biomass (Carpenter et al. 1985). Although there are fewer examples of the effect of generalist fish, which can feed on both pelagic and benthic prey, it is known that they can create a strong coupling between habitats (e.g. Schindler and Scheuerell (2002), and Vander Zanden and Vadeboncoeur (2002)). Likewise, the effects of generalist fish appear to depend on the introduced species and the trophic structure of the environment.

Salmonids were widely introduced around the world, and the available information suggests that their impacts may be severe and rapid (Crowl et al. 1992; Townsend 2003). The size-selective feeding behaviour of salmonids can change size structure and species composition of zooplankton and macroinvertebrate communities (Anderson 1980; Newman and Waters 1984; Gliwicz and Pijanowska 1989; Knapp et al. 2001). Frequently, predation on the native fauna by introducing salmonids causes ecological changes in higher levels, causing the elimination of amphibian and reptile populations (Tyler et al. 1998; Knapp 2005).

In rivers and lakes of Patagonia, the introductions of salmonids began at the onset of the twentieth century (Macchi et al. 1999). By the 1980s, all five Patagonian provinces had their own hatcheries, continuing the spread of salmonids up to the present (Pascual et al. 2002), being nowadays a widespread activity (see details in Chaps. 11 and 13). Particularly, in lakes of the Patagonian steppe region, the stocking and translocation of exotic fish have generated local extinctions of cladocerans species, reduction in zooplankton size and a general homogenisation of the systems (Modenutti and Balseiro 1994; Ortubay et al. 2006; Reissig et al. 2006; Buria et al. 2007), also causing the impoverishment of bird and amphibian fauna (Perotti et al. 2005; Cuello et al. 2006; Ortubay et al. 2006) (see Chap. 15). Specifically, some water bodies of the Patagonian basaltic plateaus, which are naturally fishless (Lancelotti 2009; Lancelotti et al. 2017), were stocked with *O. mykiss* during the last decade. In the present chapter, we discuss all the findings related to fish introductions in shallow lakes of the Strobel Plateau so far published. These studies have evidenced changes at several organisation levels in the aquatic communities due to *O. mykiss* introductions.

## 4 Influence of Fish Introduction on Autotrophic and Heterotrophic Planktonic Communities

The introduction of omnivorous fish in shallow water environments is associated with both top-down and bottom-up controls that promote the structuring of aquatic food webs. In the first type of control, fish can reduce by predation the abundance of larger zooplankton thus enhancing phytoplankton biomass (Blanco et al. 2008). In the latter case, phytoplankton development is triggered by an increase of nutrients in the water column, which can occur by direct nutrient excretion by fish (Vanni 1996, 2002) and/or sediment resuspension generated by fish feeding on benthic organisms (Schauss et al. 2002). Both processes can contribute to the shift and/or permanently turbid state of lentic water bodies (Blanco 2001).

The influence of fish introduction on the microbial planktonic communities in naturally fishless lakes was investigated in water bodies of the arid Patagonia since the beginning of the twenty-first century. One of the first studies was carried out by Reissig et al. (2006), who surveyed 18 lakes located in 4 zones of the Patagonian steppe, including fishless lakes, water bodies with only native fish and lakes where exotic fish were introduced. Besides the contrasting zooplankton structure that was evidenced between fishless lakes and fish stocked lakes – aspect that is addressed further in this chapter – this study also revealed evident differences in the phytoplankton structure. Fish stocked lakes showed higher phytoplankton abundance and a predominance of the net fraction (>20 μm) over nanoplankton; conversely, fishless lakes presented lower phytoplankton abundances and a higher proportion of nanoplankton. Another important difference was the relative abundance of cyanobacteria, which was notoriously higher in fish stocked lakes; some of the registered taxa of this group were *Aphanocapsa delicatissima*, *Dolichospermum flos-aquae*, *Microcystis aeruginosa* and *Oscillatoria tenuis*. Considering that the increase of cyanobacteria is an undesirable change in a water body, this finding was the first alert of the potential effect of the fish introduction in Patagonian lakes, providing valuable evidence for the conservation of the aquatic systems of the region.

In general, lakes from Patagonian plateaus range from mesotrophic to eutrophic and are characterised by higher phytoplankton biomass in comparison with those of the forest region (Izaguirre et al. 1990; Izaguirre 1993; Díaz et al. 2000). For the lakes located in the basaltic Patagonian plateaus, the ecological investigations on the phytoplankton communities started in 2007, and the first results were published by Izaguirre and Saad (2014). This study included 12 lakes from the Strobel Plateau and revealed differences between fishless and fish stocked lakes. The highest total phytoplankton biovolumes were found in fish stocked lakes, with a higher contribution of cyanobacteria, accounting for 84% of the total biovolume in one lake.

Within the framework of the same research project, the microbial plankton communities of the lakes located at the main basaltic plateaus of Santa Cruz (Strobel and Buenos Aires) were surveyed over a 7-year period. Particularly, some water bodies of the Strobel Plateau were visited during several late spring and summer campaigns. One of the published studies involved 14 lakes, which were sampled in December 2011 and March 2013 and were classified into three types according to the lake regime and the presence or absence of fish: fishless clear unvegetated lakes, fishless clear vegetated lakes and phytoplankton turbid fish stocked lakes (Saad et al. 2019). The study was focused on the autotrophic planktonic communities of all size fractions (pico, nano and micro), and the picoplanktonic fraction was assessed by flow cytometry. One of the main findings of the study was that the lakes exhibited differences in the limnological variables mainly in relation to the lake regime and the presence or absence of fish. On the other hand, in general, the abundances of autotrophic picoplankton and phytoplankton >20 μm were higher in fish stocked lakes, in agreement with the pattern observed in the studies previously described. The higher phytoplankton biomass in fish stocked lakes was attributed to a probable top-down effect. On the other hand, all three types of lakes presented phycoerythrin-rich picocyanobacteria (PE-rich Pcy) and picoeukaryotes (Peuk), but phycocyanin-rich picocyanobacteria (PC-rich Pcy) were only present in fish stocked lakes. In general, PC-rich Pcy are more efficient in more turbid and eutrophic aquatic environments, whereas PE-rich Pcy prevail in clear oligotrophic lakes, as was discussed by many authors (e.g. Stomp et al. (2007), Schiaffino et al. (2013), and Izaguirre et al. (2014)); thus, the presence of PC-rich Pcy in fish stocked lakes would be consistent with this pattern since these lakes presented higher turbidity (Fig. 14.2).

It is important to point out that the lower transparency in fish stocked was not attributed solely to the presence of fish, because also the resuspension of sediments during lower water levels may be important.

In a later study conducted in lakes of the Strobel Plateau, Porcel (2020) found lower mean abundances of bacterioplankton in fishless lakes than in fish stocked lakes; and among the stocked lakes, those clear vegetated presented higher abundances than the organic turbid. As for photosynthetic picoplankton, slightly higher abundances were found in turbid organic lakes with fish introduction. Although the three types of autotrophic picoplankton were present (Pcy-rich Pcy, PE-rich Pcy and Peuk), opposing the results obtained by Saad et al. (2019), PC-rich Pcy were more abundant in fishless clear vegetated lakes. The difference with the pattern previously observed was attributed to the water level decrease that occurred in the lakes during the years of sampling (2016 and 2017), which was accompanied by an increase in their turbidity. In the Buenos Aires basaltic Patagonian plateau, where all lakes lack fish, studies on the microbial planktonic communities were also carried out in lakes with contrasting regimes (clear vegetated and turbid) using flow cytometry (Porcel et al. 2019). In that paper, the effect of the water level decrease was also explored, showing higher abundances of heterotrophic bacterioplankton in more eutrophic and turbid lakes and the lowest values of autotrophic picoplankton in clear oligotrophic vegetated lakes. In these water bodies, PE-rich Pcy cells were more frequent than PC-rich Pcy cells. The water level decrease led to a switch from a clear vegetated regime to a turbid regime in some lakes, and under these conditions, an increase in heterotrophic picoplankton, PC-rich Pcy and Peuk was observed.

## 5 Influence of Fish Introduction on Zooplankton and Macroinvertebrate Communities

The zooplankton communities of lakes and ponds of the Patagonian steppe have a remarkable biogeographical importance, due to the dominance of endemic species of calanoid copepods (*Boeckella* and *Parabroteas*), cladocerans (*Daphnia*, *Macrothrix* and *Ilyocryptus*) and rotifers (*Brachionus*, *Keratella* and *Notholca*) that are well adapted to the harsh environmental conditions of this arid zone (José de Paggi 1990; Menu-Marque et al. 2000; Adamowicz et al. 2004; Reissig et al. 2006;

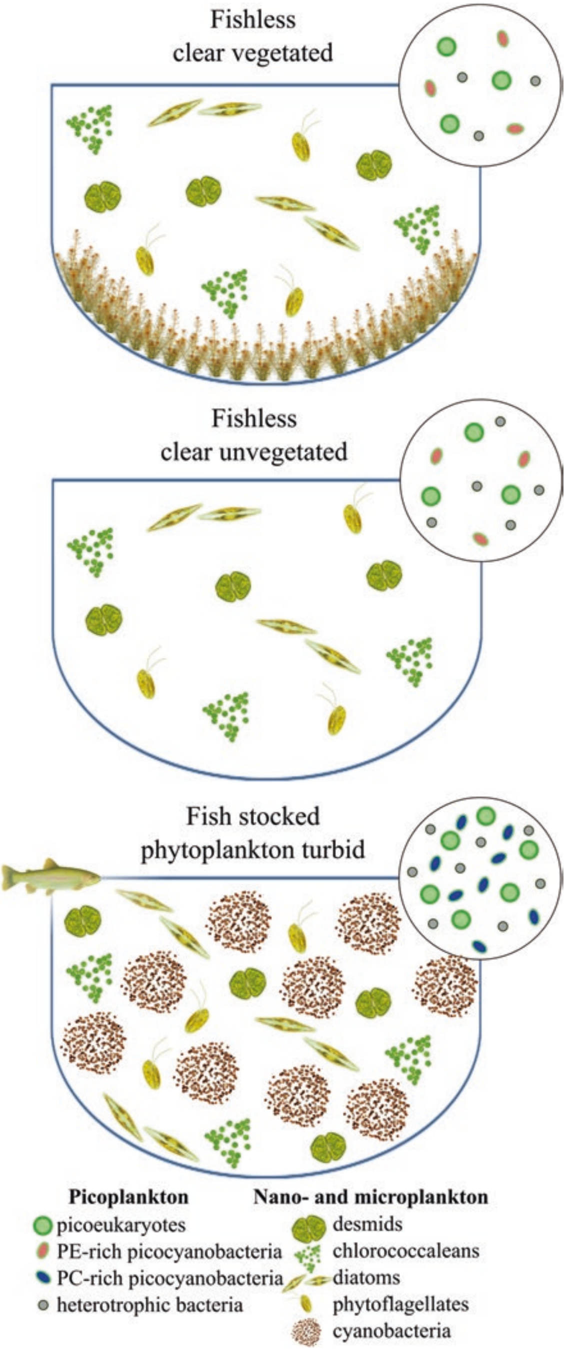

**Fig. 14.2** Diagram illustrating the typical structure of the microbial planktonic communities (pico-, nano- and micro-sized fractions) in fishless (clear vegetated and clear unvegetated) and fish stocked lakes

Marinone and Menu Marque 2010; Izaguirre et al. 2018). The regional zooplankton species richness in the studied plateau lakes of Santa Cruz Province encompasses 22 rotifers and 17 microcrustacean species, varying in individual lakes from 0 to 10 and 1 to 7, respectively, with mean local species richness of 5 rotifers and 4 microcrustaceans (Porcel et al. 2020). Despite the hostile environment, the occurrence of heterogeneous types of water bodies promotes a high planktonic diversity (Porcel et al. 2020).

The fishless lakes hold the largest representatives of the macrozooplankton species, including *Boeckella longicauda*, *Boeckella brasiliensis*, *Boeckella poppe*i (≤3.3 mm) and *Parabroteas sarsi* (≤6.5 mm) among the copepods, and *Daphnia dadayana*, *Daphnia menucoensis*, *Daphnia commutata* and *Daphnia* "obtusa group" (2–3.5 mm) among the cladocerans (Reissig et al. 2006; Lancelotti et al. 2017). Similar species composition and size structure can be found in turbid organic lakes with suspended inorganic matter, which provide UV radiation protection and visual refuge to zooplankters from predatory fish (Marinone et al. 2006). Additionally, *Boeckella meteoris* is frequent in turbid inorganic lakes (Lancelotti 2009). The frequently co-occurring *D. dadayana* and *D. commutata* and different species of *Boeckella* share most of the fishless extra-forest water bodies; however, the large *D. dadayana* are better adapted to coexist with the big predatory copepod *P. sarsi* (Marinone and Menu Marque 2010).

The highest total abundance and biomass of zooplankton and macrozooplankton were recorded in the fishless environments, where large microcrustaceans are dominant (Reissig et al. 2006; Lancelotti et al. 2017; Izaguirre et al. 2018). Conversely, in clear lakes with high predation pressure by fish, zooplankton communities encompass distinctly smaller crustacean species such as *Boeckella michaelseni*, *Boeckella gracilipes*, *Boeckella meteoris* (≤1.7 mm), *Ceriodaphnia* spp. and *Bosmina* spp., several genera of the Chydoridae family (0.3–1.3 mm) and rotifers (0.1–0.2 mm), which become dominant, thus decreasing total zooplankton biomass values and narrowing the size spectrum towards a higher frequency of smaller sizes (Reissig et al. 2006; Lancelotti et al. 2017). Trout predation on the larger macrozooplankton, such as *Daphnia*, could favour rotifer development (Izaguirre et al. 2018) by releasing them from exploitative competition and mechanical interference (Porcel et al. 2020). However, the presence of fish influences rotifer diversity but not microcrustacean diversity (Porcel et al. 2020).

Concerning the diverse macroinvertebrates that live in the water bodies of the Patagonian steppe, the community structure differs in the relative abundance of Ostracoda, Amphipoda (*Hyalella*), Chironomidae larvae and the coleopteran *Haliplus* depending on the presence or absence of fish (Lopez et al. 2021). It is remarkable that pelagic *Hyalella*, the larger crustacean in these lakes and the primary prey of trout, are absent in stocked lakes as compared to their high abundance in fishless lakes (Lancelotti et al. 2017). Additionally, a recent research on food webs evidenced that insect larvae of the families Leptophlebiidae (Ephemeroptera) and Chironomidae (Diptera) appeared only in fishless lakes (Porcel 2020; Porcel et al. submitted). The extinction and decline of aquatic insects due to fish introduction have been extensively covered in literature, as is the case of Ephemeroptera populations (Luecke 1990; Caudill 2003; Epanchin et al. 2010). Consequently, the decrease in the abundance of insect larvae could suppress the grazing pressure on the periphytic community, thus enhancing the development of algal biomass and changing algal species composition (Townsend 2003) as observed in some water bodies of the Strobel Plateau (Izaguirre and Saad 2014; Saad et al. 2019). The changes in the zooplankton and macroinvertebrate communities observed in stocked lakes are summarised in Fig. 14.3.

## 6 Influence of Fish Introduction on Food Webs

The introduction of non-native species could cause structural and functional alterations of food webs (Townsend 2003; Vander Zanden et al. 1999b). Stocked fishes, such as salmonids, are often top predators that either replace native top predators or increase the species richness of top predators, which can alter trophic relationships through cascading effects affecting the basal resources (Carpenter and Kitchell 1996; Scheffer et al. 2001; Eby et al. 2006). Additionally, a reduction in native species or the presence of non-native fishes can produce novel species interactions that have not coevolved (Tagliaferro et al. 2014) and could lead to a system with unstable trophic characteristics (Vander Zanden et al. 1999a).

Shallow lakes are dynamic ecosystems with a spatial structure characterised by a deep pelagic (open water) zone and a shallow benthic/littoral zone (Scheffer 1998), coupled with biological, physical and chemical processes (Schindler and Scheuerell 2002). In some water bodies, pelagic zone processes are described by complex interactions, which may include both the littoral and benthic zones of the lake (Pace 1993; France 1995a; Jeppesen et al. 1997; Covich et al. 1999; Rothhaupt 2000; Schindler and Scheuerell 2002; Rautio and Vincent 2007). Consequently, the magnitude of food web alterations, in this case of fish introductions, may depend on multiple factors related to the particular characteristics of each water body (Carpenter and Kitchell 1993; Pace et al. 1999; Polis 1999; Polis et al. 2000; Beisner et al. 2003). Fish are the main determinants of the food web structure and dynamics in shallow lakes (Jeppesen et al. 1997) and play an important role as habitat couplers because of their high mobility and inter-habitat omnivory (Schindler and Scheuerell 2002). In this context, fish would integrate both the benthic and the pelagic food webs, which could alter nutrient and energy pathways of the stocked lakes (Vander Zanden et al. 1999b; Schindler and Scheuerell 2002; Vander Zanden and Vadeboncoeur 2002).

Despite the abundant information about the negative effects of fish introductions on native freshwater ecosystems, very few studies give a clear image of how receiving communities might have looked before introductions, and scarce data are available on food web structure before these practices began, compromising the proposal of management tools for the conservation of the environment. Shallow lake systems of basaltic plateaus of Patagonia, with stocked and fishless lakes, offer the

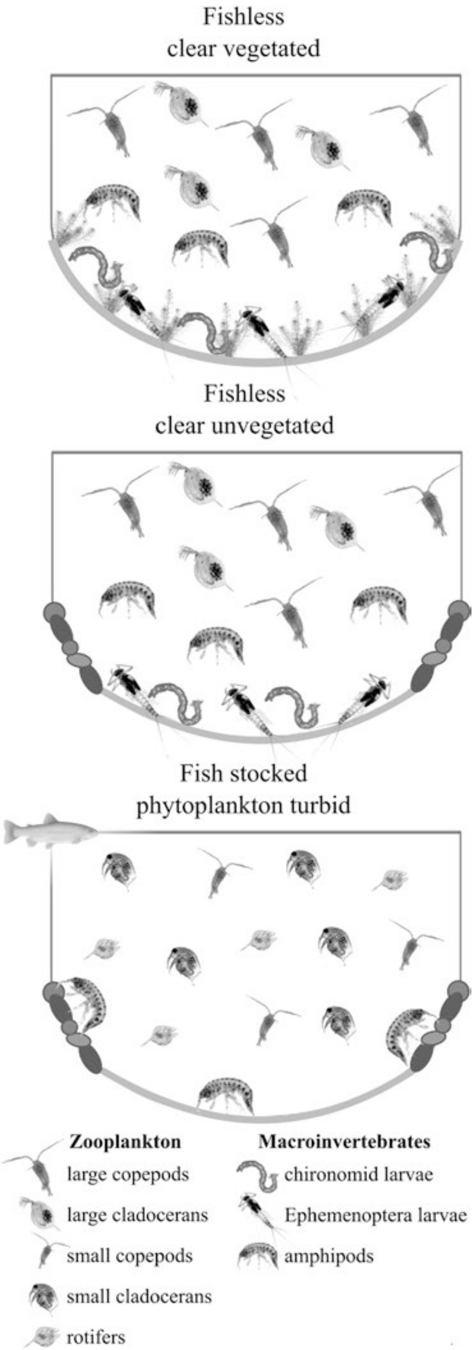

**Fig. 14.3** Diagram illustrating the typical structure of the zooplanktonic and benthic macroinvertebrate communities in fishless (clear vegetated and clear unvegetated) and fish stocked lakes

opportunity to evaluate the effects of the introduction of *O. mykiss* on the structure and functioning of the food web.

Food webs of a naturally fishless lake ("El Cervecero") of the Buenos Aires Plateau, which was one of the most important habitats for the reproduction of *P. gallardoi*, were described through the stable isotope analysis (SIA) (Adami 2016). This aquatic community comprised eight taxonomic groups, dominated by crustaceans, insects, molluscs and leeches (Hirudinea). They occupied three trophic levels, including the most conspicuous groups of organisms, such as *Lymnaea* or Dytiscidae larvae which are potential grebe's prey (Beltrán et al. 1992). The main sources of energy in this system seem to depend on benthic resources. Apparently, the structure of this community could be explained by its high macrophyte coverage and its low depth, favouring water transparency (allowing the proliferation of benthos) and providing refuge for zooplankton against predation. In this environment, *P. sarsi* (in the pelagic pathway) and Hirudinea (in the benthic pathway) were the only organisms occupying the third trophic level. The author suggested that *P. sarsi* was clearly separate from the rest of the organisms indicating that this group is sustained by pelagic sources of matter and energy, even in an environment dominated by macrophytes. A remarkable difference between the aquatic communities of both plateaus is that in "El Cervecero" (Buenos Aires Plateau), aquatic snails are dominant and amphipods are absent. Conversely, snails are scarce in most lakes of the Strobel Plateau and amphipods are abundant (Adami 2016).

The presence of predators can induce anti-predatory responses, altering the morphology, behaviour and spatial distribution of their prey (Simon and Townsend 2003; Ferrari et al. 2010; Lancelotti et al. 2017). These changes in prey behaviour are energy costly, and organisms face trade-offs between the benefits of a reduced predation risk and mortality and a potentially lower fitness associated with foraging or mating (Preisser et al. 2005). In the Strobel Plateau, amphipods represent the main trout prey (Bandieri 2011; Lancelotti et al. 2015). A study based on SIA in more than 12 water bodies showed that in the absence of trout, amphipods depend on both littoral and pelagic resources (Lancelotti 2009), being as much abundant in the water column as in littoral zones of fishless lakes, while in stocked lakes, they are restricted to littoral and benthic areas (Lancelotti, unpublished data).

A natural experiment with SIA comparing fishless and stocked lakes of the Strobel Plateau elucidated ecological questions about non-native fish introductions. These findings evidenced that the presence of *O. mykiss* has a marked influence on the food web structure and nutrient cycles of the analysed lakes (Porcel 2020), as shown in Fig. 14.4. Fish modify the trophic structure of the aquatic community of stocked lakes in this plateau, as demonstrated by a clear shift in the total isotopic niche size of the stocked lakes, towards more positive values of $\delta^{13}C$ and $\delta^{15}N$ (Fig. 14.4).

These differences in isotopic values between fishless and stocked lakes were explained by the presence of trout but also by multiple factors related to natural variations among the analysed environments such as changes in water levels, drainage patterns and substrate composition (Vanni 1996; Schindler and Scheuerell 2002; Feniova et al. 2016). The increase in basal $\delta^{15}N$ values together with the

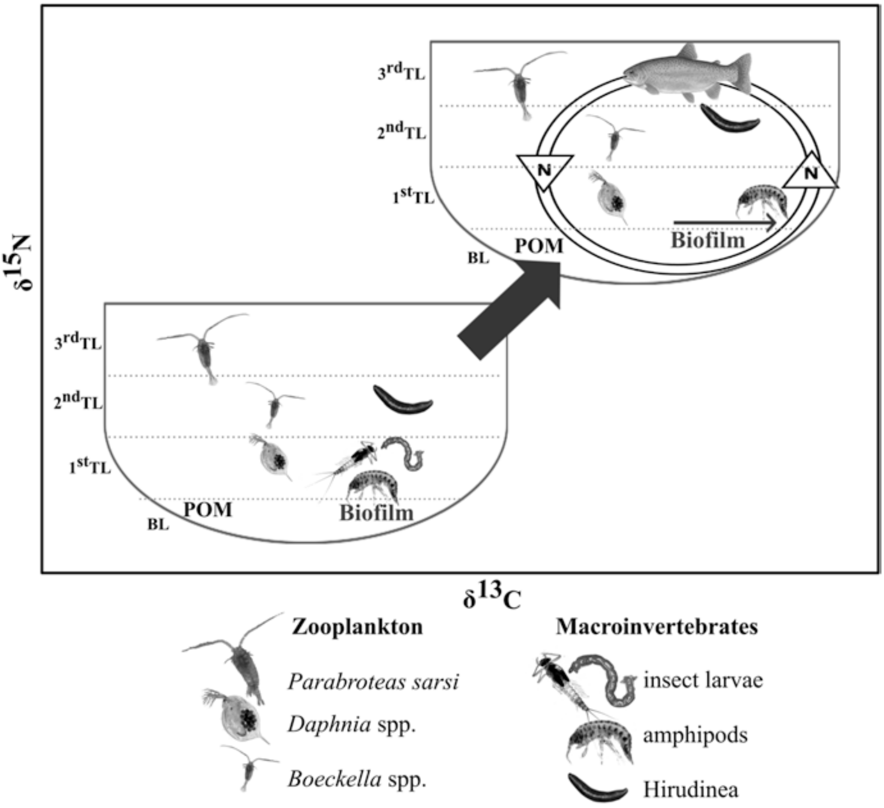

**Fig. 14.4** Simplified diagram of aquatic food webs in stocked and fishless organic lakes, based on a "natural experiment" carried out in the Strobel Plateau. The y-x axes show the $\delta^{15}N$ and $\delta^{13}C$ isotope values of the most abundant groups. $\delta^{15}N$ values reflect a consumer's trophic level (Post 2002). $\delta^{13}C$ values show contrasting carbon isotope signals depending on the habitat: organisms inhabiting littoral and benthic zones tend to have more positive $\delta^{13}C$ compared to those that inhabit pelagic zones (France 1995b). The aquatic community is represented by three trophic levels (TL) composed of the main taxonomic groups that inhabit these environments and the basal levels (BL): biofilm and particulate organic matter (POM), corresponding to the pelagic and littoral/benthic pathways. The big arrow indicates a clear shift in values of $\delta^{15}N$ and $\delta^{13}C$ between food webs with and without *O. mykiss*. The thin arrow shows the shift in the spatial distribution of amphipods towards littoral zone of the stocked lakes as an anti-predatory response. The circular arrow represents the increase in organic matter resuspension and nutrient recycling by fish presence

higher concentrations of dissolved organic carbon and higher conductivity registered in the stocked lakes was related to the presence of fish. Fish introductions increase organic matter resuspension and nutrient recycling. The microbiological decomposition of organic matter results in the initial release of isotopically light nitrogen like ammonia, and the residual organic matter becomes isotopically heavier (Macko and Estep 1985). Decomposition outcomes are observed in an accumulation of isotopically heavy organic nitrogen at the deeper depths, which can then be incorporated by the periphyton, grazing invertebrates and finally their predators such as fish (Estep and Vigg 1985). Additionally, the use of community metrics evidenced that the organisms belonging to lakes with *O. mykiss* had a more uniform species distribution, indicating greater trophic redundancy (Porcel 2020). This finding was related to the fact that fish and the big carnivorous copepod (*P. sarsi*) occupied a similar trophic role, resulting in an unaltered trophic dimension, but with a higher number of components. Other studies, also based on community metrics, reported that the introduction of a new predator caused a reduction of prey species, thus deriving on a homogenisation of energy flow pathways to top predators in tidal creeks in the Bahamas (Layman et al. 2007). Similarly, Vidal et al. (2020) suggested that fish feeding in the pelagic zone in Tasmanian lakes caused the reduction of trophic diversity of zooplankton.

Porcel (2020) also focused on individual- and population-level impacts by fish introductions in stocked lakes of the Strobel Plateau. The findings showed that *O. mykiss* generate changes in the spatial distribution and/or diet of amphipods (main prey of trout), inducing the use of alternative carbon sources (Fig. 14.4). In stocked lakes, amphipods showed higher dependency of benthic/littoral resources, which could respond to anti-predatory responses to fish presence, using macrophyte beds or rocks as refuges against predators (Porcel et al., "unpublished data"). Similar changes in the spatial distribution of prey following the introduction of fish were observed in other aquatic systems (Gliwicz and Rowan 1984; Fraser and Huntingford 1986; Simon and Townsend 2003; Stuart-Smith et al. 2008; Martínez-Sanz et al. 2010; McHugh et al. 2012). For example, the introduction of *Salvelinus fontinalis* in high-altitude European lakes altered the distributional pattern of the copepod *Cyclops abyssorumtatricus*, shifting to a near-bottom maximum, which is not typically seen in fishless lakes. The ability of *C. abyssorumtatricus* to alter its depth distribution has allowed its persistence in stocked lakes, from which other species have disappeared (Gliwicz and Rowan 1984).

## 7 Joint Influence of Fish Introduction and Climate Change

The lakes located in the Patagonian plateaus exhibit a very dynamic hydrology associated with changes in precipitation and snowfall, which account for changes in their limnological features (e.g. macrophyte cover and turbidity; Lancelotti 2009; Lancelotti et al. 2010a). In general, lakes are very sensitive to the changes in water level, thus constituting excellent sentinels of climate change (e.g. Adrian et al. (2009), and Schindler (2009)). Particularly, some shallow lakes of Argentina are good examples of the switch in lake regime associated with documented variations in water levels (e.g. O'Farrell et al. (2011), and Sánchez et al. (2014)).

In Patagonia, some climatological studies have projected a decrease in precipitation and an increase in temperature (Barros et al. 2014). This may trigger a reduction in the discharge of rivers and a water level decrease in lentic water bodies (Lancelotti et al. 2020). In particular, for the Strobel Plateau, the study by Izaguirre et al. (2018) reported a marked reduction in the water level of many arheic shallow lakes, some of which dried up completely in 2016. This decrease in the hydrometric level produced an increase in conductivity, and several lakes changed their state, from a clear vegetated to a turbid condition, with higher chlorophyll a, and declining biomass of submerged macrophytes.

A strong decrease in most arheic basins in many lakes of the Patagonian plateaus was observed during the field studies from 2006 to 2016, also confirmed by the study of time series of climate data and satellite images of the Strobel Plateau (Lancelotti et al. 2020). For the lakes of the Buenos Aires Plateau, the water level decrease also triggered the shift from clear vegetated to turbid states (Porcel et al. 2019).

The joint effect of fish introduction and water level decrease was first explored for the lakes of the Patagonian plateaus by Izaguirre et al. (2018). This study included data obtained in 31 lakes of the Strobel and Buenos Aires Plateaus, which were surveyed from 2007 to 2016. Fishless and fish stocked lakes were compared under contrasting hydrological conditions (relatively stable water levels and decreasing water levels). The reduction in precipitation, with the concomitant effect in the water level, was established as a key driver shaping both macrophytes and plankton communities. On the other hand, it was also reported that fishless lakes presented higher zooplankton and macrozooplankton biomass and lower phytoplankton biomass, in coincidence with the other studies already mentioned. Since both fish introduction and water level decrease may provoke an increase in phytoplankton biomass and turbidity, it was hypothesised that under a scenario of climate change, the reduction in precipitation may enhance the effects of fish introduction, as illustrated in Fig. 14.5.

As it was previously mentioned, fish stocked lakes develop higher biomass of autotrophic pico-, nano- and microplankton than fishless lakes, and this is emphasised during periods of low hydrometric levels (Saad et al. 2019). In the lakes of the Buenos Aires Plateau, Porcel et al. (2019) also observed a gradual increase in conductivity, turbidity, nutrients and DOC associated with the drought process that occurred from 2015 to 2017, also affecting the picoplankton structure.

## 8 Potential Effects of Trout and Climate on Hooded Grebe

Climate change and fish introduction in lakes naturally lacking them seem to exert a synergic effect on the limnological conditions of lakes, which constitute the prime habitats for Patagonian waterbird endemic species, including some threatened and near-threatened species (see Chap. 15). The concomitant effects of climate fluctuations and trout could have drastic consequences on the hooded grebe survival. The dry periods reduce the habitat availability (number of lakes) and quality (i.e. macrophyte cover and robustness) (Izaguirre et al. 2018; Lancelotti et al. 2020). These effects could be maximised due to alterations of food webs and nutrient recycling caused by trout, which could result in trophic cascades and habitat loss for hooded

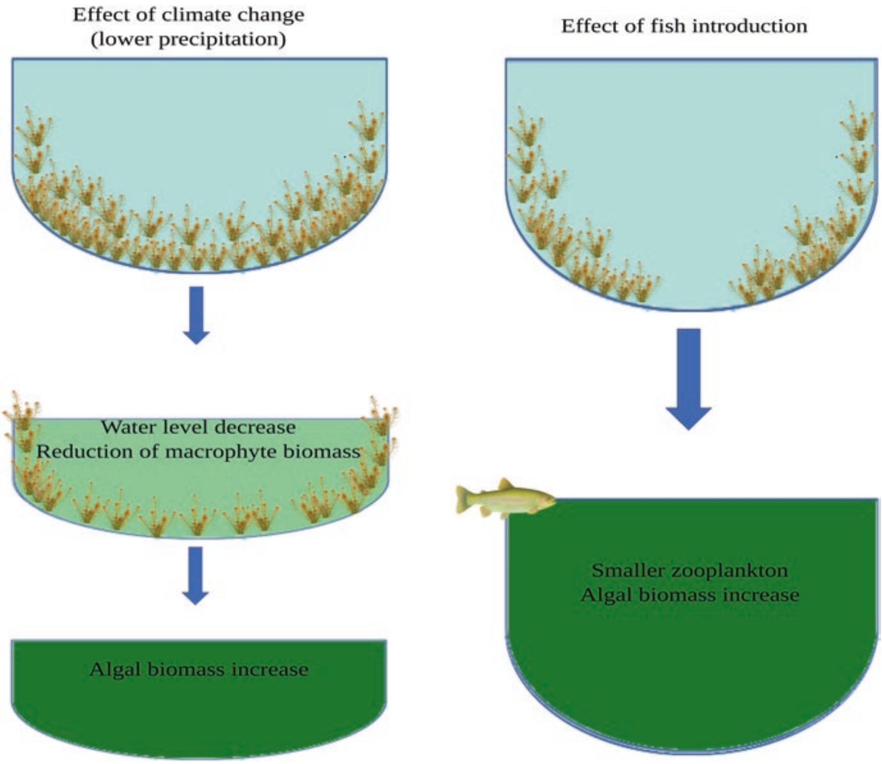

**Fig. 14.5** Diagram illustrating the joint effect of climate change (manifested by the water level decrease) and fish introduction on lake regime of water bodies from basaltic plateaus. The increase in the hue of the green colour indicates a higher biomass of phytoplankton

grebe. Additionally, the reduction of suitable habitat for hooded grebe during dry periods increases habitat overlap with trout. These two species are generalist predators that potentially depend on the same prey spectrum (Porcel 2020). Therefore, the co-occurrence of both species could originate competitive interactions, resulting particularly negative during the breeding season of hooded grebe. Trout can lead to the reduction of zooplankton size and abundance and also the relocation of amphipods (one of the main prey species for trout and grebe) from open waters to benthic/littoral zones. All these alterations could have substantial negative effects, from a nutritional and bioenergetic point of view, on adults, fledgling and chicks of hooded grebe. Besides, there are records of the desertion of active colonies due to an insufficient food supply in lakes (Fjeldså 1986).

The dry conditions observed during the last 10 years produced a significant reduction of lake availability, including most of the reproductive habitats for hooded grebe, also increasing the spatial overlap between trout and grebe's habitats (Lancelotti et al. 2020). Dry conditions were also proposed as the potential cause of macrophyte degradation, with the consequent failure of the reproduction of hooded grebe during the last three breeding seasons (Roesler's "personal communication"). Besides, the introduction of trout has been listed among the primary threats for hooded grebe (Roesler 2016).

## 9 Conclusion

In this chapter, we provide an overview and a synthesis of the current information about the ecological impacts associated with *O. mykiss* introductions in lakes of the arid southern Patagonia. The information herein reported is expected to contribute to understand the functioning of these ecosystems and, above all, to generate management tools based on biological and ecological data. We gathered fundamental data regarding the characterisation of these aquatic environments, also showing key information to understand the local and regional processes and mechanisms that structure communities, from primary producers and bacteria to top predators. Furthermore, food web studies specifically showed the effects of trout introduction on nutrient and energy pathways and their potential effects on these environments.

Preserving the water bodies of the basaltic plateaus in their natural condition is crucial for the conservation of biodiversity. Particularly, changes in the plankton and macroinvertebrates communities due to the presence of fish in lakes of the Strobel Plateau could, in turn, influence other predatory organisms, such as the aquatic birds. Under a climate change scenario, the adverse effects of the introduction of fish on the structure of aquatic communities could be magnified. For this reason, in recent years, restoration measures have been proposed and even applied in lakes stocked with *O. mykiss*, such as in the "Islote" Lake, where trout removal using gill nets was monitored by limnological studies to evaluate the "recovery" capacity of the lake. While this represents a significant first step, a permanent trout eradication initiative is needed in several lakes of the Strobel Lake Plateau to transform the present aquaculture into a more sustainable scheme in terms of habitat conservation.

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# Chapter 15
# Amphibians and Waterbirds as Bridges to Conserve Aquatic, Wetland and Terrestrial Habitats in Patagonia

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## 1 Introduction

Management strategies that consider multiple realms (i.e. marine, terrestrial or freshwater) have potential co-benefits for biodiversity conservation (Hazlitt et al. 2010; Klein et al. 2013). However, conservation planning and management strategies have been historically focused on single realms. Governmental and non-governmental environmental organisations (NGOs) often are lacking an interinstitutional working approach as well as a lack of generalised cross-realm conservation planning due to governance and technical capability issues (Adams et al. 2014). For example, in the Argentinean Patagonian provinces, land and freshwater resource management usually depends upon different environmental agencies with no clear interaction policies among them (see Chap. 9). These technical barriers owe to a poor understanding of ecological relationships. In a world wherein conservation resources are often limited, it is crucial to wisely allocate management efforts to increase conservation achievements (Carwardine et al. 2008). A better understanding of the ecological linkages and benefits of different actions across realms is crucial for a better and cost-effective resource allocation.

Freshwater and terrestrial ecosystems are linked by several ecological processes and are affected by similar threats. Biotic processes such as complex predator-prey mechanisms include interactions among species adapted to live both on land and in

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water during their life cycles. Within the vertebrates, amphibians and waterbirds have a crucial relevance within ecosystems as indicators of threats that can come from water and/or land (Duellman and Trueb 1994; Amat and Green 2010; Green and Elmberg 2014).

Amphibians are currently among the most endangered animals worldwide. With at least 37 extinct species and almost 34% of species under a threat category, amphibians are a high priority in terms of species conservation planning. On the other hand, waterbirds' dependence on freshwater habitats makes them especially vulnerable to several threats affecting aquatic, wetland and terrestrial resources globally (Young et al. 2001). In addition, the congregatory behaviour of several waterbird species increases population risks by concentrating individuals in limited areas (Ma et al. 2010). The conservation of these taxa can lead to finding cost-effectiveness management actions that integrate the protection across freshwater and terrestrial realms; thus, they can act as flagship species for sound conservation strategies (Roesler 2016; Velasco 2018).

Patagonia is home to amphibians and waterbirds highly adapted to live in seasonally extreme weather (Cei 1980; Vuilleumier 1991). Within amphibians, severe weather conditions are the cause of a low richness of species. Also, there is a high percentage of endangered species inhabiting Patagonia compared to other regions (Úbeda and Grigera 2007; Vaira et al. 2012). Regarding waterbirds, a higher number of species can be found in this region, with a lower percentage of endangered species (MAyDS and AA 2017). Nevertheless, some of the threatened waterbirds are considered among the most endangered birds worldwide (e.g. *Podiceps gallardoi* – hooded grebe; see Roesler (2016)). Most of these species also show population declines due to human-related threats and the changing environmental conditions associated with global climate change (Lancelotti et al. 2020).

In this chapter, we describe basic information about diversity, ecology and conservation traits related to some of Patagonian amphibians and waterbirds, focusing on how they represent bridges between freshwater and terrestrial environments. We also show the importance of these species in supporting ecological processes that link freshwater and terrestrial ecosystems and how conservation planning should integrate both realms to conserve threatened species. This chapter is focused on the Argentine Patagonian sector, but we consider that the conclusions can be helpful to further conservation strategies for the entire Patagonian region.

## 2 Amphibians From Patagonian Freshwaters: History, Diversity and Ecology

South America has the largest diversity of amphibians worldwide (Young et al. 2004). Of the 40 Neotropical countries, Argentina is among the top 10 with the largest amphibian fauna in the region (Vaira et al. 2017), up to 30% of which are endemic (Bolaños et al. 2008; Lavilla and Heatwole 2010; Vaira et al. 2017). Despite the low amphibian diversity of Argentinean Patagonia, the area is important in terms of conservation because of the high level of endemism and endangered amphibians (Úbeda and Grigera 2007) (Table 15.1 and Fig. 15.1).

The fossil record of South American anurans is biased, having a poor representation of most neobatracians. Some exceptions exist within the clades Calyptocephalellidae and Pipidae, which have extensive records after the late Cretaceous (Rolando et al. 2019). Some of the oldest records of anurans from South America are found in Patagonian provinces (e.g. *Vieraella herbstii*, from the lower Jurassic of northern Santa Cruz, and *Notobatrachus degiustoi*, from the middle Jurassic of northeast Santa Cruz (Cei 1980; Úbeda 1998)). Other examples such as *Shelania pascuali* in Chubut and *Calyptocephalella gayi* in Neuquén highlight Patagonia's relevance for the history of amphibians (Cei 1980). Other relevant records are *Avitabatrachus uliana* in Neuquén province (Baéz et al. 2000, 2022); *Calyptocephalella pichileufensis* (Gómez et al. 2011), *Calyptocephalella satan* (Agnolin 2012) and *Llankibatrachus truebae* (Baez et al. 2003) in Río Negro province; and *Calyptocephalella sabrosa* (Muzzopappa et al. 2021) in Chubut province.

Three hypotheses were proposed to explain the current distribution pattern of anurans in southern South America: (i) the impoverished hypothesis (Darlington 1965), which suggests that the Patagonian forest amphibians constitute a poorly differentiated group from extant clades; (ii) the ancient assembly hypothesis (Cei 1962), which states that the *Nothofagus* forest anurans are a remnant of the tertiary amphibians isolated by ecological barriers and (iii) the complex history hypothesis (Vuilleumier 1968), which indicates that the current amphibian assembly is the result from a sum of the evolutionary histories of the different groups that compose it. However, a recent different hypothesis considers that the current diversity and distribution of Patagonian amphibians can be explained by two main anuran components. The first group would be composed by survivors of widely distributed Gondwanan taxa (e.g. *Calyptocephalella and Telmatobius*) while the second composed by *Rhinella* and *Pleurodema*. This last genus probably got into Patagonia during Plio-Pleistocene times when Chacoan environments expanded southwards (Agnolin 2012). Regarding to the genus *Atelognathus*, some researchers assume a recent expansion in association with a favourable period, from a single source population restricted to the southernmost part of its distributional range (Barraso and Basso 2018).

Summarising, the current Patagonian amphibian fauna is considered both relictual and with a high level of endemism and microendemism (Úbeda and Grigera 2007). The two most diverse genera are *Eupsophus*, with species restricted to forested areas in southern latitudes, and *Alsodes*, with species also distributing along the arid Andean slopes of Central Chile and Argentina (Blotto et al. 2013).

The highest richness of amphibian species is found in the northern region of Patagonia, with a decreasing number towards the south. In Argentina, the lowest richness is found in Santa Cruz province, with only four known species, while this group is absent from insular Tierra del Fuego (Vaira et al. 2012). On the other hand, the amphibian diversity declines from the west to east, associated with the drastic decrease in environmental complexity (Perotti et al. 2005). Amphibians from Patagonia are distributed in two main habitats, the Valdivian and southern forests

**Table 15.1** Amphibian species from the Argentinean Patagonia, conservation status and population trends listed in the IUCN (International Union for Conservation of Nature) Red List and Argentinean Red List (Source: Vaira et al. 2012)

| | Population trend | |
|---|---|---|
| Families and species | IUCN category | Argentinean category |
| **Alsodidae** | | |
| *Alsodes gargola* | Least concern | **Vulnerable** |
| *Alsodes neuquensis* | **Endangered – decreasing** | **Endangered** |
| *Alsodes pehuenche* | **Critically endangered – decreasing** | **Critically endangered** |
| *Alsodes verrucosus*[a] | **Endangered – decreasing** | Data deficient |
| *Eupsophus calcaratus* | Least concern – **decreasing** | Least concern |
| *Eupsophus emiliopugini* | Least concern – **decreasing** | **Vulnerable** |
| *Eupsophus roseus* | Least concern – **decreasing** | Not assessed |
| *Eupsophus vertebralis* | Least concern – **decreasing** | Data deficient |
| **Batrachylidae** | | |
| *Atelognathus nitoi* | **Vulnerable** | **Vulnerable** |
| *Atelognathus patagonicus* | **Critically endangered – decreasing** | **Critically endangered** |
| *Atelognathus praebasalticus* | **Endangered – decreasing** | Data deficient |
| *Atelognathus reverberii* | **Vulnerable – decreasing** | **Vulnerable** |
| *Atelognathus solitarius* | Data deficient | Data deficient |
| *Batrachyla antartandica* | Least concern | **Vulnerable** |
| *Batrachyla fitzroya* | **Vulnerable** | **Vulnerable** |
| *Batrachyla leptopus* | Least concern | Least concern |
| *Batrachyla taeniata* | **Least concern – decreasing** | Least concern |
| *Chaltenobatrachus grandisonae* | Least concern | Data deficient |
| *Hylorina sylvatica* | **Least concern – decreasing** | **Vulnerable** |
| **Bufonidae** | | |
| *Nannophryne variegata* | **Least concern – decreasing** | Least concern |
| *Rhinella arenarum* | Least concern | Least concern |
| *Rhinella papillosa*[b] | Not assessed | Least concern |
| *Rhinella rubropunctata* | **Vulnerable – decreasing** | **Vulnerable** |
| **Leptodactylidae** | | |
| *Leptodactylus luctator* | Least concern | Least concern |
| *Leptodactylus mystacinus* | Least concern | Least concern |
| *Pleurodema bufoninum* | Least concern | Least concern |
| *Pleurodema nebulosum* | Least concern | Least concern |
| *Pleurodema somuncurense* | **Critically endangered – decreasing** | **Critically endangered** |
| *Pleurodema thaul* | Least concern | Least concern |
| **Odontophrynidae** | | |
| *Odontophrynus americanus* | Least concern | Least concern |
| *Odontophrynus occidentalis* | Least concern | Least concern |
| **Rhinodermatidae** | | |
| *Rhinoderma darwinii* | **Endangered – decreasing** | **Endangered** |

[a] Uncertain presence in Argentina (Blotto et al. 2013)

[b] Could be a synonym of *Rhinella spinulosa* (Vera Candioti et al. 2020)

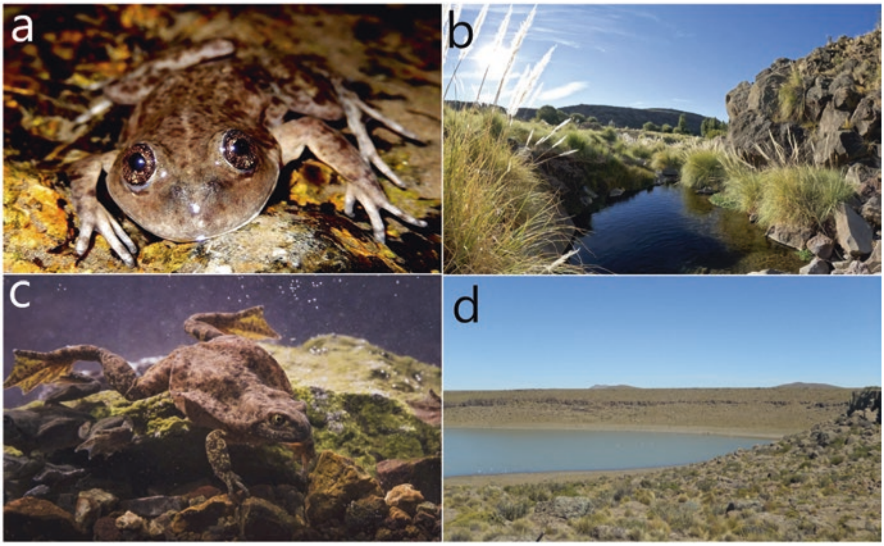

**Fig. 15.1** Examples of emblematic threatened Patagonian amphibians in their freshwater and wetland habitats. (**a**) The El Rincon stream frog, *Pleurodema somuncurense.* (Photo by Federico Kacoliris) in (**b**) the thermal waters of the Valcheta Stream. (Photo by Melina Velasco, upper right). (**c**) The Patagonia frog, *Atelognathus patagonicus.* (Photo by Rodrigo Calvo, down left) in (**d**) the temporary steppe lagoons. (Photo by Federico Kacoliris, down right)

and the steppe and mountain wetlands (Cei 1980; Perotti et al. 2005). The four amphibians with the southernmost distribution at global level (*Batrachyla antartandica*, *Chaltenobatrachus grandisonae*, *Nannophryne variegata* and *Pleurodema bufoninum*) are found in both Argentinian and Chilean Patagonia (Atalah and Sielfeld 1976; Úbeda et al. 2010; Ortiz 2015; Cisternas-Medina et al. 2019). The Argentinean Patagonia hosts 32 amphibian species (Table 15.1) belonging to 6 families, within which Batrachylidae and Alsodidae have the greatest richness. Approximately a quarter of these is endemic, including *Batrachyla fitzroya* in Chubut province; *Alsodes neuquensis*, *Atelognathus patagonicus* and *Atelognathus praebasalticus* in Neuquén province; and *Atelognathus solitarius* and *Pleurodema somuncurense* in Río Negro province. In turn, *Atelognathus reverberii* is endemic from the Somuncura Plateau (see Chap. 9) shared by Rio Negro and Chubut provinces.

There are recent studies that propose new taxonomical arrangements for some Patagonian amphibians; thus, there is some controversy between information exposed in the IUCN Red List, and scientific literature published later than IUCN assessments were made, thus reducing the number of enlisted species. This is the case of *Atelognathus nitoi*, which has recently been considered as a senior synonym of *A. ceii* and *A. salai* (Barrasso and Basso 2018), and the Argentinean populations of *Alsodes australis*, now considered as *A. gargola* (Blotto et al. 2013). The record of *Alsodes verrucosus*, from Argentina, is based on two populations (Cei 1987) of dubious identity, and hence its presence should be considered as still uncertain (Blotto et al. 2013).

Among the amphibians living in Patagonia, 44% occur in freshwater and wetlands (e.g. streams, lakes and ponds) or humid soil within forest areas, while only 18% live in streams or ponds located in tablelands or plateaus dominated by steppe vegetation.

Amphibians show a particular life cycle that depends upon the existence of freshwaters. This feature could be considered a drawback for Patagonian amphibians, since these habitat types occupy merely 5% of the region (Perotti et al. 2005). However, Patagonian species are highly adapted to these extreme conditions. The Patagonian freshwater ecosystems are mainly represented by lakes, ponds, rivers, streams, "mallines" and several kinds of ponds (see Chaps. 3, 9 and 10). Lakes are more common near the Andean Mountain range and are surrounded by forest (see Chap. 3). Some species prefer these forestry areas, where they find refuge under tree bark, trunks, dry branches or inside caves and pits (Ghirardi and López 2017). In turn, wetlands such as ponds and *mallines* are common in arid areas such as tablelands or plateaus and show several amphibian species adapted to live under contrasting seasonal conditions including dry periods. In particular, mallines are an essential habitat for native amphibian species (Perotti et al. 2005).

Some Patagonian amphibian species such as *Rhinoderma darwinii*, *Batrachyla antartandica* and *Hylorina sylvatica* have a distribution range restricted to the Valdivian refuges (Úbeda and Grigera 2007). Other species like *Rhinella rubropunctata*, *Nannophryne variegata*, *Pleurodema thaul*, *Alsodes gargola*, *A. monticola*, *A. verrucosus*, *B. leptopus*, *B. taeniata*, *Eupsophus roseus*, *E. calcaratus and E. vertebralis* just occur in high-altitude ponds of Argentina and Chile (Perotti et al. 2005). The presence of *Atelognathus nitoi* was considered as a strict microendemism for the Laguna Verde in the Nahuel Huapi National Park, Río Negro, Argentina. At present, this species is also found in Chile, since populations of *A. ceii* and *A. salai* now belong to this specific taxon (Barrasso and Basso 2018; Alveal and Díaz-Páez 2021).

*Atelognathus reberverii* is endemic from temporary and/or permanent clay ponds located in the Somuncura Plateau, provinces of Río Negro and Chubut. This species, as most amphibians, depends on water for reproduction and larval development. However, it has adaptations to spend extensive periods in terrestrial habitats when ponds are dry. Even when ponds have water, it is common to find individuals under rocks at long distances from the pond boundary. *Atelognathus patagonicus* is another micro-endemic species that only occurs at endorheic ponds in Neuquén province. This species exhibits two morphotypes, aquatic and terrestrial, adapted to significant seasonal changes when ponds become dry. The aquatic form, with loose skin and interdigital membranes on the hind legs, shows a yellow-orange coloration on the belly and lower thighs and stays all the time within ponds while they have water. When ponds are dry, individuals exhibit some external changes (e.g. loss of skin and reduction of interdigital membrane) in order to reduce water loss. This so-called terrestrial morphotype or littoral form is more resistant to dry conditions and usually lives under rocks due to the moisture of this microhabitat (Cei 1980).

*Pleurodema somuncurense* is a case of a micro-endemic species only occurring in a small stream (Valcheta Stream) located in a dry ecotonal physiognomy between forest and steppe in Rio Negro province (León et al. 1998). This stream is fed by hot springs located in the headwaters and thus shows warm temperatures all year round. *P. somuncurense* evolved to live in these constantly warm waters, becoming an almost fully aquatic species with adaptations such as interdigital membranes on the hind legs and some loose skin that improves oxygen exchange. This frog is active all year-round and lays eggs on the slow-flowing stream banks.

*Alsodes neuquensis* is a semiaquatic species, endemic from Neuquén province. This frog occurs and breeds in small mountain ponds and streams located in volcanic plateaus. These habitats are surrounded by open forests of *Araucaria araucana* and *Nothofagus antarctica*. This species has a long larval development of one year or even longer which is likely an adaptation to face the extreme weather of Patagonia (Cei 1976).

Some species, such as *Eupsophus calcaratus* and *Rhinoderma darwinii*, live in wet habitats but not necessarily close to freshwater ecosystems. The former is found in temperate forests of Chile and Argentina, under fallen logs and rocks, in dark, cold and wet sites with woody vegetation, generally near but not in streams, ponds and rivers. Its reproductive cycle (i.e. egg-laying and larval cycle) is developed in water-filled cavities in the soil, under stones and logs. For this reason, individuals require very humid or water-saturated soils (Úbeda 2000). *R. darwinii* is an almost fully terrestrial species that lives in temperate forests of Argentina and Chile. As *E. calcaratus*, *R darwinii* does not require freshwater habitats to complete its development. However, unlike the former, females of *R. darwinii* lay the eggs in small shelters located on the humid soil, among the litter, in the same site where males vocalise (Busse 1970). After two weeks, when the muscular movement of the embryos begins, males incorporate them into their vocal sacs, where the larval cycle completes. Subsequently, juveniles are expelled to the terrestrial environment (Busse 1970; Cei 1980).

One of the most widespread species, *Pleurodema bufoninum*, can be found in several habitat types because of its adaptation to face low temperatures and droughts by spending long periods underground or under rocks in the bushy steppe, usually far away from the water.

The remaining species are not habitat-specific and can occur in forests, *mallines*, steppes, grasslands and shrubs, mostly near freshwaters (Cei 1980).

## 3 Waterbirds From Patagonian Freshwaters: History, Diversity and Ecology

Waterbirds' diversity from Patagonian freshwaters includes at least 76 species of 15 families (Table 15.2 and Fig. 15.2). Research on waterbird communities in Patagonia has mainly concerned habitat diversity, population structure and the importance of

**Table 15.2** Waterbirds' species of the Argentinean Patagonia, conservation status and population trends listed in the IUCN Red List and the Argentinean Red List (MAyDS and AA 2017)

| | Population trend | |
|---|---|---|
| Families and species | IUCN category | Argentinean category |
| **Anatidae** | | |
| *Dendrocygna viduata* | Least concern | Least concern |
| *Cygnus melanocoryphus* | Least concern | Least concern |
| *Coscoroba* | Least concern | Least concern |
| *Chloephaga picta* | **Least concern – decreasing** | **Vulnerable** |
| *Chloephaga poliocephala* | **Least concern – decreasing** | **Vulnerable** |
| *Chloephaga rubidiceps* | **Least concern – decreasing** | **Critically endangered** |
| *Merganetta armata* | **Least concern – decreasing** | Least concern |
| *Tachyeres patachonicus* | **Least concern – decreasing** | Least concern |
| *Lophonetta specularioides* | Least concern | Least concern |
| *Speculanas specularis* | Least concern | Least concern |
| *Spatula versicolor* | Least concern | Least concern |
| *Spatula platalea* | Least concern | Least concern |
| *Spatula discors* | **Least concern – decreasing** | Least concern |
| *Spatula cyanoptera* | Least concern | Least concern |
| *Mareca sibilatrix* | Least concern | Least concern |
| *Anas bahamensis* | **Least concern – decreasing** | Least concern |
| *Anas georgica* | **Least concern – decreasing** | Least concern |
| *Anas flavirostris* | **Least concern – decreasing** | Least concern |
| *Netta peposaca* | Least concern | Least concern |
| *Heteronetta atricapilla* | Least concern | Least concern |
| *Oxyura jamaicensis* | Least concern | Least concern |
| *Oxyura vittata* | Least concern | Least concern |
| **Aramidae** | | |
| *Aramus guarauna* | Least concern | Least concern |
| **Rallidae** | | |
| *Rallus antarcticus* | **Vulnerable** | **Endangered** |
| *Pardirallus sanguinolentus* | Least concern | Least concern |
| *Porphyriops melanops* | Least concern | Least concern |
| *Porzana spiloptera* | Least concern | Least concern |
| *Gallinula galeata* | Least concern | Least concern |
| *Fulica rufifrons* | Least concern | Least concern |
| *Fulica armillata* | Least concern | Least concern |
| *Fulica leucoptera* | Least concern | Least concern |
| **Phoenicopteridae** | | |
| *Phoenicopterus chilensis* | **Near threatened** | **Near threatened** |
| **Podicipedidae** | | |
| *Rollandia rolland* | **Least concern – decreasing** | Least concern |
| *Podilymbus podiceps* | Least concern | Least concern |
| *Podiceps major* | Least concern | Least concern |

(continued)

**Table 15.2** (continued)

| | Population trend | |
|---|---|---|
| Families and species | IUCN category | Argentinean category |
| *Podiceps occipitalis* | **Least concern – decreasing** | Least concern |
| *Podiceps gallardoi* | **Critically endangered** | **Critically endangered** |
| **Charadriidae** | | |
| *Pluvialis dominica* | **Least concern – decreasing** | Least concern |
| *Charadrius semipalmatus* | Least concern | Least concern |
| *Charadrius collaris* | **Least concern – decreasing** | Least concern |
| **Recurvirostridae** | | |
| *Himantopus mexicanus* | Least concern | Least concern |
| **Pluvianellidae** | | |
| *Pluvianellus socialis* | **Near threatened** | Least concern |
| **Scolopacidae** | | |
| *Limosa haemastica* | Least concern | Least concern |
| *Calidris bairdii* | Least concern | Least concern |
| *Calidris fuscicollis* | **Least concern – decreasing** | Least concern |
| *Calidris melanotos* | Least concern | Least concern |
| *Gallinago paraguaiae* | Least concern | Least concern |
| *Gallinago stricklandii* | **Near threatened** | **Endangered** |
| *Phalaropus tricolor* | Least concern | Least concern |
| *Phalaropus fulicarius* | Least concern | Least concern |
| *Tringa flavipes* | **Least concern – decreasing** | Least concern |
| *Tringa melanoleuca* | Least concern | Least concern |
| **Rostratulidae** | | |
| *Nycticryphes semicollaris* | **Least concern – decreasing** | Least concern |
| **Laridae** | | |
| *Chroicocephalus serranus* | Least concern | Least concern |
| *Chroicocephalus maculipennis* | Least concern | Least concern |
| *Leucophaeus pipixcan* | Least concern | Least concern |
| *Chroicocephalus chirocephalus* | Least concern | Least concern |
| *Larus dominicanus* | Least concern | Least concern |
| *Gelochelidon nilotica* | **Least concern – decreasing** | Least concern |
| *Sterna hirundinacea* | Least concern | Least concern |
| *Sterna trudeau* | Least concern | Least concern |
| **Ciconiidae** | | |
| *Ciconia maguari* | Least concern | Least concern |
| *Mycteria americana* | **Least concern – decreasing** | Least concern |
| **Phalacrocoracidae** | | |
| *Phalacrocorax brasilianus* | Least concern | Least concern |
| *Phalacrocorax atriceps* | Least concern | Least concern |
| **Ardeidae** | | |
| *Ixobrychus involucris* | Least concern | Least concern |
| *Nycticorax* | **Least concern – decreasing** | Least concern |

(continued)

**Table 15.2** (continued)

| | Population trend | |
|---|---|---|
| Families and species | IUCN category | Argentinean category |
| *Butorides striata* | **Least concern – decreasing** | Least concern |
| *Bubulcus ibis* | Least concern | Least concern |
| *Ardea cocoi* | Least concern | Least concern |
| *Ardea alba* | Least concern | Least concern |
| *Syrigma sibilatrix* | Least concern | Least concern |
| *Egretta thula* | Least concern | Least concern |
| **Threskiornithidae** | | |
| *Plegadis chihi* | Least concern | Least concern |
| *Theristicus melanopis* | Least concern | Least concern |
| *Platalea ajaja* | Least concern | Least concern |

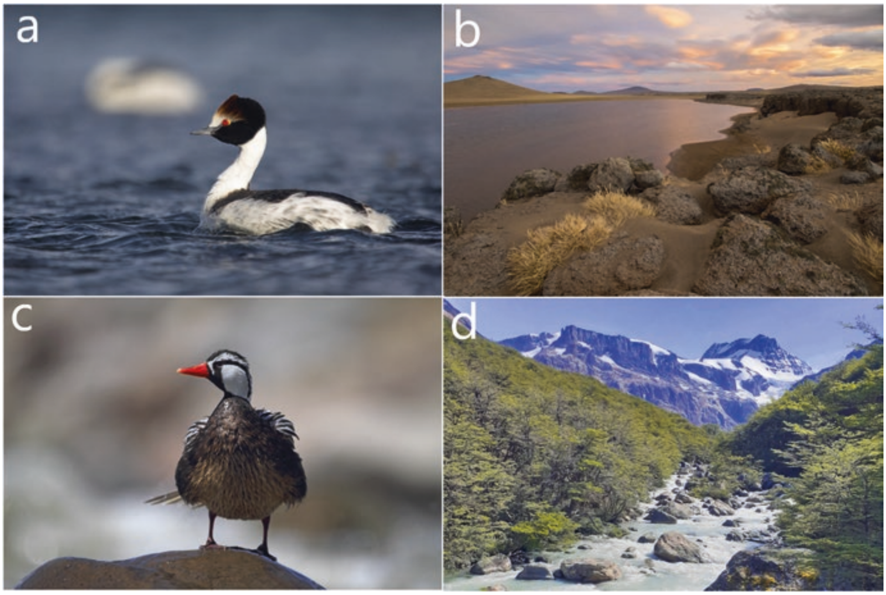

**Fig. 15.2** Examples of emblematic Patagonian threatened waterbirds in their freshwater and wetland habitats. (**a**) The hooded grebe, *Podiceps gllardoi.* (Photo by Gonzalo Pardo), in (**b**) a shalow lake of Santa Cruz plateau. (Photo by Gonzalo Ignazi, upper right). (**c**) The torrent duck *Merganetta armata.* (Photo by Hernán Povedano) (**d**) in Las Vueltas River. (Photo by Soledad Ovando)

migratory species for ecological processes in continental freshwater ecosystems, essentially in the northern and central areas of the country (Bucher and Herrera 1981; Echevarria and Chani 2000; Romano et al. 2005).

Along the Andes, most waterbodies are oligotrophic (see Chaps. 3, 4, 10 and 14) and therefore of little importance for maintaining large flocks of birds. However, certain characteristic species can be found there, such as *Podiceps major* (great grebe), *Theristicus melanopis* (black-faced ibis), *Chloephaga poliocephala* (ashy-headed goose), *Tachyeres patachonicus* (flying steamer duck) and *Anas specularis* (spectacled duck) (Scott and Carbonell 1986).

The presence and abundance of waterbirds are influenced by local environmental characteristics and the specific demands of each species (Weller 1999). In Patagonia, large lakes and rivers are prominent in the Andean region (Iglesias and Pérez 1998). In contrast, the Patagonian steppe receives less than 300 mm of rain per year, thus representing one of the aridest extensions in Argentina (Cabrera 1976; Morello et al. 2012) (see Chap. 1). Whereas permanent waterbodies are rare in this area, a system of several basaltic plateaus, dotted with natural depressions that collect water from snow and ice melt, is prominent in the region (Iriondo 1989, Chap. 9). The topography of such plateaus facilitates the development of complex shallow lake assemblages, with a relatively high environmental heterogeneity (Lancelotti et al. 2009) (see Chaps. 10, 11 and 14).

Waterbird abundance at local scales depends on habitat characteristics, food abundance and the availability of suitable sites for reproduction or resting (Wiens 1989). Other factors affecting freshwaters used by waterbirds include sex, dominance, pairing status, flocking and stage of the life cycle. Species composition is usually associated with the arrival of migratory species which are added to those already present in the area. All these parameters influence the resources needed and the birds' access to habitats where such resources are available (Canziani and Derlindati 2000; Romano et al. 2005).

Some studies on the population and ecology of migratory Nearctic shorebirds have included abundance patterns at continental sites (Laredo 1996; Montalti et al. 2003; Lanctot et al. 2004). A few species overwinter in inland lakes and several sites of importance for shorebirds have been identified (Di Giacomo 2005). In addition, waterbird surveys conducted in isolated shallow lakes, provided general information on waterbird occurrence (Fjeldså 1985, 1986; Imberti 2005).

Patagonia has had a complex geological and vegetational history in the Tertiary (Romero 1986) and the Quaternary (Pleistocene-Holocene; Heusser 1989). During the Late Pleistocene, western Patagonia was covered by the largest icecap in South America (480,000 km$^2$; Hollin and Schilling 1981), rendering a substantial area of Patagonia uninhabitable. In consequence, the fauna species must have either retreated their distribution range or suffered extinctions (Markgraf 1989). By greatly affecting the distribution of vegetation types and markedly altering the width of continental margins, glacial-interglacial cycles must have had a major impact on the avifauna (Lei et al. 2014).

Different authors (Vuilleumier 1972, 1985, 1991; Acosta et al. 2021) demonstrated that past environmental factors, probably Pleistocene in age, have indeed promoted speciation of both waterbirds and landbirds in Patagonia. Because the climate and vegetation of the Patagonian Pleistocene-Holocene period are well known (Markgraf 1989), this region offers great promise for studies in bird evolution.

Hypotheses on speciation patterns in *Phalacrocorax* and *Tachyeres* involve vicariance events associated with glaciation and deglaciation cycles over the last 100,000 years (Livezey 1986). In these scenarios, the western Patagonian icecap was the main barrier that isolated Pacific and Atlantic coast populations of ancestral taxa. Subsequent melting of that ice barrier allowed a new contact, and a later glacial advance led to another cycle of vicariance or geographic isolation.

Speciation has occurred across a broad taxonomic spectrum, from caracaras and ducks to furnariids and buntings (Fjeldså 1985). Thirty-four percent of the Patagonian bird species (73 out of 217) show some evidence of speciation caused by at least one of the following four processes: (i) range disjunctions accompanied by weak differentiation, (ii) parapatry and hybridisation, (iii) secondary range overlaps and (iv) isolation in habitat relicts (Vuilleumier 1991). Vicariance events have therefore played an important role in the evolution of the Patagonian avifauna (Vuilleumier 1991; Acosta et al. 2021). Whereas the occurrence of vicariance patterns and of overlaps highlights the role of speciation or increase in species numbers, the occurrence of relicts underscores the role of extinction or loss of diversity (Acosta et al. 2021).

Species richness and abundance of Patagonian waterbirds are fundamentally affected by the size and productivity of freshwater ecosystems. Habitat heterogeneity determines the assemblage complexity in terms of species diversity. Species richness and abundance increase with shoreline length and waterbody size. While shoreline length presents a close relationship with species number and abundance during autumn and winter, wetland area was the main variable influencing waterbird populations during spring and summer. A close relationship between species richness and area was described, wherein larger wetlands support a higher number of bird species (González-Gajardo et al. 2009). Additionally, bird abundance is related to water level fluctuation and wetland area (Froneman et al. 2001). Although bigger freshwater ecosystems can provide more microhabitats, thereby attracting a greater number of species, Garay et al. (1991) showed that smaller freshwater habitats maintained higher waterbird density and diversity than larger ones. In this context, the structural and vegetation heterogeneity shows an important relationship with bird assemblages. Shoreline length and shoreline development indices were considered as determinants of bird abundance by Hudson (1983), who suggested that in similar-sized wetlands, bird abundance will be higher in those that present a more irregular perimeter, thus offering more refugees. The waterbird communities located in the Patagonian steppe display a heterogeneous species structure and relative species abundance (González 1996; Gatto et al. 2005). Further south in Patagonia, the highest abundance of waterbirds is found in volcanic foothill plateaus, which are rich in minerals and highly productive (Fjeldså 1985). Lakes harbouring the most species and individuals were those with alkaline waters and reed beds (Fjeldså 1985).

# 4 Conservation Status and Threats Affecting Amphibians and Waterbirds in Patagonian Freshwaters

## *4.1 Amphibians*

The amphibian fauna of Patagonia is one of the most threatened in the country. According to the National Assessment of Native Amphibians, 75% and 65% of species of the Patagonian steppe and forests, respectively, are included in a threat category (Vaira et al. 2017). Neuquén and Río Negro provinces have the highest percentage of amphibian threatened species (Vaira et al. 2012). The IUCN Red List assessment includes 11 species falling within a threat category, three of them listed as critically endangered, four as endangered species and four as vulnerable species (Table 15.1). Another seven Patagonian amphibians considered as of least concern show decreasing populations while two species are listed as data deficient, and one was still not assessed (*Rhinella papillosa*).[1] These last species can likely be assessed or be moved (in the case of those showing a declining trend) into a threatened category in the near future, even increasing the percentage of Patagonian endangered amphibians.

The high percentage of endangered amphibian species is likely related to the high degree of endemism, with some species restricted to very small areas (Úbeda and Grigera 2007). The small distribution range is associated with small population sizes and a high degree of specialisation. These conditions contribute to increase susceptibility of endemic Patagonian amphibians to anthropogenic threats. According to Lavilla (2000) and Lavilla and Heatwole (2010), the main threats affecting amphibian populations in Argentina are habitat loss or fragmentation, chemical and biological contamination of freshwater ecosystems, invasive species, illegal pet trade and several factors related to climate change (e.g. increasing UV radiation, temperature and frequency of droughts). Among these threats, livestock breeding, invasive fishes, emerging diseases and climate change were recognized as the main negative factors affecting amphibian species in Patagonia (Úbeda and Grigera 2007; Vaira et al. 2012; Velasco 2018).

As for the three most endangered Patagonian amphibians, local extinctions of *A. patagonicus* and *P. somuncurense* subpopulations have been recorded. Although both species occur in different habitat types (*A. patagonicus* inhabits tableland ponds while *P. somuncurense* is found in a small endorheic stream), both are threatened by similar factors (impacts on their habitat by livestock and predation by invasive fish species) (Velasco 2018; Cuello et al. 2009) (see Chap. 14).

Sheep and cattle rising is widespread in Patagonia (Instituto Nacional de Tecnología Agropecuaria 2015). Lack of appropriate management of livestock has promoted overgrazing and consequent desertification (Mazzoni and Vazquez 2009; Nanni et al. 2020). As documented in other regions (Burton et al. 2009; Schmutzer

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[1] Although this species could be a synonym of *Rhinella spinulosa* (Vera Candioti et al. 2020)

et al. 2008), livestock also affects the Patagonian amphibian freshwater habitats by trampling, drinking, polluting with urine and faeces and grazing over hydrophytic vegetation, thus causing water eutrophication and loss of reproductive habitats. At least 8 out of the 11 Patagonian threatened amphibian species are affected by this threat. *Pleurodema somuncurense* is a good example of this type of impact. The small stream inhabited by this frog is under a high livestock pressure due to an exchange of sheep and goats for cattle, in combination with other threats like works for water management (e.g. small dams and channels) which have caused a relictual subpopulation of this species to lose a number of key locations and even promoted local extinctions (Velasco 2018). Because of livestock impacts, a stream portion was fenced off to avoid animal access, promoting a quick recovery of both hydrophilic and riparian vegetation (Arellano et al. 2017b). Subsequent translocation of frogs into this restored habitat resulted in the re-establishment of that population as recorded of reproductive events and natural recruitments, which confirmed suitable habitat conditions for frogs (Martínez Aguirre et al. 2019).

Invasive fish species are one of the main drivers of the current extinction of amphibians (Collins 2010). In Patagonia, invasive fish species represent 43% of the total freshwater fish species (Macchi and Vigliano 2014) (for details, see Chaps. 13 and 14). Most of these are salmonids that have caused negative changes in native biota due to predation, competition and changes in trophic webs (Ortubay et al. 2006; Cussac et al. 2012) and are among the main causes of extirpation and decline of many native freshwater species (Buria et al. 2007, 2009). Despite this negative impact, few studies in Argentina have addressed the effect of this threat on amphibians. One of such studies was carried out in the Valcheta Stream, Somuncurá Plateau, by Velasco et al. (2018). A negative effect of invasive rainbow trout (*Oncorhynchus mykiss*) on *Rhinella arenarum* and *P. somuncurense* was observed, with a decrease in occupancy recorded for both species.

Two emerging infectious diseases (ranavirus and chytridiomycosis), caused by microparasites *Iridovirus* and *Batrachochytrium dendrobatidis* (chytrid fungus), were associated with amphibian decline in almost all continents (Collins 2010). Amphibian ranaviruses were reported in at least 105 species of amphibians in 25 countries. Gross signs of ranavirus infection are not always apparent; they can be confused with other factors, and mortality events are not easily observed due to their rapid progression and the fast decomposition of dead hosts (Duffus et al. 2015). In Argentina, ranavirus was only described in the Patagonian *Atelognathus patagonicus* (Fox et al. 2005). *B. dendrobatidis* was already reported for eight Patagonian amphibians, four of which are within a threatened category (*A. pehuenche*, *A. patagonicus*, *A. reverberii* and *P. somuncurense*) (Arellano et al. 2015, 2017a; Ghirardi et al. 2014). Although this pathogen can be very harmful to many species, there is no evidence that it is causing a population decline, except for an *A. patagonicus* mortality event recorded in Laguna Blanca (Ghirardi et al. 2014).

Climate change is also a main driver of the global amphibian decline (Blaustein and Wake 1990; Stuart et al. 2004; Schivo et al. 2019). Environmental temperature and moisture patterns can influence amphibian ecology, physiology and behaviour because they must maintain the moist skin for oxygen and ionic exchange (Lips et al. 2005). In Argentina, climate projections for the next two or three decades show a general warming of 0.5–1.0 °C all over the country and reductions of precipitation in the dry area comprising the central-western region and Patagonia (Barros et al. 2015). Studies conducted in the lakes of the Austral Patagonia plateaus evidenced a water level decrease as a consequence of reduced precipitation, affecting macrophyte and plankton communities (Izaguirre et al. 2018) (see Chap. 5). Unfortunately, at present, studies assessing the effect of climate change on Patagonian amphibians are lacking. However, evidence demonstrates negative impacts, as in the case of *A. patagonicus*, where a severe and uncommon drought caused a decline of almost 90% of the extant population (Cuello et al. 2009; IUCN 2019b).

## *4.2 Waterbirds*

Even though most waterbirds are categorised as least concern by the IUCN, at least 20 species (33%) inhabiting Patagonia are experiencing a global decline (Table 15.2). Besides, five species are under a threat category: *P. gallardoi*, *R. antacticus*, *P. socialis*, *G. stricklandii* and *P. chilensis* (Table 15.2).

The waterbirds from Patagonia are exposed to a combination of threats which includes logging and livestock grazing and trampling in their watersheds, introduction of alien species, volcanic eruptions in breeding areas, agrochemical pollution, oil exploitation, light pollution, mining and hydroelectric dams (del Hoyo et al. 1992; O'Donnell and Fjeldså 1997; Imberti and Casañas 2010; Roesler et al. 2014; Roesler et al. 2016; Fasola and Roesler 2018). Increasing projects of wind farms represent a potential new threat for birds and other biota like bats (Berkunsky pers.obs.).

The destruction and degradation of Patagonian natural grassland by livestock grazing and trampling as well as by other introduced herbivores have negatively impacted the mainland breeding habitat of many bird species such as the *P. socialis* and species of genus *Chloephaga* (sheldgeese) (Fjeldså 1986; Dinerstein et al. 1995; Cossa et al. 2018). Livestock is responsible for at least 14% of sheldgeese nest losses and disturbances of breeding pairs (Cossa et al. 2018). Trampling of nests and chicks by grazing animals is a potential threat to *Pluvianellus socialis* (Magellanic Plovers) (Ferrari et al. 2003). Moreover, intensive grazing of the steppes also affects wetlands because bare soils are dragged by wind and deposited in ponds and mallines (Fjeldså 1986). In addition, waterbirds have been affected in some places wherein hydrophytic vegetation has been grazed and trampled by cattle (Fjeldså 1986).

Rainbow trout shape the community of aquatic invertebrates, affecting species' dominance and size structure. These invertebrates represent the primary prey item of many waterbird species, including endangered species such as the hooded grebe (see Chaps. 11 and 14). In turn, zooplankton grazing pressure on phytoplankton may be reduced by trout, thus promoting phytoplankton growth, increasing nutrient recycling and potentially causing a trophic cascade effect (Eby et al. 2006). A recent study in the Lake Strobel plateau found that stocked lakes have substantially higher cyanobacteria abundances than fishless lakes (Izaguirre and Saad 2014), indicating changes in the phytoplankton communities (see Chaps. 5 and 14). These processes may be more severe in vegetated lakes, where the abundance and quality of macrophytes can be affected, altering the suitability of these waterbodies as reproductive habitats for grebes. The introduction of trout has been correlated with a decline in breeding numbers of hooded grebe and other waterbirds at certain lakes (Konter 2008) (see Chap. 14).

Recently introduced predators such as the American mink (*Neovison vison*) (Peris et al. 2009) have changed the trophic structure of freshwater ecosystems in Patagonia. This species is a predator introduced extensively into Europe, Asia and southern areas of South America. In Argentina, mink farms were started in the 1950s in the southern province of Chubut. Because of successive escapes, either accidental or through the abandonment of nurseries (Pagnoni et al. 1986), this mustelid has increased its range to the north and east, following the Andean numerous rivers and lakes. The impact of American mink on native Patagonian freshwater species is well documented. This alien mammal has already caused a pronounced decline of the critically endangered *P. gallardoi* all over its distribution range (Fasola and Roesler 2018).

Among other introduced predators, the Patagonian grey fox (*Dusicyon griseus*), introduced in Tierra del Fuego as a control for the European rabbit (*Oryctolagus cuniculus*), is perhaps contributing to the decline of *Chloephaga rubidiceps* (Ruddy-headed goose) (Chebez and Bertonatti 1994). Predation by feral cats and dogs was also reported as a severe threat for *P. socialis* in non-breeding sites (Ferrari et al. 2003).

## 5 Needs for the Long-Term Conservation of Amphibians and Waterbirds in Patagonian Freshwaters

The current situation of Patagonian amphibians and waterbirds is of great concern. Therefore, development of management strategies is urgent in order to promote the long-term conservation of these species and their habitats. This section aims to review the progress made in planning and application of management actions and necessary research on these taxa. Information provided by the IUCN website was summarised for each of the species identified above, taking into account conservation actions and research needed, and conservation actions in place were taken into account. We also included information from the National Red List for amphibians (Vaira et al. 2012) and the national action plan for amphibians and suggestions pointed out by Úbeda and Grigera (2007). Finally, some of these actions and others not specified in the IUCN framework are discussed.

## *5.1 Actions Needed*

The IUCN recognises different types of conservation and research needed and actions in place (actions that are being or were conducted) (Table 15.3). Therefore, during the assessment process, specialists are asked to use this classification scheme to indicate both conservation actions and research needed for the evaluated species. In addition, specialists are also asked to consider the most urgent and essential actions that can be taken in the short term.

**Table 15.3** First- and second-order categories of (**a**) conservation actions, (**b**) research needed and (**c**) actions in place recognised by the IUCN Red List

| **a Conservation actions** | |
|---|---|
| 1. Land and water protection (***6***) | 1.1. Site/area protection (**13**) (***5***); 1.2. Resource and habitat protection (**9**) |
| 2. Land and water management (***4***) | 2.1. Site and area management (**5**) (***3***); 2.2. Invasive/problematic species control (**5**); 2.3. Habitat and natural processes restoration |
| 3. Species management (***4***) | 3.1. Species management (**1**); 3.2. Species recovery; 3.3. Species reintroduction; 3.4. Ex situ conservation |
| 4. Education and awareness (***2***) | 4.1. Formal education; 4.2. Training; 4.3. Awareness and communication |
| 5. Law and policy (***2***) | 5.1. Legislation (**2**); 5.2 Policies and regulations; 5.3. Private sectors standard and codes; 5.4. Compliance and enforcements |
| 6. Livelihood, economics & other incentives | 6.1. Linked enterprises and livelihood alternatives; 6.2. Substitution; 6.3. Market forces; 6.4. Conservation payments; 6.5. Non-monetary values |
| **b Research** | |
| 1. Fundamental research (***13***) | 1.1. Taxonomy; 1.2. Population size, distribution and trends (**14**) (***13***); 1.3. Life history and ecology; 1.4. Harvest, use and livelihoods; 1.5. Threats (**10**); 1.6. Actions |
| 2. Conservation planning | 2.1. Species action/recovery plan; 2.2. Area-based management plan; 2.3. Harvest and trade management plan |
| 3. Monitoring (***1***) | 3.1. Population trends (**7**) (***1***); 3.2. Harvest level trends; 3.3. Trade trends; 3.4. Habitat trends |
| **c Actions in place** | |
| 1. Research and monitoring | 1.1. Action recovery plan (**1**) (***2***); 1.2. Systematic monitoring scheme (**2**) (***17***) |
| 2. Species management | 2.1. Successfully reintroduced or introduced benignly (**1**); 2.2. Subject to ex situ conservation (**3** ) |
| 3. Land/water protection | 3.1. Conservation sites identified (**2**) (***22***); 3.2. Area-based regional management plan (**2**); 3.3. Occurs in at least one protected area (***21***); 3.4. Invasive species control or prevention |
| 4. Education | 4.1. Subject to recent education and awareness programmes; 4.2. Included in international legislation (***12***); 4.3. Subject to any global management/trade controls (***1***) |

Numbers between parentheses following some actions indicate the number of species of amphibians **(bold)** and waterbirds ***(bold italics)*** for which the need of that action ((**a**) and (**b**)) or its progress (**c**) has been identified

For amphibians, the evaluations carried out to date indicate that among the conservation actions needed, most of the specialists recognised as main actions those related to area protection and as secondary ones those associated with area management (Úbeda and Grigera 2007) (Fig. 15.3). Vaira et al. (2012, 2018) recognise a strong need to alleviate threats on amphibians' habitats, managing invasive species and monitoring the effects of emerging diseases. According to the IUCN, this implies actions to identify, establish or expand national parks and other legally protected areas by conserving or restoring sites and habitats in the first case and the environment in general in the second.

Among the first-order research needs listed for Patagonian amphibians, basic research is recognised as a priority, followed by studies based on monitoring and, finally, those that involve planning (Fig. 15.3). Regarding second-order research needs, category 1.2 refers to studies of past and current trends, and category 3.1 indicates studies of future trends.

As for Patagonian waterbirds, a lack of information on conservation and research actions needed is observed in the IUCN assessments for most species. Of the total species assessed for Argentinean Patagonia, only eight (10%) show detailed information regarding these items.

The IUCN classifies actions in place in two order categories. The first- and second-order categories contain 4 and 11 actions, respectively (Table 15.3). Among amphibians, distribution of at least 15 species fully or partially overlaps with at least one protected area of any category in Argentina or Chile.

In the case of waterbirds, the in-place action assessment is much more detailed than for amphibians. In summary, for approximately 29% (22) of the Patagonian waterbird species, conservation sites have been identified; 28% (21) of the species

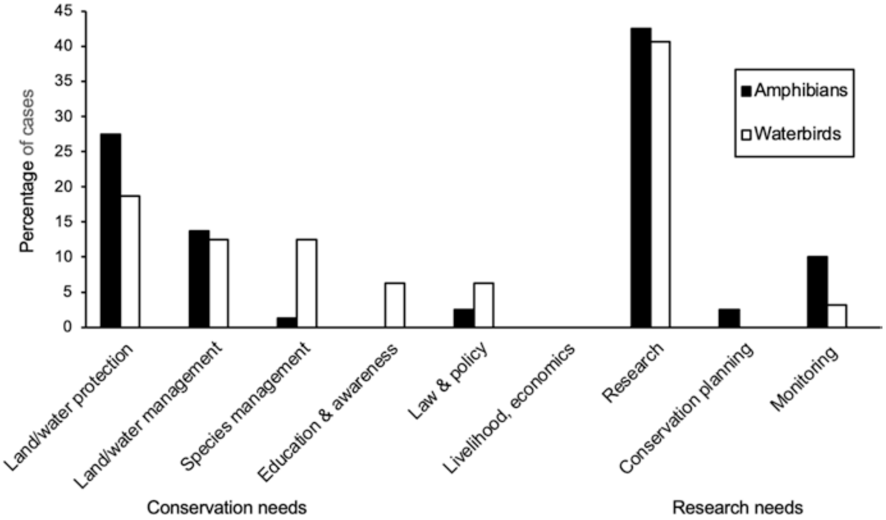

**Fig. 15.3** Percentage of cases for each category of conservation and research needs identified in the IUCN Red List. (Source IUCN 2021)

occur in at least one protected area; for 22% (17), there is a systematic monitoring scheme. Actually, the recent creation of the Patagonia National Park had as main objective the protection of nesting sites of the critically endangered *P. gallardoi* (BirdLife International 2019). At least 16% (12) are included within international legislation. Nevertheless, only 3% (2) of species are included in an action recovery plan, while barely 1% (1) is subject to international management/trade control.

These assessments indicate that the protection and management of aquatic, wetland and terrestrial habitats are recognised as a priority for both amphibians and waterbirds. Studies to evaluate past, present and future population trends were also highlighted as a priority for both taxa. Despite the documented effectiveness of highly endangered species management, this action was only considered as a priority for a strikingly low number of amphibian species (Arellano et al. 2017b; Martínez Aguirre et al. 2019). This could be related to the lack of information at national level regarding the effectiveness of management actions to improve the conservation status of threatened populations and species. In fact, at a national scale, few projects are pragmatically including species management actions although it is worth mentioning that some of the most important ones are being developed in Patagonian freshwaters (see details in Sect. 7).

## 6 Conservation Priorities Based on Endangered Species and Habitats

Because conservation resources are limited, it is mandatory to decide where to focus the management and conservation efforts. There are several ways to determine conservation priorities (Fattorini 2006; Álvarez-Berastegui et al. 2014; Kacoliris et al. 2016). In this section, we will focus on one of the simplest methods for prioritisation, based on the protection priority of species and their habitats considering their conservation status. A list of priority species is provided below, whose distribution range is showed in Fig. 15.4 in order to identify priority areas for their protection.

*Priority Amphibian Species*

- *Alsodes verrucosus* (endangered). This species is known from two localities in Andean Chile: Cautin and Puyehue, along 11 km of the El Salto Basin estuary, on the west side of the Puyehue volcano and adjacent to Puyehue National Park. It occurs in rivers, streams and creeks associated with forests (IUCN SSC Amphibian Specialist Group 2019a).
- *Alsodes neuquensis* (endangered). This species occurs in small alpine/montane lakes and streams in volcanic tablelands from Lonco Luan Plateau, Primeros Pinos, and nearby areas, Moquehue stream, Batea Mahuida Pond and Caviahue, in Neuquén province, Argentina (Cei 1976, 1987; Úbeda et al. 2012; IUCN SSC Amphibian Specialist Group 2019b).

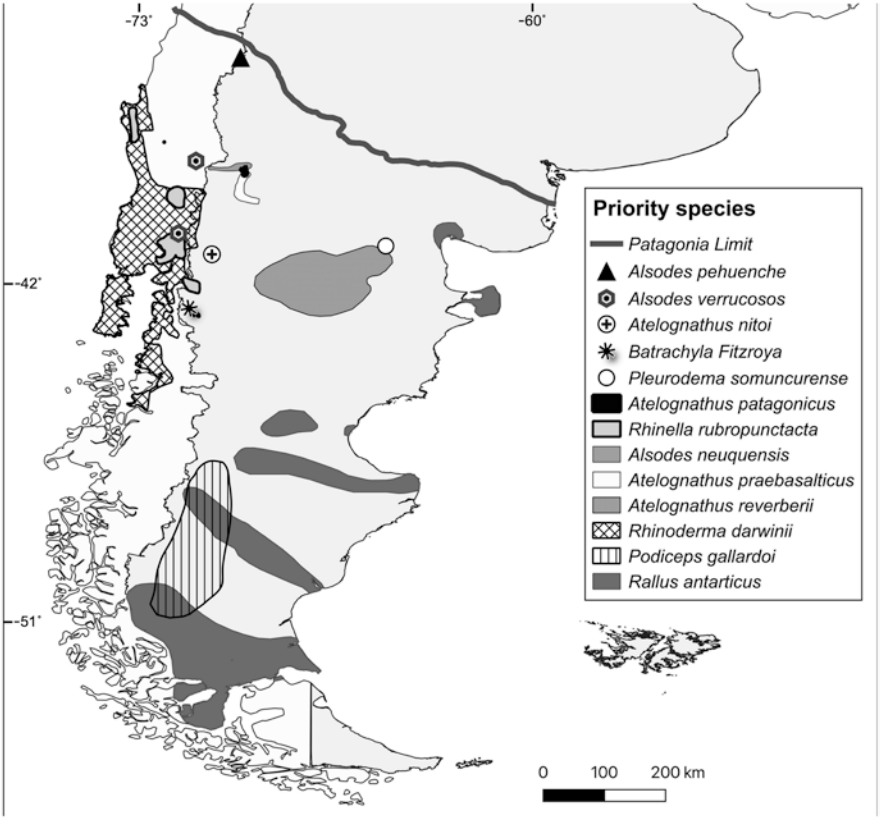

**Fig. 15.4** Range of priority species among amphibians and waterbirds from Patagonia with a focus on Argentinean species

- *Alsodes pehuenche* (critically endangered). This species occurs in Chile and Argentina. In Argentina, this amphibian species was recorded in several streams of the Pehuenche Valley, in Malargüe Department, Mendoza province, Argentina (Corbalán et al. 2010).
- *Atelognathus patagonicus* (critically endangered) and *A. praebasalticus* (endangered). Both species are endemic from a system of endorheic and isolated lagoons scattered on the basaltic plateau in and around Laguna Blanca National Park, in midwestern Neuquén province (Cuello et al. 2017; IUCN SSC Amphibian Specialist Group 2019c, d).
- *Atelognathus reverberii* (vulnerable). This species is endemic from isolated and temporary lagoons scattered over the Somuncurá Plateau in Río Negro and Chubut provinces (Cei 1969; Martinazzo et al. 2011).
- *Batrachyla fitzroya* (vulnerable). This species was recorded just in the type locality (Isla Grande in Lake Menéndez; Basso 1994), a protected island located in

Los Alerces National Park, Chubut province. This species reproduces in ponds and vegetated and shallow shores of the lake (Pastore et al. 2013).

- *Pleurodema somuncurense* (critically endangered). This species is known only from the headwaters of Arroyo Valcheta in Somuncura Plateau, Río Negro province (Velasco et al. 2016).
- *Rhinella rubropunctata* (vulnerable). This species occurs in seasonal freshwater marshes and pools located in the temperate forests of Chile and Argentina. In Chile, it is currently recorded in the following five sites: Lago Todos Los Santos and nearby Petrohué (Los Lagos region), Lanalhue Lake in Arauco Province and Cerro Adencul (Araucania Region) and Puelo (M. Mora pers. comm. 2018). In Argentina, the species is restricted to southern Río Negro and northern Chubut provinces, consisting of a single record from Los Alerces National Park (Úbeda and Basso 2012).
- *Rhinoderma darwinii* (endangered). It occurs mainly in Chile and tangentially in Argentina, where it is known from 11 localities in Neuquén, Chubut and Río Negro provinces. It occurs in wet forests and wetlands inside forests like bogs, marshes, swamps, fens and peatlands (Soto-Azat et al. 2013).

*Priority Waterbird Species*

- *Rallus antarcticus* (vulnerable). This species occurs in Chile and Argentina. In Argentina, it inhabits marshy Patagonian steppe wetlands located in Santa Cruz and Chubut provinces (de Miguel et al. 2019).
- *Podiceps gallardoi* (critically endangered). This species occurs in Chile and Argentina and in Argentina breeds on a few basaltic lakes in the interior of Santa Cruz, extreme southwest Argentina. The only known wintering grounds are located in the estuaries of Río Coyle, Río Gallegos and Río Chico on the Atlantic coast of Santa Cruz (Roesler et al. 2011; Roesler 2016).

The conservation of the amphibians and waterbirds identified as priority species for Patagonian freshwaters implies the protection of different aquatic, wetland and terrestrial habitats distributed throughout their ranges. That is, this prioritisation of species also implies a prioritisation of sites to be protected. In turn, the efforts to conserve some of these priority species can also protect others. For example, the endeavours to protect *Rhinoderma darwinii* contribute to the protection of other species inhabiting within their range (e.g. *Alsodes verrucosus* and *Rhinella rubropunctata*) as well as the remaining biodiversity (Fig. 15.4). In this sense, some of the Patagonian priority amphibians and waterbirds can be considered as umbrella species.

However, protecting areas through the establishment of natural reserves is not always enough to effectively conserve endangered species. Pragmatic management aimed at recovering species and restoring habitats is urgently needed as well. The following section gives some examples of conservation projects focused on priority amphibians and waterbirds in Patagonian freshwaters.

## 7 Amphibians and Waterbirds as Flagships to Conserve Freshwaters and Terrestrial Habitats

In this section, we will describe, through some concrete examples, how some species of amphibians and waterbirds have served as flagship species for the protection of habitats and their biodiversity. Particularly, we highlight the hooded grebe (*Podiceps gallardoi)* conservation project, which is one of the main species-framed conservation projects in Patagonia.

### El Rincón Stream Frog Conservation Project

The El Rincón stream frog (*P. somuncurense*) is endemic of the hot springs of the headwaters of the Valcheta Stream at the Somuncurá plateau in northern Patagonia. This species is currently restricted to a few isolated subpopulations along the headwaters of this stream (IUCN SSC Amphibian Specialist Group 2016; Velasco et al. 2016; Velasco 2018). In the last four decades, the frog population has dramatically declined, resulting in the extinction of some subpopulations (Velasco 2018). A total of 70% of the current subpopulations have small sizes (less than 250 mature individuals), which threatens their long-term viability (Velasco et al. 2019).

The drastic decline of this frog was mainly caused by (i) the expansion of the invasive rainbow trout (see Chap. 14), an aggressive predator, associated with the extinction of the frogs along the stream (Velasco et al. 2018), and (ii) the habitat destruction by livestock, which overgrazes and tramples vegetation, reducing the availability of food, shelter and breeding sites. In addition, decomposition of livestock faeces in the water promotes eutrophication (IUCN SSC Amphibian Specialist Group 2016; Velasco et al. 2017; Arellano et al. 2017b).

The IUCN lists this species as critically endangered; the Zoological Society of London includes the species in the top 100 EDGE (evolutionary distinct and globally endangered) species, and the Amphibian ARK identified the urgent need for its captive rescue and management (IUCN SSC Amphibian Specialist Group 2016). Furthermore, conservation experts from the Amphibian Survival Alliance and the IUCN-SSC Amphibian Specialist Group recommended urgent conservation actions to protect this species (http://www.amphibians.org). Based on priorities stated in the Amphibian Conservation Action Plan of the IUCN-SSC-Amphibian Specialist Group (Wren et al. 2015), the El Rincón Stream Frog Conservation Action Plan listed the main threats and necessary conservation actions (Kacoliris et al. 2018). This latter plan also followed the suggestions of the Argentinean Conservation Action Plan for Amphibians (Vaira et al. 2018).

In 2012, a work team started a conservation programme with the overall goal of ensuring the long-term viability of this species and its habitat through three main objectives:

(1) Alleviate the main aquatic threat to frogs by removing invasive trout from the stream headwaters and restoring the habitat for this frog and other aquatic species. Removal of trout is being achieved by creating fish barriers to restrict access of trout upstream and removing trout upstream those barriers.

(2) Improve terrestrial and wetland habitat quality. This objective focuses on increasing food availability, shelter and breeding sites and reducing the eutrophication in critical habitats for frogs (i.e. hot springs). Actions included avoidance of livestock access to hot springs by fencing, which contributed to a quick habitat restoration (Arellano et al. 2017b).

(3) Recover frog subpopulations in restored habitats wherefrom the species had been extirpated. This objective combines ex situ breeding followed by reintroduction of individuals in restored sites. Between 2017 and 2021, four reintroductions were conducted with at least one subpopulation successfully re-established in the wild (Martínez Aguirre et al. 2019).

Another component of this project is the effective implementation of a natural reserve by acquiring land to protect habitats. Since the conservation of El Rincón stream frog depends upon the alleviation of both aquatic and terrestrial threats, the conservation of this single species leads to the protection of the entire habitat and its biodiversity. In summary, the current achievements of this project demonstrate how an amphibian, because of its particular life cycle that integers aquatic, wetland and terrestrial environments, can be successfully used as a flagship to promote conservation at a bigger scale.

**Patagonian Frog Conservation Project**

The Patagonian frog (*A. patagonicus*) is an endemic species that only lives in a small number of isolated shallow lakes scattered over the volcanic tablelands of north-western Argentinean Patagonia (Cuello et al. 2009). This species used to be common in these freshwater ecosystems, with the largest subpopulation (50% of the total number of individuals) inhabiting the Laguna Blanca shallow lake located in the homonymous National Park, the main and unique permanent waterbody in the area. Unfortunately, exotic predatory fishes were introduced in this shallow lake for touristic purposes, even though the area was declared as National Park in 1945 (Fox et al. 2005). The effect of these top predators was evident, and after a decade of surveys without a single record, in 2004, this subpopulation was declared extinct (Fox et al. 2005). Smaller subpopulations remain in temporary and isolated ponds, facing human-related threats. Moreover, between 2010 and 2016, an unusually prolonged and severe drought desiccated these ponds, increasing the estimated population decline of the species from 50% to more than 90% (Cuello et al. 2017).

Based on the available information regarding the status of the Patagonia frog, the National Park Agency implemented a management plan for its conservation, based on habitat protection and fish control. Habitat protection focused on small ponds and included fences to avoid habitat destruction by livestock. The fish control is conducted to reduce fish numbers (Buria, Pers. Comm.). However, the natural recolonisation by frogs is unlikely because of the poor condition of the corridors and the remaining stock of introduced fish in the shallow lake. Furthermore, summer droughts are now more frequent due to climate change, thus increasing the extinction risk for this species unless subpopulations can be re-established in the Laguna Blanca National Park.

In 2018, after a series of workshops with stakeholders, an action plan to ensure the long-term viability of this species was proposed (Kacoliris et al. 2020), with the vision of ensuring meta-population dynamics by creating sanctuaries within the National Park based on habitat restoration, management of threats and re-establishment of subpopulations. As part of this vision, in 2018, an ex situ survival colony of Patagonia frog was established in the Laguna Blanca National Park. Current activities aim to continue in this direction in order to have individuals for future reintroductions.

**Hooded Grebe Conservation Project**

The hooded grebe (*P. gallardoi*) inhabits and breeds in a few lakes in the Santa Cruz province, Argentina. The species might also be a summer visitor in Torres del Paine National Park, southern Chile, but at present, there are not confirmed records in that country (Roesler et al. 2011; Roesler 2015). This species makes floating nests over aquatic vegetation which also supports the thrive of several aquatic invertebrates which are part of its diet (Chebez and Bertonati 1994). Hooded grebes breed in colonies but have a very low reproductive rate (0.2 chicks year 1 per adult; O'Donnell and Fjeldså (1997)). Although colonies can establish in marginal areas of its distribution range (O'Donnell and Fjeldså 1997), individuals commonly show a high fidelity towards the plateaus where they were born (Roesler et al. 2016).

In the 1980s, the total population of this waterbird was of up to 5,000 individuals (Fjeldså 1984; O'Donnell and Fjeldså 1997), but during the past 25 years, it has suffered a population decline of about 80% (Roesler 2016). The main threats driving this decline seem to be the predation impact by invasive species (American mink and exotic salmon and trout) as well as habitat loss by climate change (Imberti and Casañas 2010; Casañas et al. 2013; Roesler et al. 2016) (see also Chap. 14). American mink threaten grebes during all their life stages by predating on nests, chicks and adults (Roesler 2015; Fasola and Roesler 2018). This impact is very high, since just one mink can kill more than half the adults in a breeding colony (Roesler et al. 2011). Invasive fish (trout) not only can predate on chicks (Konter 2008) but also competes with this waterbird for food and modifies water conditions, making lakes unsuitable habitats to grebes (Roesler 2015; Izaguirre et al. 2018; Francisco et al. 2019). Like endangered frogs, hooded grebes (as well as other waterbirds) are also being impacted by climate change. Unpublished data as well as some climate predictions show a decrease in winter snowfall, increase in temperatures and decrease in precipitations for Patagonian plateaus (Burgos and Ponce 1991). These scenarios foresee longer dry seasons which will result in a loss of aquatic habitats for this species (Konter 2008). Therefore, the hooded grebe was listed as a critically endangered species in the IUCN Red List and is considered one of the most endangered bird species in Argentina.

NGO Aves Argentinas along with Ambiente Sur are coordinating conservation efforts to ensure long-term survival of hooded grebes. In 2014, these NGOs boosted the creation of Patagonia National Park, a 52,000 ha protected area that covers half of the breeding colonies of hooded grebes (Roesler 2015; BirdLife International 2019).

Since 2009, intense research and monitoring activities have been conducted to understand hooded grebe distribution, ecology and the impact of threats in order to help guiding conservation efforts (Roesler 2015; Roesler et al. 2016). A programme called "Colony Guardians" has also been established to protect nests from invasive predators, achieving a significant increase in survival rates among several colonies (Roesler et al. 2016). Also, awareness raising activities, including displays, theatre productions, video and outreach material reached over 100,000 people.

At present, management plans to eradicate American minks and invasive trout from the high plateau areas are in place (Roesler et al. 2016). Current data gathered through these programmes showed that minks are decreasing in target areas. Moreover, a captive rearing programme is conducted with the aim of raising wild eggs in captivity and releasing them back to its habitat as a way to increase survival and recruitment (Roesler 2015). However, as for 2020, no captive-reared chicks had so far been released (Roesler com pers). Extensive conservation efforts are showing that management actions applied were successful and now the population is stable. If an increase in hooded grebes is achieved in the future, this endemic species might have a higher chance of avoiding extinction.

## 8 Final Remarks

More basic information concerning ecological linkages, benefits, off-site effects and cost-effectiveness of different actions across realms is much needed to enhance biodiversity conservation (Adams et al. 2014). On a specific level, deepening current ecological knowledge of those taxa linked to both freshwater and terrestrial environments would be helpful to develop cross-realm conservation strategies. Amphibians and waterbirds can be used as models and flagships for this purpose. Argentinean Patagonia hosts several priority species among these groups, including some of the most endangered amphibians and waterbirds worldwide. These species act as ecological linkages between almost all the freshwater types from this region and the terrestrial habitats, as most of them occur in shallow lakes, ponds and other wetlands, and some of them also inhabit rivers and streams. Considering that aquatic, wetland and terrestrial environments often share the same threats, the efforts needed to protect these species from those threats will also help to protect their freshwater and terrestrial habitats.

Among the mentioned threats, habitat disturbances caused by livestock overgrazing and trampling, and invasive species, mainly predatory fish and minks, are causing population decline in several endangered native species. Some studies already account for the impact of these threats. For instance, Cossa et al. (2020) identified the negative effect of livestock on *Chloephaga* spp., and Velasco et al. (2018) recorded a decrease in occupancy of native amphibians due to trout presence. In addition, climate change was also recognised as an important threat for critically endangered species with effects already observed, like in the case of *A. patagonicus* and *P. gallardoi* (IUCN SSC Amphibian Specialist Group 2019b; Lancelotti et al.

2020). Projected scenarios should be used to assess the potential impact of climate change on most of the priority species of amphibians and waterbirds from Patagonia to urgently develop conservation strategies to mitigate its effects. Particularly, damming and mining projects located in areas with presence of endangered amphibians and waterbirds should be closely monitored, and compensatory strategies for their conservation should be put in place.

Regarding conservation actions required to protect these species and their habitats, researchers, conservationists and managers consider the creation of new protected areas and management of these areas and existing ones. However, the establishment of new protected areas is not always the best strategy to ensure long-term biodiversity conservation (Montesino-Pouzols 2012). On the other hand, recovery strategies framed on species management have proven effective for some Patagonian amphibians and waterbirds (Roesler et al. 2016; Martínez-Aguirre et al. 2019). In the former case, species management was poorly identified as a main need during the assessment conducted by the IUCN. This may be due to the absence of conservation projects that apply species management, with the consequent lack of detailed information about the effectiveness of these methods at local scale (see Arellano et al. (2017b) and Martínez-Aguirre et al. (2019)). Adaptive management and data from successful examples from other countries should be used as starting points to fill this gap.

Conservation priorities in the Argentine Patagonia were identified for, at least, nine amphibian's and five waterbird's species due to their conservation status. These species can be used as flagships to effectively protect their freshwater and terrestrial habitats and so protect all the biodiversity they contain. Some of the most significant conservation projects based on endangered species are being carried on with Patagonian species (El Rincón stream frog and the hooded grebe projects). These conservation projects are removing and/or alleviating the main threats for both species, and consequently, they are also promoting the protection of all biodiversity. These conservation projects have demonstrated how single species can act as flagships and umbrellas, acting as bridges to conserve aquatic, wetland and terrestrial environments.

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# Chapter 16
# Cultural Limnology in Patagonia: Knowledge and Water Management in Mapuche Rural Communities

**Soledad Molares, Daniela Morales, Juana Aigo, and Juan Carlos Skewes**

## 1 Introduction

### *1.1 Cultural Limnology: An Overview of Local Freshwater Knowledge*

Knowledge, practices, and values that different societies can establish with freshwaters are complex and differ according to culture and geography, as well as to different historical moments. Among urban societies, for example, waters are considered as part of the natural capital of each politically delimited space, which allows for human and animal life, as well as for economic and industrial development. In this sense, water bodies play the role of input for agricultural states growth, routes for fluvial transport of consumer goods, generators of hydroelectric energy, and even channels for waste release (Benez et al. 2010; Gartin et al. 2010). As long as their availability and access are politically determined, waters establish different opportunities for the growth of certain forms of progress, Western modernity, and wealth accumulation (Merlinsky et al. 2020).

Cultural limnology, as a scientific transdiscipline which approaches these issues, is interested in understanding the knowledge and practices associated with

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hydrological cycles and their biology, their relationship with festivities, also with annual productive calendars, with water management and governance, etc. (Trawick 2001; Gartin et al. 2010; Linton and Budds 2014; Usón et al. 2017). In recent years, these issues have begun to gain importance among scientific, technological, and governmental sectors, due to questions from different social and academic spheres to the prevailing logic of industrial and commercial use of water bodies (tourism, forestry industry, hydroelectric plants, mining), which many times exclude local populations from decision-making (Marino et al. 2009), and also due to failures in biological conservation, sanitation safety planning (OMS 2016), and water safety plan implementation (WHO 2005; Martínez et al. 2014; Morales et al. 2020).

In this sense, cultural limnology can document alternative scenarios to those of globalization, as examples of other ways of relating to water. Indeed, in rural and indigenous contexts around the world, alternative forms to relate with water are possible. Generally, these relationships are based on close, time-anchored contact of societies with their environment (Woodward et al. 2012; Guerra et al. 2019) and respond to complex, multidimensional and particular ways of seeing the world, in continuous adaptation to various socioeconomic change processes (Berkes and Turner 2006; Molares and Ladio 2014; Iniesta 2015).

The environmental perspectives of indigenous and rural societies are usually invisible in the whirlwinds of hegemonic scientific thought, or they are defined from Western frames and representations (Usón et al. 2017). As suggested by water anthropology, a field self-declared as relational, it is necessary to overcome the opposition between nature and culture in order to understand that social and ecological distinctions associated with water are nothing but semantic conventions (Ballestero 2019).

In this sense, it is necessary to admit that water is a critical component of landscapes and territories and that it is related to the way people identify themselves. Waters are at the basis of different residential arrangements and provide for diverse subsistence practices and modes of inhabiting the land. Such a basis implies recognizing some agency in this and other elements of the environment and, for some peoples, the possibility of engaging an intersubjective relationship with them (Rivera-Cusicanqui 2013; Di Giminiani 2018). In this broad sense, water is a fundamental part of local communities' biocultural memory (Caparelli et al. 2011; Skewes et al. 2011, 2012; Aigo et al. 2020).

Biocultural memory about waters manifests in particular symbols and practices, whose meanings are encoded in the material and nonmaterial culture, in language, iconographies, and myths, constituting part of their traditional limnological knowledge (Aigo and Ladio 2016). This knowledge is of community character, since it is transmitted from generation to generation through language and shared practices; it is also of spatial character because it adjusts to the inhabited landscape. Due to its dynamic nature, knowledge expands with the cognitive processing of hydrological events (e.g., droughts, storms, floods) and their effects on topography, flora, fauna, and local economies (Toledo and Barrera-Bassols 2008). Limnological knowledge, in this sense, implies the adoption of new criteria and sufficient evidence to respond to environmental events, frequently catastrophic, and even to state or private corporations. Thus, it constitutes critical knowledge for the communities' biocultural resilience and resistance (Iniesta 2015).

From technical to ritual dimensions, behavioral norms contrive the relationship with waters, which, in turn, impacts the landscape. Water should be obtained from streams; wetlands are regarded as numinous places, and its consumption is regulated during spiritual and community ceremonies. Waters cannot be separated from human and nonhuman components of the environment. Its availability and use patterns give rise to diverse waterscapes, and even under environmental stress, its culturally informed management allows for the continuation of life (Skewes et al. 2012; Aigo et al. 2020). The definitions of water and waterscapes require a biocultural dimension, capable of containing its materiality, ecology, and history, where the complex life networks, places, and worldviews concur (Skewes et al. 2011; Ceballos et al. 2012). Though aligned with other points of view in the scientific multidisciplinary field of limnology (Edmondson 1994), cultural limnology perspective incorporates this symbolic and living character of waters, which governmental and conservation policies should take into account in indigenous and rural territories.

## *1.2 Historical and Geographical Context of Mapuche Limnological Knowledge*

Mapuche are the largest indigenous people in the Southern Cone. Throughout *Wallmapu* (Mapuche ancestral territory), between the Atlantic and Pacific oceans, the Andes mountain range has been, since ancient times, land of refuge and circulation space for different communities located both to the East, that is, *Puelmapu*, and to the West, that is, *Gulumapu* (today Argentina and Chile, respectively). In these vast landscapes, postglacial remnants, the snowy regime, rains, and the contribution of water from the temperate forests nourish the main river basins on which these entire ecosystems and its human and nonhuman components depend (see Chap. 9).

In the nineteenth century, capitalist cattle, agricultural, and timber industries, promoted by the Argentine and Chilean States and sustained by military occupations, the misnamed "Conquest of the desert" in Argentina and the "Pacification of the Araucanía" in Chile, took over the best and most fertile valleys and farmlands, forcing Mapuche communities to move to rugged mountains and drier lands, drastically reducing its territory (Zabala 2000; Briones and Del Río 2002). Today, Mapuche people live in rural communities, or many of them have migrated to the cities looking for jobs and educational opportunities. Mapuche people have a strong identity and keep most of their traditions and language alive. However, the Chilean and Argentine States of the twenty-first century have not yet resolved their relationship with Mapuche society, which continues to be one of the most impoverished and marginalized social groups in both countries (Bengoa 2012).

The long-standing relationship between Mapuche and waters has sedimented a deep biocultural memory recreated in the new scenery the communities confront. This memory allows them to discern the behavioral aspects of the elements, which are imperceptible for western observers. The flight of a bird, the breeze, or an escaping fox are signals that, when deciphered, allow the community to anticipate events or to orient their behavior according to patterns ingrained in their living practices (Guevara 1913). As suggested by Bengoa (2003) and Carabias et al. (2010), the history of Mapuche is closely connected to rivers and water streams. They intensively used the courses of water to create navigation systems, allowing exchanges and social bonding in the vast *Wallmapu* territory. The abundance of trees allowed the construction of *wampos* (canoes) used as a primary means of transportation. Such was the importance of navigation that in the Andean lakes where the invasion took over the land in the early XX century, it was prohibited by Chilean authorities and substituted by boats and ships managed by private companies (Skewes 2019).

Language, practices, and cosmology reflect the importance of water in Mapuche ways of life. Cultural spiritual beings such as *Shumpall*, *Abuelito Huentellao*, *Manquian*, and others maintain their presence either as embodied life forces in the rocks on the coast or as immanent spirits in the lakes. The word *lafquen* indistinctively applied to the ocean or the great lakes has a mystical meaning. Likewise, the Milky Way corresponds to the *wenuleufu*, the river of the sky, where the original ancestors of the contemporary Mapuche live. Water sacredness and its deep meanings translate into the reverential treatment water must receive. Waters are living entities within a continuum of relationships sustained as far as reciprocity and mutual respect between them and humans exist, a meaning that invites us to transcend the utilitarian notion (Di Giminiani 2018).

In the context of environmental disasters due to overexploitation of goods provided by nature, the biological conception of nature refuges demands a new approach of a biocultural nature (Barthel et al. 2013). In Patagonia, the sub-Antarctic forests and the riverine groves that make up the green spots of the steppe are inextricably linked to Mapuche communities. As such, these biocultural refuges protect the waters, while people keep alive their claim for the lands that have been taken away from them in the name of development. The communities' accommodation to waterscapes in a context of devastation and uncertainty is a critical point from which to study waters in its multiple facets and expressions.

In this chapter, we deal with some aspects of the Patagonian cultural limnology, analyzing the Mapuche experience in different rural enclaves (Fig. 16.1) where waterscapes, as part of Mapuche universe, incarnate the spatial accommodation of aesthetic, symbolic, and economic practices that characterize these people (Skewes et al. 2012; Skewes and Guerra 2016; Ibarra and Riquelme 2019, Aigo et al. 2020). The case studies are in Argentina—Costa del Lepá (42° 41′ S–70° 45′ W), Gualjaina (42° 43′ S–70° 32′ W) (Sub-Andean steppe, Chubut province), and Raquithue and Lafquenche (39° 56' S–71° 03' W) (Sub-Antarctic forest, Neuquén province)—and in Chile: Maihue (40° 15' S–72° 01' W) and Rupumeika (40° 18' S–71° 58' W) (Sub-Antarctic forest, Los Ríos Region) (Fig. 16.1). A qualitative approach which included interviews, observation, participatory workshops, and hikes with the

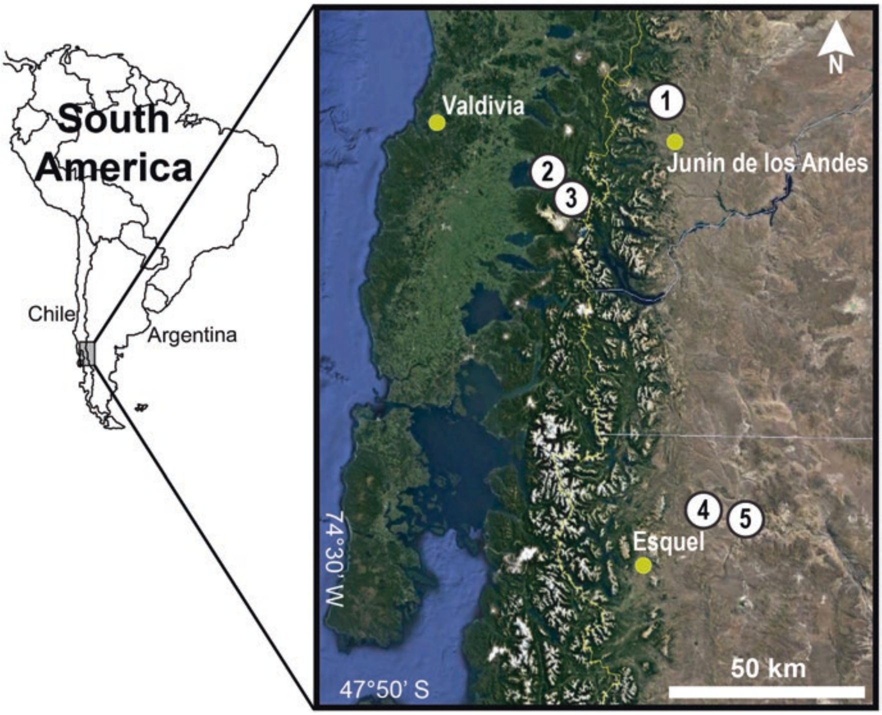

**Fig. 16.1** Map of study area. 1: Raquithue and Lafquenche, Chimehuin, Neuquén Argentina; 2: Maihue, Los Ríos Chile; 3: Rupumeika, Los Ríos Chile; 4: Costa del Lepá, Chubut Argentina; 5: Gualjaina, Chubut Argentina

residents was the primary tool of this research work, while observing ethical guidelines, including consent forms with the collaborators in these communities (International Society of Ethnobiology 2006; United Nations 2012; Albuquerque et al. 2014).

## 2 The Becoming of Waters: Some Experiences in Mapuche Communities of Patagonia

### *2.1 Drinking Waters in Patagonian Arid Zones*

From Eastern Andes cordillera basins to Patagonian arid steppes, water flowing superficially, together with springs, *mallines* (glacial wetland, see Chap. 10), and other groundwater forms, make up complex networks which sustain the settlement of Mapuche communities in the territory. Surface waters, often scarce in summer and cloudy or muddy during fall floods, are mainly used to cultivate food and forage; groundwater, which emerges discontinuously in land surface, is mainly considered as drinking water. This is so in Costa del Lepá and Gualjaina communities (Chubut river basin) (Fig. 16.1), where, as in the great majority of communities in the region, there is no main water supply provided by the State administration.

Groundwater preference and the relationships built with them when used as drinking water come from organoleptic, thermal, and ecological criteria, which constitute a guide for perception and quality testing processes, as well as from the deep symbolic meaning of springs as powerful entities having *ngen* (strength in *mapuzungun*). These criteria, as mnemonic and heuristic tools, allow to remember, associate, and transmit essential information during the teaching-learning processes on this kind of water, as well as on the need to protect and value springs as a key biocultural heritage for survival. In this sense, for Costa del Lepá and Gualjaina communities, water emerging in springs is "sweeter" and "fresher" than others, and it "runs"—unlike stagnant water—"leaving bacteria on both sides" guaranteeing its purity and the development of many *lawenes* (medicinal plants) such as *Mentha* spp. (menta, poleo), *Equisetum bogotense* (limpia plata), *Apium prostratum* (apio de campo), among others, of great cultural importance (Molares and Ladio 2014). Besides, their ancients (ancestors) used to drink water from springs, with which medicine is prepared, and also the one with which "mate" is made (yerba mate, typical infusion of *Ilex paraguariensis*) so that "a little foam is formed"; that is why it is considered the healthiest water. Inversely, tap or bottled water tastes like chlorine, and it is drunk only when there is no other option (away from the house or after occasional flooding), or under health agents' indication (official medical system).

Unlike rivers, streams, or big water springs that are for community use, small springs are often of exclusive family use. Each family accesses water using buckets, jugs, or sometimes hoses, rarely pumps; they clean it from algae and keep dogs, farm animals, and trespassers away, thus avoiding to disturb the spring, who bubbles angrily and even punishes by drying up or emerging somewhere else. When springs dry up, drinking water runs out.

In recent years, increasing droughts caused many places to dry up in the region; according to some elderly people, this phenomenon was the result of transgressing certain respect and reciprocity customary rules toward *ngenko* (water protecting force, guardian of the water), thus disrupting the sacred order of springs (Morales et al. 2020). The communities internally discuss this problem with the premise that it is necessary to strengthen the practices of gratitude and reciprocity to improve the situation. Organization around the *nguillatun* will be key, as will also be described in Sect. 2.3. In the meantime, the adaptive response of the people is the continuous exploration of the territory in search of new small water springs, a practice that has been developed in the region for generations, though more intensively nowadays. Recognizing this intermittency in behavior and changing availability of springs has placed them in an outstanding place regarding territory preservation, even though there are some other ways to have access to groundwater, such as wells (Fig. 16.2) and drilling.

However, it is important to point out that local preference for groundwater sources is in tension with its microbiological aptitude according to laboratory

**Fig. 16.2** Costa del Lepá's woman drawing drinking water from a well near her house. (Source: Photograph by Daniela Morales)

parameters which indicate that water is not apt for human consumption in more than a half of the springs (e.g., high values of *Escherichia coli*, *Pseudomonas aeruginosa*, and coliform bacteria, as recommended by the World Health Organization and the Argentine Food Code). This is even more significant for rivers and streams, since water is not suitable for drinking in any of them, mainly due to the presence of livestock and their defecation and the household waste (plastics, cans, etc.) and/or due to the proximity of latrines and human feces (Morales et al. 2020).

Despite having received information from local health workers, recommendations to boil water, supply of chlorine tablets or other disinfectant products, inhabitants of these communities do not consider the use of these treatments a positive step. For people, it is not always possible to boil water as firewood is a scarce resource, while the use of chlorine is not preferred because it is considered bad for the stomach. In general, local people do not really believe that consumption of untreated water can have negative effects on their health, and so they do not follow the recommendations.

It is possible that the gap between local perspectives and State interventions is due to a lack of understanding between both points of view. That is, the perception of water quality is not necessarily the same between cultural perspectives, which are not "related to imaginary ways of seeing the world, but to the real worlds that are being seen" (*sensu* Viveiros de Castro 2004). In this context, making progress with the idea of health interculturality might help to understand these complex cultural relationships and collaborate so that the different approaches can interact in an atmosphere of equality and mutual respect. As shown in this case study, interdisciplinary approaches were useful to identify disagreements in the perception of water quality measured by microbiological and local parameters; this could be a starting point toward better communication between communities and the official care system.

## *2.2 Waters for Agriculture: A Design Based on Accommodation to Water Network*

Subsistence economy in the Patagonian steppe is mainly based on sheep and goat raising and, to a lesser extent, on family horticulture. This practice allows for the supply of vegetables, fruits, medicinal and ornamental plants, firewood, and ceremonial elements. In recent years, selling these products has allowed to diversify monetary income and contributed to a diet which includes healthy food free from agrochemicals (Ladio et al. 2013). This is remarkable if we consider the distance which separates these rural communities from commercial centers and the limiting edapho-climatic characteristics they cope with: rainfall never over 150 mm a year, intense winds, and the predominance of poor soils (Davel et al. 2015), which turn horticulture into a great challenge. In addition to this, we must consider precarious conditions of basic services and poor State support, as it has been already mentioned.

This is the context shared by Costa del Lepá and Gualjaina communities (Fig. 16.3), wherein the management of horticultural systems depends on local knowledge which guides water and soil use practices and preservation. Part of this management comprises working within the horticultural sector (greenhouses, orchards, and farms), selecting the plant varieties most resistant to water scarcity, and preserving soil moisture (e.g., mulching). Likewise, it is necessary to organize the environment by installing forest curtains, which reduce wind effect and provide shade, and to locate the domestic horticultural infrastructure in the most humid areas within the family farm, which implies following the surface water networks and, sometimes, the springs largest streamflows (Morales et al. 2020).

In these domesticated sites of the landscape (Casas et al. 2016), people make their own design for irrigation, based on water intakes, canals, and flood plains. Water is captured from rivers or streams with water intakes made with stones, sandbags, animal hides, tree trunks, and other materials and diverted through lateral canals toward cultivation areas. Opening and closing canal circuits are decided through agreements between neighbors, considering social codes based on the efficient use of water, equitable distribution, and reciprocity among human and nonhuman beings (dogs, cows, fish, birds, sheep).

However, the scenario of water scarcity resulting from variations of the hydrogeological cycle and water reduced supply at the headwaters due to climate change (Garreaud 2018) that causes crop and pasture loss and animal mortality in the region (Roveta 2008) and has forced people to search for new management practices in order to satisfy their basic needs, such as the incorporation of innovations promoted

**Fig. 16.3** Horticulture in Costa de Lepá, Chubut Argentina. (Source: Photograph by Soledad Molares)

by State organisms (drip systems, groundwater pumping, and water reservoirs which allow to keep up to 20.000 liters). But, in general, these measures have been planned and executed in a top-down way, without considering the symbolic precepts of water, that is, not disturbing the springs, or the community or family practices of traditional management. These technologies are not available for all inhabitants, either because they are not selected as beneficiaries, or because the proposals are misunderstood and then rejected, or due to the lack of external inputs or own resources to complete the installations (e.g., bulldozers). These causes are poorly understood by the State's promoting organizations, while the innovations proposed, due to their lack of consensus, often deepen social inequalities and break community ties (Aigo et al. 2020).

### *2.3 Waters in the Transition from the Sacred to the Political Dimensions*

Mapuche worldviews and practices understand water is related to human behaviors, framed by *azmapu* (norms that regulate human relations with nature), who explains bonanza or scarcity periods (Melin et al. 2016). One of the most prominent ritual expressions of conciliatory inspiration between human behavior and the world is the *nguillatun*. In this celebration, through animal sacrifice, the ritual community restores the cosmological order that daily life alters. The Rupumeika *nguillatun*, in the eastern border of the Maihue Lake in Chile (Fig. 16.1), exemplifies, with particular precision, the practice that sacralizes the relationship with water. In that place, the Rupumeika Alto, Rupumeika Bajo, and Maihue communities come together for the celebration (Moulian 2012; Guerra et al. 2019).

During the ceremony, the ritual norms restrict water use and consumption. The congregation in the sacred place must abstain from using water beyond their bodily needs. The ritual practitioners invite their guests to carry buckets of water from the lake, situated 200 meters away from the *rehue* or sacred site, indicating them to be careful while pulling or carrying water: no spilling must occur. Any undesirable event, including spilling or shaking off water, is a dark omen, and any bad behavior during the ceremony carries undesirable consequences for the ritual and daily life. Intense and dry heat or heavy rain are unequivocal signs of the *azmapu* transgression during the *nguillatun*.

The ritual practice is oriented toward *Puelmapu*, the East, where life, sun, and water come from. Prayers and sacrifices invoke *ngechen*, people's custodian spirit, to have a regular season with no heavy rains or drought so they have no problem harvesting their crops. The *trutruka*—a five-meter long wind instrument made of caña colihue, an autochthonous bamboo (*Chusquea culeou*)—is the only object that, along the four-day ceremony, receives water as a means for protection but also because it is the ritual voice, which connects with *ngechen*, indicating the different phases of the ritual.

Climate change becomes noticeable in the celebration of the *nguillatun*. Absence of rain in the mountain results in the ritual's lack of firewood: rivers, in the past, used to carry fallen trees after the alluviums which followed the storms. Historically, the *nguillatun* benefited from the accumulated wood in the lake. Today's absence of wood forces the acquisition and felling of trees from forest renewals, which increase the community's dependence on money. Simultaneously, water scarcity threatens the social networks in Costa del Lepá and Gualjaina (Sect. 2.2) and increases costs for the performance of ceremonies in Rupumeika. However, the ritual community, united around the celebration of the *nguillatun*, starts playing new roles for the local organization's benefit. Indeed, the religious leaders of the *nguillatun* headed a social mobilization against capitalist projects.

Beyond the social dynamics related to climate change, the expansion of capitalism represents a great risk for Mapuche communities. Mountain waters are object of the voracious capital either for energy production or for tourism or real estate ventures. The emergence of powerful interests provokes inevitable tensions among residents to whom the outside business offers money or other benefits in exchange for their acquiescence.

The isolation of Rupumeika Alto and Rupumeika Bajo communities was intensified by a landowner who impeded their access to the city by restricting them from passing through his property (PiensaChile.com 2021). Until late in the 1980s, Maihue remained ill-suited in terms of roads and means of transport. Most of their traffic was toward Argentina on horseback, a shorter trip than moving to southern Chilean cities. Navigation was the only means to connect peoples from Maihue and Rupumeika and to meet in times of the *nguillatun*. A municipal, poorly maintained motorboat named Santita (Little Saint) provided the service until 2005 when it sank, where 17 people died in the accident (Fig. 16.4). Land ownership, lack of infrastructure, and poorly maintained means of transportation are part of the policies that imprisoned these communities in what used to be the *meika* (medicine woman) route: trails and riverbanks where she used to gather medicinal plants.

The entire scene changed dramatically with the expansion of hydroelectric projects and tourism. The announcement by SN Power Hydroelectric Co. of a project in Maqueo Port, in Maihue Lake in 2007, created a new context for the ritual community which, until then, had the sole purpose of protecting the relationship between the community and water (Environmental Justice Atlas 2019). The community members took a leading position in a mobilization that forced the company to

**Fig. 16.4** La Santita boat disembarking practitioners who attend the "nguillatun". (Source: Photograph by Debbie Guerra (ca. 1999))

abandon its project. The translation of the biocultural memory into political practice and territorial control would continue from then onward, until access was limited to tourists in March 2020, to prevent Covid-19 propagation.

From cultural limnology and forestry perspectives, by protecting their lands and planting native species, Rupumeika and Maihue communities expand the vegetation in territories that, in their absence, would be occupied by invading vacation properties characterized by high degrees of water consumption and pollution (Hidalgo and Zunino 2017). Likewise, from a territorial point of view and due to collective action headed by the ritual community, Mapuche people increase their space of autonomy. In this sense, through their ritual relations and political practices, Mapuche communities tend to recover part of the territories they lost during the Spanish conquest and that they lose to Chilean capitalist expansion. Sustainability and water protection depend on processes that integrate political, sacred, and hydric dimensions.

## *2.4 Waters in the Emergence of Memories and Rights Vindication*

In Chimehuin River (Limay river sub-basin) (Fig. 16.1), since 2000, the necessary evolution of environmental co-management between National Parks Administration and Mapuche Raquithue and Lafquenche communities represents a key fact to achieve more symmetrical relationships between stakeholders, with the aim of preserving biodiversity. Currently, Huechulafquen-Paimun and Epulafquen Lakes, located at the head of the Chimehuin river, make up a dynamic socioecological system where different situations emerge around the management of water spaces and wherein Mapuche communities constantly try to express their local knowledge (Fig. 16.5). In these situations, they respond through community organizational processes, ensuring the reproduction and/or legitimation of their local knowledge. This is a complex cultural framework where territorial relationships take place in a political sphere wherein different interests are disputed, on the one hand the protection of the waters and on the other the production of tourist services based on their consumptive and nonconsumptive use.

This great lake system, along the famous Chimehuin River, has become a well-known tourist destination over time. These landscapes host all kinds of water bodies (e.g., streams, rivers, lakes, natural springs, *mallines*) (Chaps. 9 and 10), with abundant rain and snow during winter. Both the rich diversity of water bodies and sub-Antarctic forest constitute multiple habitats for a wide variety of human and nonhuman inhabitants. In these territories, the forms of water use have changed over time, largely as a result of the appearance of foreign cultural elements to Mapuche communities and the consequent changes in their traditional ways of life. Traditionally the waters of the territory have been used by local people for human and animal consumption, for personal hygiene, and for irrigation of family gardens. Currently waters are also used for touristic activities.

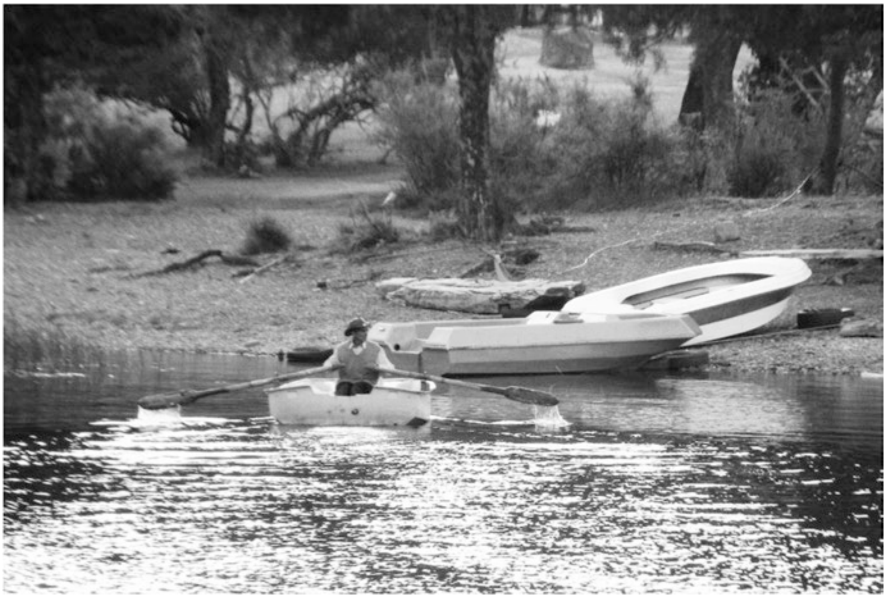

**Fig. 16.5** Mapuche residents crossing by boat at the crossing of the Huechulafquen and Paimun lakes. (Source: Photograph by Juana Aigo)

In their daily lives, people understand that nature, of which they are part of, is alive. Hence, in the face of waters' multiple manifestations, there are signs and signals that guide human behavior. Lakes, streams, rivers, springs, and wetlands are protected by different *ngenko*, so an intersubjective and dialogical emotional bond is created: the person *che* owes respect to water *ko* and to its different *ngenko* (Aigo et al. 2020). As with drinking waters of Costa del Lepá and Gualjaina and sacred waters of Rupumeika, *ngenko* will respond to people according to the way they are treated. People from Raquithue and Lafquenche constantly observe waters' behavior, for example, if the lake wakes up calm or angry, and also its physical and chemical appearance: with what color, smell, taste, and temperature it wakes up.

People from Chimehuin basin have a wide ecological knowledge of these environments, of their different human and nonhuman inhabitants, and of the changes produced in them. The *inai lonko* (Mapuche authority), a woman from Raquithue community, affirms her concern about the increase in heat sensation in summer and about the change of waters' temperatures, which used to be cold in the past and are presently warm. In the same way, people of Raquithue community record in their memory recent changes in waters as a consequence of the latest seismic movements and volcanic eruptions in the area (see Chap. 3). According to the Mapuche word, after the *pillan* (volcano, main water supplier of these environments) has trembled and breathed, the lakes have expressed their anger through sudden movements and forceful changes in water color.

On the other hand, when community members talk about aspects related to land use planning within the basin, local knowledge about Mapuche water geography emerges. The local toponymy based on *mapuzungun* names is revealed through the narratives of the people behind the colonial history of this region. From an intra-community process of decolonization, when people of these communities talk about the hills and the waters, they mention the names imposed on the official maps and those indigenous names that they know well and do not forget. A Mapuche person mentions the true name of one of the hills *futa ufko*, which refers to a waterfall that lives on that hill, different from the name "Cerro de los Ángeles" imposed by the *winka* (foreigners). Likewise, people expose their knowledge to tourists about other water inhabitants, such as native and introduced fish, whom they refer to by their generic name *challwa*.

Mapuche limnological knowledge around the Huechulafquen-Paimun-Epulafquen lake system reflects the intimacy of the bonds of reciprocity, respect, and care between people and waters, as well as the strong social control that exists through customary rules of use and calls for attention or sanctions for those who violate them. Mapuche communities, through orality, keep alive the collective memory of the basin. In particular, the *epew* (stories) constitute one of the traditional forms of transmission of the *kimün* or Mapuche knowledge, from the elderly people to new generations within these communities. In this way, knowledge is translated within the basin into ethical and practical guidelines for decisions regarding use and management of the different water spaces and is enriched from new lived experiences with nature. These experiences are frequently manifested through the observation of changes or signs in nature which are later retransmitted orally in contexts of conversations or *nütram* within communities.

The current co-management within Chimehuin basin is complex and poses daily challenges that require strong roundtables with the participation of Mapuche and non-Mapuche people, private tourism service providers, and members of National Parks Administration, in order to consider different points of view and make decisions about what to do and how in each area of the territory. In this context, each year, through internal assemblies, the communities review, rethink, and agree about behaviors toward the environment, and in dialogue with National Parks' members, they determine tourists' uses in the different water spaces. At present, the most attractive recreational activities enabled for tourism are associated with water; this presents the intense challenge of reaching agreements between parties.

Waterscapes within the Chimehuin basin are undeniably dynamic ecosystems in constant tension. Currently, Raquithué and Lafquenche communities propose management actions considering the Mapuche point of view; people raise the idea of "letting certain environments rest," as opposed to their concession to subject them to tourism exploitation. Among the actions, use and protection of spaces such as thermal springs (hot waters) fundamental for people's health due to the unique therapeutic properties of its waters and medicinal plants are discussed.

From the Mapuche point of view, the communities defend the concept of "letting rest," which implies allowing the *ngenko* to return to their places of origin and rest undisturbed. Through internal consensus, communities oppose new concessions and exploitation by private agents of the thermal springs "*Lawenko* Hot Springs" located around the lakes. In the same way, they claim to reduce the usage of the waters and the consequent impact on them by questioning other concessions that involve sailing with large vessels such as catamarans. In a context of constant tension between Western and Mapuche views, the parties involved in water co-management face the challenge of recognizing and articulating different types of knowledge to arrive at consensual decisions on water care (Aigo et al. 2020).

## 3 Conclusion

The role of water in its material and nonmaterial components of Mapuche memory underscores the importance of understanding and incorporating the cultural dimension in Patagonian limnology. While the diverse situations explored in this chapter highlight the role of water in the making of the Mapuche world, they also suggest that water stands in an intersubjective relation with people. This dialectical relationship explains the emergence of diverse waterscapes in Patagonia (Di Giminiani 2018; Aigo et al. 2020), that is, conceiving water as an agent, Mapuche worldview allows a locally centered view of the waterscape: each site is unique and emerges from the relation that the community establishes with that agent in that place. Hence, in Patagonia, waterscapes vary from region to region.

The testimonies gathered here give evidence about the importance of these waterscapes for water supplying for consumption, food, medicinal plants, and firewood and as habitat providers for terrestrial and aquatic fauna, too. Also riverbanks are the most important and preferred residential and productive areas for horticulture and animal breeding, wherein humidity creates the required microclimate for survival. However, utilitarian access to water occurs in a context of multiple meanings, all of which account for humans' duties and responsibilities concerning waterscapes.

Mapuche communities base their relationship with the hydric world on mutual respect, and they understand that water's behavior reflects human behavior (Aigo and Ladio 2016). Ritual regulations, spiritual beings, and transcendental dimensions speak about a peculiar engagement between humans and waters. Communities translate the biocultural memory through its daily practices in waterscapes' transformations; thus, collective identity incarnates in the environment, molding their self-perception and relationship with it (Skewes et al. 2011, 2012). In this sense, the social practices observed by the Mapuche under the *azmapu* aim at gaining autonomy and biocultural conservation simultaneously (Skewes et al. 2012; Ibarra and Riquelme 2019; Morales et al. 2020).

Faced with a changing world, Mapuche communities confront challenging dilemmas. Either in adopting new technologies or in their political mobilization against external investors, they find themselves confronting undesirable consequences of such transformations. In Costa de Lepá and Gualjaina, the needed investments produce, as a result, a process of growing social disparities, while in Maihue and Rupumeika, if mobilized, the corporations try to gain loyalties by giving away some resources. Differentiation and division are threats pending on the communities. However, communities find a way to avoid this contradiction, by organizing themselves and strengthening their identity and territorial control. This is the case of Raquithue and Lafquenche, wherein the emergence of water memories makes it possible to strengthen community decisions in the political arena of co-management with the State and private corporations, legitimizing ancestral rights over water (Aigo et al. 2020).

Facing various socioenvironmental difficulties, Mapuche limnological knowledge appears diverse and flexible, without losing its identity features. As urbanization and environmental change expand, the need to embrace the biocultural dimension in water management is growing. It is increasingly recognized that the inclusion of a diversity of worldviews on the relationships between people and the environment is necessary toward a more sustainable and fair world (see Chap. 18). However, establishing bridges for dialogue between knowledge systems seems not an easy task. Conceptual differences and lack of disposition to listen to one another are part of the barriers that need to be overcome.

In this line, it is important to recognize that limnology and Mapuche traditional knowledge are parallel ways of understanding the waterscape: the former deals with biological, chemical, and physical features of lakes and other freshwater bodies, and the latter speaks about the life of water. Yet, the physicality of freshwater eludes the community's attention, and its spirituality is equally elusive to the scientists' eyes. Cultural limnology is a crossroad enlightening both the community and the scientists' comprehension of the sociobiological processes concerning the waterscape. The early dismissal of either of these views has undesirable consequences. People's attitude toward water, for example, secures its careful use and subsequent good quality. In a period of hydric crisis, the generalization of such an attitude is a challenge for society.

In the construction of a new space for this dialogic encounter, it will be necessary to find ways to exchange ideas and language understandable to all. In other words, decolonized forms of research that put the multiple epistemologies on an equal footing (Delgado and Escóbar 2006). In this new discourse, the condition will be the full recognition of biocultural diversity and therefore the organic integration of methods and categories from different environmental perspectives (Oelschlaeger and Rozzi 1998; Ibarra and Pizarro 2016; Estévez et al. 2010). The transdisciplinary approach of cultural limnology can help not only to understand the dynamics between culture and waters but also to discuss other ways of dealing with contemporary socioenvironmental problems and promote an atmosphere of equality and mutual respect between ways of seeing waterscapes.

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# Chapter 17
# Political Ecology, Water Valuations and Feasibility of Water Law Deliberation in the Province of Tierra del Fuego, AIAS (1993–2016)

**Karina Giomi, Alejandro Schweitzer, and Adriana Urciuolo**

## 1 Introduction

This chapter explicates processes of social appropriation of water in Patagonia, specifically in the context of the Province of Tierra del Fuego, Antártida e Islas del Atlántico Sur (Tierra del Fuego, AIAS), in Argentina, bonded with its historical-spatial configuration and the uses of the territory. In Sect. 2, theoretical tools of Political Ecology are introduced, particularly through contributions made from Ibero-America, around the notion of languages of valuation of nature to result in the valuations of water and its relation with economic, social, and environmental interests and by the classification between public, private, and common goods. This section highlights the notion of incommensurability between valuations and the tension between legal significance versus sociopolitical and cultural significances of water valuations.

In Sect. 3, we present a case study that details the conflict between water valuations, regarding the obstacles that hampered the legislative treatment of successive Water Law projects between 1993 and 2016 in the Province of Tierra del Fuego,

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AIAS, Argentina. According to Law No 13,640, each time a project is presented to the Provincial Legislature up to 2 years may elapse for its treatment; otherwise, the project expires and must be presented again. Over the aforementioned period, 13 Water Law projects were presented and lost their validity without having been treated by the Provincial Legislature. However, focusing the analysis on this institution was not enough. The difficulty to achieve a Water Law should be understood from the viewpoint of the effective practices on water in the province and, in this way, made visible in terms of an environmental conflict and a debate between water valuations. The political feasibility of a project includes not only a formal feasibility, that is, based on institutional conditions, but also a contingent feasibility, which involves territorial processes of building legitimacy (Dussel 2006). To capture these two dimensions, the work articulated the political-institutional level with the political-cultural and territorial level (Tapia 2008). In this way, the environmental dimension of the conflict was analyzed, not only its legal aspect. The methodology follows a qualitative research design, guided by criteria of an environmental rationality in the construction of knowledge and included analysis of documentary sources, such as regulations and reports and interviews with social, institutional, and private actors. Also, in this section the study case is contextualized through the presentation of the overall characteristics of Water in the Province of Tierra del Fuego, AIAS.

Section 4 describes the main activities that involve water and land uses in the provincial watersheds, indicating actors, predominant interests, and their diverse environmental effects. This contributes to show the main conflicts between actors due to the lack of specific ruling between 1993 and 2016. Furthermore, we analyze valuations of water in relationship to Water Law in Tierra del Fuego, AIAS, and the defense of water as a public good, while Sect. 5, presents a timeline of the process and the main contributions of Law No. 1,126 on Integrated Management of Water Resources. Finally, in Sect. 6, the conclusions take up theoretical considerations in relation to the debate of the valuations promoted by various actors in the studied case. This allows the resignification of political representation as a space for multiple mediations and negotiations around water as a public and common good. The social construction of these valuations has been fundamental, on the one hand, to counteract the predominance of economic interests and the imbalance that this implies for water regulation and, on the other, to achieve the approval of Law No. 1,126 on the Integrated Management of Water Resources, in 2016, and its regulation in 2021.

## 2 Theoretical Framework

### *2.1 Political Ecology and Valuations of Water*

Political Ecology constitutes a critical transdisciplinary field of increasing relevance in the investigation of the co-production or intrinsic relationships between society and nature. Internationally, the perspective of Moore (2020) has gained influence through the category of *world ecology*. This methodological approach consists in unifying in single analytical framework the tension between the *logic of capital* and the *history of capitalism.*[1]

The commodification and appropriation of nature[2] has been made possible by a simplification of value, based on a false assumption: "nature can be reduced to an interchangeable part" (Moore 2020) and from a link with a homogeneous and abstract production of time and space bonded to the symbolic and scientific order of the modern European world of the fifteenth and sixteenth centuries. That order fixed nature as exteriority: "never before has Nature as an external object become the organizing principle of a civilization" (Moore 2020). Capitalist civilization could only unfold because it represented nature as external. To disarm this assumption, Moore deploys the relational thinking of *world ecology* through the category of *historical nature (oikeios)* (Moore 2020). The problem of historical nature must be addressed in its many levels of time and space, understanding the historical change from the active co-production between human beings and nature and in a double internalization: the society in the nature and the nature in the society.

Successive times of capitalist development are ruled by *world ecology regimes* that establish relations and norms of reproduction, at the same time including "symbolic, cultural, and scientific processes that are central to the re-elaboration of *oikeios* from modernity" (Moore 2020). This framework is very relevant to the current conflict around water valuations, which cannot be understood in a dehistoricized, decontextualized, or fragmented way. A relational thinking is revealed, which could be considered part of a contemporary movement that includes great theoretical efforts in various latitudes.[3]

In this sense, among numerous studies and authors in Latin America, a *common grammar* has been developing from an *eco-territorial turn* in the production of knowledge (Svampa and Viale 2014), which is expressed in a plurality of languages, alternative to the hegemonic ones, which allow the articulation of various crises in a relational way: *development crisis*, *crisis of scientific-instrumental rationality*, *global environmental crisis*, and *Western civilizational crisis* in the framework of transnationalized capitalism (Porto-Gonçalves 2006; Leff 2008). Therefore, the notion of social appropriation of nature, as used here, aims to emphasize, based on an analytical cut, the notion of languages of valuation of nature (Martínez Alier

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[1] The logic of capital allows to incorporate the theory of the value form and to understand in abstract terms the generation of surplus value, and at the same time, the historical geography of capitalism allows to pose in a broader project the relations of value to "unify the history of capitals, natures and social class struggles as mutual relational movements in the modern world-system (Moore 2020)." According to Moore (2020), socially necessary unpaid work time includes both human and extra-human natures: "Such unpaid work could be provided by human natures - women or slaves, for example - or extra-human, such as forests, lands or rivers." So, the historical condition for socially necessary work time is socially necessary unpaid work. Thus, nature is constitutive and active in the accumulation of abstract social labor.

[2] For more references on Moore's categories and valuations of nature, see Schweitzer (2020).

[3] See Palacio (2006) for an overview of the field of Political Ecology and Latin American Political Ecology.

2004, 2006, 2009) in terms of discourses of knowledge-power constitutive or co-producers of nature and society. In this way, a difference is introduced between debating the ways of establishing a certain value within the same valuation system and the very different question of continuing to debate between different valuation systems. The first case includes the debates between those who, in different ways, seek to measure or quantify nature by reducing it to a market value. This presupposes an instrumental scientific rationality scheme overlaid with assumptions of an instrumental economic rationality: valuation (by a rational actor) is defined as an expression of a value or measure, established by scientific observation, in which an action or object (means) contributes to the achievement of objectives established in a valuation system (purpose) and, in that sense, acquires relative importance compared to others (Penna and Cristeche 2008). In this case, the evaluative debate on nature takes place on instrumental assumptions that can even underlie certain environmentalist discourses, since, according to Eduardo Galeano (1997), even "its executioners" judge as necessary to protect nature. In this perspective, scientific knowledge intends to subsume other types of valuation languages.

On the contrary, in the framework of Political Ecology, the category of languages of the valuation of nature allows us to capture other forms of valuations that respond to ethical-political positions and lead to processes of competing territorialities (Valiente and Schweitzer 2016). They introduce qualitative logics, which cannot be quantified, for example, cultural, philosophical, ethical, ethnic, rights of nature, among others.[4] Scientific knowledge is one source of knowledge among others. In this case, the valuation languages may not be translatable, which means, they may be incommensurable[5] with each other, and in this sense, any prioritization of one valuation over others will imply power relations.[6] The following question stands out: "Who has the power to simplify complexity, discarding some valuation languages

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[4] From the analyses of several cases, Martínez Alier (2006) enumerates *pieces of vocabulary from the resistance* in the face of environmental conflicts linked to different *valuation languages* such as: (a) indigenous ecological language, (b) demands for "a tax on the depletion of a natural resource," (c) business responsibility demand, (d) ecological debt from the North to the South, (e) language of food security and food sovereignty, (f) racism environmental language, (g) environmental justice language, (h) space-environmental language such as environmental inequalities, (i) ecological track, (j) eco-feminism, and (k) environmentalism of the poor or popular environmentalism. This list arises from situated studies and shows how, in Tierra del Fuego Province, the defense of water as a *public good* cannot be merely understood in its legal meaning but as the valuation language that acted as the language of the resistance facing the privatization discourse on water.

[5] Leff (2003) upholds that political ecology is established "there, wherein nature as well as culture resist the homologation of immeasurable values and processes (symbolic, ecological, epistemological, political) to be absorbed in terms of market values."

[6] For a broad analysis of the difference between processes of (economic) valuation and valuation of nature, the work of Schweitzer (2014), Valiente, and Schweitzer (2016) is recommended and also for a debate on development models or alternatives to the development (Gudynas 2010; Schweitzer 2014; Escobar 2005, 2012; Lang et al. 2013; Svampa 2016).

and imposing others?" (Martínez Alier 2006). This highlights that, for decision-making in environmental conflicts, valuations are mediated by a political debate:

> Power stands out here on two different planes. The first is the ability to impose a decision on others, for example, to expropriate or steal resources, to set up a polluting factory, to destroy a forest or to occupy a space to dump waste. The second is the "procedural power", that is, the force to impose a method of decision on the conflict, a language that excludes and triumphs over other languages of valuation. This happens when the cost-benefit analysis is imposed, or the decision is subordinated only to the results of an environmental impact study as the final criterion (Martínez Alier 2006).

Then, a valuation system is imposed over another by power relations, a question that leads to emphasizing political debates. These debates cannot be analyzed only on the formal legal plane but require an analysis of the multiple mediations involved in socioterritorial dynamics that imply negotiations, relations of differential forces, and hegemonic logics. In this sense, the notion of *environmental justice* (the right to a safe, healthy, and productive environment for all) has revealed that environmental risks have an unequal social and geographical redistribution, based on antagonistic and differential power relations, thus overlapping poverty maps with those of environmental degradation ("ecology of the poor") (Martínez Alier 2004; Porto-Gonçalves 2006; Svampa and Viale, 2014). In this framework, the Political Ecology of Water in Latin America (Isch 2011; Merlinsky 2013a; Ávila-García 2015) integrates the notion of *water justice* since "water justice is just a part of environmental justice and social justice" (Isch 2011). Environmental justice can involve demands for access, distribution, recognition, representation, or participation (Fraser 2008; Merlisnsky 2013a) in relation to the environment; however, it is not necessarily associated with struggles for recognition (Martínez Alier 2006), as on many occasions it transcends particularities and becomes an egalitarian claim: "the environmental justice movement appeals to egalitarian principles, of a communitarian type, by claiming for an equitable distribution of environmental advantages and damages" (Merlinsky 2013a). Then, the notion of environmental justice, as part of the integral approach of Political Ecology, is not reduced to a classical legal-contractual vision:

> Environmental justice is not solved in the judicial disputes that may arise as a result of socio-environmental conflicts. It is, on the contrary, a questioning of development models and the roles played by each of the social actors involved, and, therefore, the fundamental scenario of resolution is linked to power, in whose hands it is found and in whose favor it is used (Isch 2011)

In this sense, valuations of water linked to economic, social, and environmental interests stand out, while at the same time the distinctions between private, public, and common goods are highlighted. The analysis of the possible existing classifications is not comprehensive; therefore, only the classifications relevant for this case study are presented.

### 2.2 The Valuations of Nature and Water from the Classification Between Public, Private, and Common Goods: Legal Meanings and Sociopolitical Meanings

The modern Western tradition of thought installed the vision of nature as the exteriority of the social. On the contrary, within the framework of Latin American thoughts in recent decades, the vision of nature as a political issue has been linked to the problematization of the concept of the public. The idea of *the public* varies according to the sociohistorical and territorial contexts. In a general way, it is opposed to *the private*; however, this dichotomy is also a construction: “what at one time was referred to the private due to a hegemonic social order (…) it is perfectly feasible to be taken to the public arena when it is processed in a political way” (Retamozo 2006). This has happened with the notion of nature in Latin American countries.

During the 1950s, the hegemonic notion of economic goods rested on the assumption that goods are classified according to intrinsic or natural properties: pure private goods are excludable and competitive (or rivalrous). Excludable meaning that whoever owns it, or exercises the property right, can prevent access to the person who has not paid the required price. Competitive meaning that the use by one individual decreases the amount of the good available for others. In the aforementioned approach, most goods must be considered private, regarding the market as the normal resource allocator. In this sense, public goods are negatively defined as those goods in which the market rules are not met (i.e., no one can be excluded from their consumption, and consumption by one individual does not reduce that of others, so they are not excludable or competitive). Given the existence of a public good, intervention by the State or by social organizations as producers of such goods would be justified, although later, neoliberal theorists questioned the assumption of state intervention as a necessary consequence (Laval and Dardot 2015). Social theory thinkers Laval and Dardot (2015) give as an example of the above the European Union’s doctrine that replaced the notion of public good “preferred to speak of ‘service of general interest,” leaving room for private production subject to a set of conditions established by the public authorities. In this case, the authors’ opinion is that property rights left the door open to granting favorable guarantees to privatization policies and commodification inherent to neoliberalism.

In opposition to the assumptions of neoliberalism, in different continents, there has been a review of the dichotomies between public and private, through the category of commons.[7] For example, the public-private dichotomy was revised in the case of nature. From the position of an economic neo-institutionalism, the American political scientist Elinor Ostrom (2000), Nobel Prize in Economics (2009), stated a

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[7] It should be clarified that the concept of commons acquires different meanings in different geographical and historical contexts and in different topics, for example, intellectual property, patents, and genes (Helfrich 2008). In Latin America, the concept is constructed and acquires different meanings around valuations of nature.

**Table 17.1** Public, private, and common goods according to E. Ostrom

| Classic dichotomy | | | Nature, special case of public good: “common goods” (not commodities) |
|---|---|---|---|
| Goods use | Private | Public | Common |
| Excludable | Yes | No | Not excludable: everyone is granted access |
| Competitive | Yes | No | Yes competitive: the consumption of one limits the other |

Source: own elaboration adapted from Laval and Dardot (2015)

particular type of goods[8] that she calls *common goods* (commons), as indicated in Table 17.1. Common goods, like public goods, are not exclusive, but they are competitive like private goods, so they have special characteristics. This has influenced views on water rights (Pacheco-Vega 2014). However, according to Laval and Dardot (2015), the way of understanding the commons proposed by Ostrom remains within economic rationality, without proposing an alternative rationality: “The construction of the commons is imposed in certain particular situations, for certain specific goods, without questioning at all the rationality of markets or the State” (Laval and Dardot 2015). While it is true that Ostrom posits social cooperation in the management of the commons, using contributions from game theory to reformulate the notion of the rational individual of hegemonic neoliberal economics, she does not make a break with that framework.[9] Thus, this theory does not account for power relations and the phenomena of exploitation; it omits to consider that institutions are embedded in global capitalism and in local communities with their own histories and dynamics.

In conclusion, for Laval and Dardot (2015), Ostrom’s vision “prevents us from reflecting on the possible political constitution of the common as a generalized alternative rationality” (Laval and Dardot 2015). Precisely, that will be the authors’ wager. They do not start from an essentialist vision that understands *the common* as a *property* of goods; on the contrary, *the common* is the effect of *practices*: “only practical activity can make things become common, just as only this practical activity can produce a new collective subject” (Laval and Dardot 2015). The common appears as a political principle, that is, it constitutes the possibility of proposing an alternative (or incommensurable) rationality to hegemonic economism: “The common is not a good...It is the political principle from which we must build commons...it is the political principle that defines a new regime of struggles on a global scale” (Laval and Dardot 2015).

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[8] In the classification of economic goods by Ostrom there is a fourth type: *club goods*, which are excludable but not competitive and in which a price can be established and the individual consumption does not reduce that of others.

[9] For Laval and Dardot (2015), the limits of Ostrom’s position lie in the categories she employs, since she continues to associate the concept of common with a property of goods and because the thesis of institutional diversity continues to be founded on rational individuals understood as selfish agents who, without government coercion, make decisions on the most convenient contractual arrangements based on comparing costs and benefits.

In Latin America, from the 1970s—characterized by the crisis of the Welfare State and the consolidation of neoliberal theories—to the present, various currents of Political Ecology, social movements, and other organizations, within the framework of the global environmental crisis, have sustained a strong criticism of economic and extractivist development models. They have questioned the hegemonic neoliberal perspective that regards the market as the most efficient way of allocating resources that should operate freely without State intervention. The critical problematization of these assumptions was emphasized, mainly, after the external debt crisis occurred in Latin American countries due to the privatization and deregulation policies consolidated during the 1990s. The struggles for the common acquire particular relevance around environmental issues. In fact, Latin American struggles against extractivism, privatization, and dispossession of nature and water are advancing. Consequently, it is possible to speak of a common grammar or eco-territorial turn in the production of knowledge including approaches such as the Latin American Environmental Thought, Modernity-Coloniality Program, Environmental History, and Andean Philosophy, among others. Ávila-García (2015) argues that accumulation by dispossession (Harvey 2004) is a logic in force in current capitalism, which presents new mechanisms such as "the commodification of nature in all its forms; the affectation of goods, until now common, that make up the global environment (land, water, air); and the degradation of the natural habitat" (Ávila-García 2015). In this framework, the author distinguishes the role of the State in its visible and invisible form. In the first, the State appears as a legal framework under the argument of the harmony of interests. In the second, the State contributes to strengthening the interests of elites and dominant groups through the advancement of legal changes that promote trade agreements, the opening of foreign investment, the implementation of infrastructure megaprojects, among others, "of great interest to capital, claiming that they are of public utility and national interest" (Ávila-García 2015). Alternatively, over the last decades, studies indicating the need for an appropriate intervention by national states and/or supranational bodies (Petrella 2008) in water management have increased. The abovementioned conclusions have been reflected in diverse global actions, for example, with the existence of different antecedents, the United Nations (2003) declared the period 2005–2015 as an international decade of actions for "Water for Life." On the other hand, at the beginning of this century, the need for participation in water regulation was stressed by various organizations, associations, communities, and social movements.[10] Since 2000, in Latin America, the so-called Water War in Cochabamba, Bolivia, made visible a global debate on the risk of privatization of the resource. This provoked a struggle of different Latin American social movements, and also in other countries, until in 2010, the United Nations General Assembly, through Resolution 64/292 (United Nations 2010), finally recognized "drinking water and

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[10] Martínez Alier (2006) highlights the "International Rivers Network" as a source of information on water-related conflicts. Additionally, The Latin American Water Tribunal has registered several cases of hydrological injustice, see: https://tragua.com/

basic sanitation" as an "essential human right."[11] Furthermore, an international alliance for research, training, and political action was launched, called *Water Justice Alliance* which is devoted to comparative and interdisciplinary research about mechanisms of accumulation and conflicts around water. Its objective is to reach democratic water policies to generate practices for the sustainable development as well as the equitable distribution of water (Boelens et al. 2011; Arroyo and Boelens 2013).[12] Therefore, in general, it is possible to observe the conceptual distancing from the Dublin principles or, at least, the broadening of their perspectives, by highlighting that water is a natural, vital, irreplaceable, common, and essential good for life (Boff 2014). In accordance with the above, there is a strong theoretical debate regarding the views on water as a public good and as an economic good, a debate that acquires increasing social relevance (Vega López 2016). Likewise, in the synthesis of Vega López (2016), a relationship is observed between the visions of water and different interests, economic, social, and environmental (Table 17.2).

In the case of water, in the 1990s, the valuations of water were mediated on a global scale by the hegemonic vision of the so-called "Dublin Declaration on Water and Sustainable Development" or "Dublin Principles," established in 1992 during

**Table 17.2** Three visions on water: economic good, human right, and ecosystem element

| Water is a… | Economic good | Human right | Key ecosystem element |
|---|---|---|---|
| Access and rights | Water is a commodity, whoever pays have access to it | Water is a universal right | Water is an integrated and fundamental part of an ecosystem |
| Organizations that promote it | World Bank, International Monetary Fund | United Nations, UNESCO | United Nations Environment Program (UNEP) |
| Main elements of governance of the resource | Market instruments, commercial transactions | Supply by local governments in accordance with legislation | Global management plans that integrate a vision of sustainability in the region |
| Generated problems | Privatization | Inequality | Deterioration of ecosystems |

Source: Vega López (2016), based in part on the classification of Soares and Vargas (2008)

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[11] Boff (2014) affirms that "The initiative was promoted by the president of Bolivia, Evo Morales Ayma and was supported by 35 countries, all from the South of the world, it was approved with great difficulty, by 124 votes in favor, 42 abstentions and no votes against. Rich nations such as the United States, Canada, the United Kingdom, Russia, Japan and the entire European Community were strongly opposed, as a result of the commercial interests that their multinational companies have within the water market." On the other hand, the documents promoted by different social movements were more ambitious and conveyed the idea of a biocentric rationality beyond the anthropocentric vision that would make it possible to declare water as a *universal right* by understanding water as part of the life of the planet: it pertains democratization of access to water, not only for humans but also for all life forms. Water belongs to the universal common good. Therefore, it has to be in public and community hands.

[12] For more information see https://justiciahidrica.org/

the International Conference on Water and the Environment. The declaration established four guiding principles for water management:

> (1) Water is a finite and vulnerable resource, essential for life, for development, and for the environment. (2) Water development and management should be based on a participatory approach involving users, planners, and decision-makers at all levels, making decisions at the lowest possible level that is suitable. (3) Women play a central role in the provision, management, and safeguarding of water. (4) Water has an economic value in all its competitive uses and must be recognized as an economic good (Solanes and González-Villarreal (2001).

Dublin's vision focused on water as an economic good by underscoring two characteristics of private goods: competitive and excludable. On the one hand, it highlighted competitiveness—that is, consumption by some reduces the simultaneous water availability for others—it thus highlighted scarcity. On the other hand, it assumes that since water is a finite vulnerable resource, it should be managed privately. This can be understood within a neoliberal framework that sought and seeks to impose the assumption that the market would achieve a more efficient management in case of "scarce" resources, implicitly introducing for water the condition of excludability. Furthermore, it emphasized the lowest levels of participation in water management which, in a context of strong promotion of privatization models, could be understood as an attempt to minimize the role of the State in the abovementioned management.

The idea that water has an economic value should not be confused (insofar as it is part of a production process, or in terms of the distribution of the resource, so there may be a price assignment) with the idea that water has to be considered a "commodity," which means to understand it as another merchandise, a private good, over which neoliberal perspectives promote market regulation (with the arguable argument of a more efficient management): "Commodification, in this context, refers to the circulation of water as a private good whose exchange value includes a profit that is appropriate by a private agent who holds the property right" (Echaide 2012).

These debates have been brought to the attention in the field of Political Ecology, making more and more visible the relevance of political factors in the analysis of the discourse of scarcity (Cajigas-Rotundo 2007; Schweitzer y Valiente 2018). Leff (2008) agrees with the geographer Porto Gonçalves (2006) in challenging the discourse of water scarcity as a product of human action and related to economic valuation models "it has been necessary to manufacture its scarcity, to subsume it within the logics of the economy" (Leff 2008). Both authors point out the need for other valuations of water from another type of rationality: "The current dispute over the control and management of water stems from the environmental crisis, and also reveals the crisis of hegemonic instrumental rationality in the science of colonial modern society" (Porto Gonçalves 2006).[13]

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[13] "With that, environmental injustice becomes even more widespread,…At the end of the day, water circulates not only through rivers, through the air ... but also under the social form of various goods - fabrics, automobiles, agricultural raw materials and minerals - in general, in the form of

In relation to Latin American countries, some authors (Echaide 2012; Svampa and Viale 2014) define two major paradigms of legal thinking around water (beyond differences in regulation between types and uses of water):

- *Dominial paradigm*: It distinguishes the public domain and the private domain. Nature is understood as a public good or a private good according to the subject of law that has its domain.[14] In Latin America the vision of water as a public good predominates, with the exception of Chile, which adopts the privatization approach. Although this has generated complex situations in the basins shared between Chile and Argentina (Urciuolo et al. 2009), this situation may presently change, since Chile is undergoing a constitutional reform, and a change toward the conception of water as a public good is a paramount demand from various social movements (*Movement for Water and Territories*, 2021).
- *Environmental paradigm:* It is incipient and, at present, minoritarian. The clearest cases are the Constitutions of Ecuador and Bolivia. In Bolivia, the private appropriation of water resources, as well as the concession of the good or its services is prohibited: “This paradigm… understands water –and in some cases all of nature– not as an appropriable resource but as an entity subject to rights” (Echaide 2012).

Among these visions, according to Echaide (2012), global neoliberalism operates as an extralegal context or material source of law, spreading the idea that market management is the most suitable way of solving the water crisis; in other words, it is a story constructed from positions of power. Therefore, a critical assessment of the various perspectives and/or valuation paradigms requires a broader approach to power relations. Increasingly, against the view of extractive economy that regards natural goods as merely economic resources, a blooming perspective that conceives them as *common goods* has been cemented in Latin America: “the notion of common goods is clearly different from that of public good”; “the concept of common goods appears today as one of the keys in the search for an emancipatory paradigm, beyond the Market and the State” (Svampa and Viale 2014). When goods are considered common, neither the individuals nor the States have full authority over them; on the contrary, all the groups and inhabitants of the country have the right to participate in the decisions about their use, destination, exploitation, and/or protection. This supposes a differentiation for critical Latin American thinkers between the *state public* and the *common public*. For example, there is a distinction between *surface of politics* and *subsoil of politics*, in Tapia’s words (2008), or between *potestas* and *potentia*, according to Dussel (2006). In summary, the category of *common goods*, from a legal perspective, is related to the concept of *common heritage* and,

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tangible goods and in this way, we can understand the hydrological imbalance driven by the generalized market logic” (PortoGonçalves 2006).

[14] “It is a public good when they are property of the States and are expressly individualized (territorial seas, rivers, streets, squares, etc.) or are affected to the use and enjoyment by all inhabitants. Private goods are all those that are not public goods” (Svampa and Viale 2014).

from a sociocultural perspective, with the concepts of *networks of daily sociability* and *identity production* (Schweitzer 2014; Svampa and Viale 2014) (Table 17.3).

Table 17.3 shows a tension within the environmental paradigm between the notion of common in the juridical sense and the sociopolitical meaning of community. This is an alternative rationality, which we will try to apply to the analysis of the case study:

> It is not a battle or a choice between public or private...Access to resources generally reflects the interests of those groups that can influence the construction of local, national and international rules about sharing. Therefore, instead of only debating the forms and models of water governance – public, private or community; centralist or decentralized, etc. – it is fundamental to analyze the power structures and operational mechanisms behind water games, and how these materialize in current forms of sharing (Boelens et al. 2011).

Then, the networks of everyday sociability introduce a different way of analyzing the obstruction of the Water Law in Tierra del Fuego, AIAS, between 1993 and

**Table 17.3** Valuations of nature: dominance and environmental paradigms

| Nature valuations | | | |
|---|---|---|---|
| Dominance paradigm<br>The state public | | Environmental paradigm<br>The common public | |
| It involves a statist-productivity matrix<br>A common language about nature is shared: utilitarian conception (unknowing of other valuations that cannot be represented by a market price) | | It involves the notion of common goods<br>Disputes over various languages and valuations of nature<br>Construction of territorialities | |
| State | Market | Citizen participation | Eco territorial logic |
| *Public good*<br>Nature as a public good | *Private good*<br>Nature as a resource | *Common good*<br>Anthropocentric foundation: goods<br>Registry: legal/patrimonial | *Common good*<br>Ontological foundation: community *(ethos pro community)*<br>Registry: networks of daily sociability, local identities |
| It is the domain of the States, and they are expressly individualized goods (seas, rivers, others) or those that are affected to the use and enjoyment of all inhabitants | Natural goods as a strategic resource and commodity: a tradable, undifferentiated good, without greater added value, whose price is defined by the international market | All groups and inhabitants must participate in the decision of its use and/or protection. Need to keep out of the market those resources and goods that, due to their nature of natural, social, and cultural heritage, belong to the community and have a value that exceeds any economic value | In Latin America, it refers to *the common* in intimate relation to natural goods and the dispute over the valuations and forms of construction of territoriality. It is also linked to preexisting communities and to forms of resistance to the accumulation of global capital |

Source: Own elaboration, synthesis of Svampa and Viale (2014)

2016, since in the arguments of the interviewees are embodied, at the same time, visions about Fuegian society, from long-standing power relations, and diverse environmental ecological judgments, understood "as a construction of arguments about society, as complex refractions of all kinds of struggles that have been carried out in other spheres" (Merlinsky 2013a).

# 3 The Imbalance Between Valuations of Water as an Obstacle Factor for a Water Law in the Province of Tierra del Fuego, AIAS

## *3.1 Case Study*

During an extended period without a Water Law, irregular situations of water use that in fact responded to particular interests of different actors were generated. These actors had a vested interest in avoiding the legislative treatment or the sanction of regulations that might compel them with a norm on water use. For this reason, these irregular situations constituted increasing hurdles for the treatment of a regulation that might revert the existing situation in pursuit of the general interest on water. The repeated refusal to treat a Water Law between 1993 and 2016 became a paradigmatic case study (Vasilachis de Gialdino 2006) that could be used to analyze similar cases.[15] Until 2016, 13 Water Law projects were proposed to the Provincial Legislature and successively lost parliamentary status as the deadlines for their treatment expired according to Law No. 13,640. Throughout those years, the Legislature achieved the sanctioning of several laws related to the environment and the regulation of resources; however, contrary to all expectations, the Water Law was simply not treated.

To deal with this difference, the notion of political feasibility was employed (Dussel 2006). This means that there is a process by which a project opens to institutional possibilities, from a legitimate framework. It involves an institutional feasibility (formal) and a contingent feasibility valuation (territorial), related to a process of social construction. In other words, political feasibility implies that the relationship between the *surface of politics* and *subsoil of politics* must be analyzed (Tapia 2008). This way, Political Ecology moves away from the classic vision of liberal politics which focuses on formal representation institutions, so as to show power relationships in the social appropriation of nature. What needs to be understood is that the issue was at risk of being reduced to its legal aspect—the lack of a Water Law—instead of exposing what it actually expresses: an environmental conflict.

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[15] For example, in the Province of Tierra del Fuego, more than a dozen projects have already been presented to declare the Peninsula Mitre as a Protected Area, without being treated until now. To learn about the project: https://www.manekenk.org.ar/wp-content/uploads/2020/10/Peni%CC%81ínsula-Mitre-y-Anexos-Info-Final.pdf

The notion of nature valuation languages (Martínez Alier 2006) means that access to nature by human beings intrinsically implies a historical and discursive appropriation. According to the information above, in order to articulate the formal institutional framework and the territorial processes for water valuations, the problem-question was constructed as follows: Have the water valuations of different social actors had an impact on the feasibility of the treatment of a draft of the Water Law in the Provincial Legislature of the Province of Tierra del Fuego, AIAS (1993–2016)?

The aim of this study was to characterize water valuations by different actors and to show the imbalance between economic, social, and environmental interests. The prevalence of economic over other types of interests constituted an obstacle for the viability of a legislative treatment of a Water Law in Tierra del Fuego, AIAS (1993–2016). In terms of regulating the use of water, the Water Law would allow a balance in pursuit of a common interest, although it would not eliminate the conflict since it is intrinsic to the social aspect.

As for the methodology, based on a qualitative design research, documentary sources were used—management information, Law projects, existing legislation, studies on hydrological resources, and others—in order to identify the activity in the basin and also conflicts related to the use and management of water in the absence of specific rules. In addition, interviews were conducted to actors from different sectors: (a) state: technical area of government, political staff of government, and scientific institutions; (b) social organizations: environmental civil associations; (c) private: oil company, livestock activity in farms, and private service to fishing reserves; (d) Public opinion makers: private journalism. Viewpoints of the actors involved allowed us to identify interests, arguments, and valuations on water in the territory and at the same time to find key events that were used to elaborate the general sociohistorical processes in the case study.

## *3.2 Water in the Province of Tierra del Fuego, AIAS. Overall Characteristics*

At present, Tierra del Fuego, AIAS Province, has three territorial areas: the Eastern sector of the main island, Isla Grande de Tierra del Fuego, and nearby islands (as the western sector belongs to the Republic of Chile); Southern Atlantic Argentinean Islands (Malvinas, Georgias and Sandwich); and Argentinean Antarctica, Orcadas del Sur Islands, and Southern Shetland Islands. Due to geopolitical and historical reasons, the provincial government controls just the first of the mentioned territories (Ministerio de Planificación Federal, Inversión Pública y Servicios 2008). Additionally, and also for geopolitical reasons, based on Law No 1,126 about the Integrated Management of Hydrological Resources from Tierra del Fuego, AIAS, the Province regulates interior water basins and coastal areas on the Eastern side of Isla Grande. Nevertheless,

> It claims as its own the existing water resources on the Malvinas, South Georgia and South Sandwich Islands, islets and adjacent and surrounding archipelagos, the already established maritime spaces of the Argentine Sea and those that, in the area that corresponds to its jurisdiction, have been established by the 40th Commission on the Limits of the Continental Shelf that depends on the United Nations Convention on the Law of the Sea (UNCLOS) until they are established by the relevant international treaties and resolutions as well as the regulations that consequently may dictate the national authorities (Law No. 1,126/2016).

On the Argentinean sector of the Isla Grande de Tierra del Fuego, the water basins have been classified under a hydrological criteria (Iturraspe and Urciuolo 2000; Iturraspe and Urciuolo 2006) connected to the presence of regulating storage, weather conditions, vegetation, and others. According to their hydrological behavior, four water basins have been defined: (a) Northern, steppe water basin; (b) Central zone, transition water basin, (c) Southern, mountain range water basin; and (d) Eastern, peatland water basin (Chap. 9). This information is relevant because the type of watershed determines the feasibility and appropriateness of the different uses of soil and water, as well as the water and environmental criteria to be incorporated in the regulations for their management.

As for the link between the use of the water and the socioproductive structure of the local territory, different moments could be identified. Toward the end of the nineteenth century within the National Territory of Tierra del Fuego (1884–1990), the city of Ushuaia was founded in 1884. It had the functions of a capital city, with services provided by the Army, public administration, and a State prison. Even though the city of Rio Grande was populated as far back as 1883 with the Candelaria Mission and stood as the seat of San Sebastian Department, it was officially founded in 1921. By then, the livestock and wool productive circuit was established in the large farms from the northern zone. The activity was considered of main importance up until 1960, when hydrocarbon activities[16] were consolidated. From 1970 on, and particularly in the mid-1980s, when the "Industrial Promotion" Law No 19640 came into effect, a special taxation and customs system was established that promoted industrialization. That event determined the course of the development in the region toward textile activities, electronic industry, and plastic factories which by 1988 represented a 63% of the provincial gross geographic product. The event caused a new migratory flow[17] in addition to the expansion of the services and commercial sectors.

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[16] At present, many farms are under hydrocarbon exploitation (which puts great pressure on the use of land and groundwater), for which they receive payments for easement costs, making them more profitable as well as less productive. This reflects in the decline in rural employment (Ministerio de Hacienda 2017).

[17] Ministerio de Planificación Federal, Inversión Pública y Servicios (2008). The population of the Province of Tierra del Fuego, AIAS, increased from 13,400 inhabitants in 1970 to 27,000 in 1980, 70,000 in 1991, and 101,079 in 2001.The 2010 National Census indicated there were 127,205 inhabitants, and the 2021 projection, based on data from the 2010 National Institute of Statistics and Census, anticipates 177,697 inhabitants, including 291 from Antarctica (INDEC 2015) The new census is scheduled for 2022.

During the 1990s, several changes were introduced on the territory of Tierra del Fuego. First, the national open market policy weakened industrial growth. Second, provincialization in 1991 resulted in changes in the administrative organization. Finally, the reform of the National Constitution in 1994 assigned the full original domain of all natural resources within their territories (Art. 124). According to this, provinces Sexert complete management of the resources: environmental permissions, control, and police authority in general. Provinces retain all the powers which are not expressly delegated to the Nation.[18]

From those years on, Ushuaia became a tourist destination. Also, as of today hydrocarbon activity has become of great relevance to the province as it represents 74.7% of its main exports. Furthermore, in 2016, Vega Pléyade offshore gas deposit began its working production (according to the Production Report from the Inland Revenue). On the other hand, in recent years, industrial growth has been weakened due to the use of national and international neoliberal policies which have opened importation. From 2016 onward, national and provincial authorities aim at a diversification of the productive array of the island which might imply an even growing pressure on natural resources (see also Chaps. 9 and 10).

The demographic growth of the last decades generated a great demand for urban land for the settlement of industrial, tourist, and service supply activities as well as house building. This has exposed the lack of urban infrastructure and also conflicts around informal settlements (Fank 2019), in particular, the need for projects and works aimed at guaranteeing the adequate quality and quantity of water for human consumption. Overall, an inadequate management of natural resources due to lack of planning (Ministerio de Planificación Federal, Inversión Pública y Servicios 2008), inadequate localization of activities, and incompatibility of land use and water have caused negative environmental effects in the province. Furthermore, there are difficulties for the implementation of protection and conservation policies, added to a deficient management of the current environmental regulation.

## 4 Conflicts and Water Valuation Languages

### *4.1 Actors, Interests, Main Uses of Water in Provincial Hydrological Basins, and Environmental Effects*

Each water basin exhibits specific characteristics and requires detailed studies, and therefore Table 17.4 only presents a general status of water uses in the province.

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[18] The following competences are delegated by the provinces to the Nation: (a) full powers to dictate governmental codes such as Civil Codes, Penal Codes, and Mining Codes, which rule uniformly all through the country. (b) New sorts of concurring powers between both governmental levels on the subject of the environment such as the minimum budget for environmental protection, which are compulsory all through the national territory. The Nation has the power to enact rules which include the minimum budgets of protection without altering the local jurisdictions (Art. 41, National Constitution), while the provinces have the right to pronounce any complementary regulations.

**Table 17.4** Actors, main uses in water basins in the province of Tierra del Fuego, AIAS, and environmental effects

| Actors | Uses and practices in basins | Environmental effects |
|---|---|---|
| Farms | Extensive stock breeding (bovine and ovine sheepherding) | Biological diffused pollution (fecal coliform bacteria) in medium and low sectors in the basins.<br>Degradation of grasslands and wetlands |
| | Livestock on the banks of waterways | Biological pollution of water. Deterioration of salmonid habitat |
| | Livestock in forest areas | Degradation of forest areas due to overgrazing |
| | Draining of wetlands for sheepherding | Degradation and loss of wetlands |
| | Watering of pasture by overflooding, northern farms | Alteration of natural water sources by deviation and flooding. Degrading land effects |
| Farms<br>Fishing Reserve<br>Tourists and inhabitants | Sport Fishing (International Tourism)<br>Clandestine Fishing | Reduction of salmonids populations |
| Farms<br>Oil extraction business | Unauthorized, underground water usage | Soil salinization. Salt water intrusion. Possible alteration of the natural hydrological system |
| | Petrol commercial activity | Groundwater pollution. Possible pollution of groundwater reservoirs and land subsidence |
| Mining firms | Draining of wetland for peat extraction | Degradation and disappearance of peatland areas |
| | Quarries and material removal | Rising level of turbid water levels. Effects on Salmonidae environment |
| Fruit and horticulture productions | Watering for horticulture | Diffused pollution |
| General population<br>Local authorities<br>State agents | Housing developments on floodplains. Nearby areas under risk of flooding | Reduction of water drainage tracts during extraordinary events. Faulty drainage system |
| Population in general.<br>Urgent housing needs | Urban activities. Unloading of untested residual water. Filling of bank edges and use for housing developments<br>Removal of solid waste | Bacterial and organic pollution. Obstruction of natural drainages causing risk of flooding<br>Loss of natural oceanic habitats<br>Alteration of natural river banks. Negative effects on Salmonidae community. General deterioration of drinkable water |

(continued)

**Table 17.4** (continued)

| Actors | Uses and practices in basins | Environmental effects |
|---|---|---|
| Industries | Industrial Activity<br>Unloading of residual material on urban tracts<br>Waste disposal | Bacterial and organic pollution<br>Obstruction of natural water drainages<br>Alteration of riverbanks<br>Pollution and alteration of natural environments affecting birds and Salmonidae populations |
| | Industrial agrochemical uses | Chemical pollution |
| | Aquaculture | Deviation of natural watercourses |
| | Water extraction for bottling | |
| | Forest exploitation. Peatland exploitation | Alterations in hydric systems of the basins. Loss of capacity of hydric regulation. Soil degradation |
| Travel agents and tourists, government | Tourism<br>Insufficient controls over the activity | Negative effects of anthropic activity on soils and wetlands. Different sorts of pollution |
| General public | Recreation | Negative effects of anthropic activity on soil and wetlands |
| Governmental Organizations, Hydric Resources Direction Environmental NGOs, State Scientific Institutions, Community | Habitat<br>Existence and legacy values | Actions to foster control, regulation, and protection of natural water resources |

Source: Adapted from Urciuolo (2010)

Our approach allowed us to identify actors and conflicting environmental, economic, and social interests which also express distinct valuations about water. Activities with economic interests predominate, and the rural sector of the farms is one of the main water users. Water issues are linked to large estates resulting from the social configuration of the Southern Patagonia space[19]:

> Territorialising dynamics are expressed in permanent processes of spatial adjustment spanning multiple dimensions: infrastructure adjustments, equipment, changes in regulatory frameworks, weakening of State control, dispossession of populations and communities, and land grabbing practices. This way, ecological-distributive conflicts – i.e. disputes over access to water, land, soil and clean air among others- emerge generating control mechanisms, disciplinary measures and repression of resistance groups. These are social conflicts inasmuch social relationships are interweaved from a material basis. They are conceived and have consequences in the alteration of the social-metabolic processes of capital (Schweitzer 2020).

The environmental effects derived from the different activities show incompatibility between uses, as well as the increasing pressure on water and the need for regulation (Fig. 17.1).

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[19] For a comprehensive analysis of the social production of space in Southern Patagonia from the notion of frontiers of expansion of capital, see Schweitzer (2011, 2014, 2016).

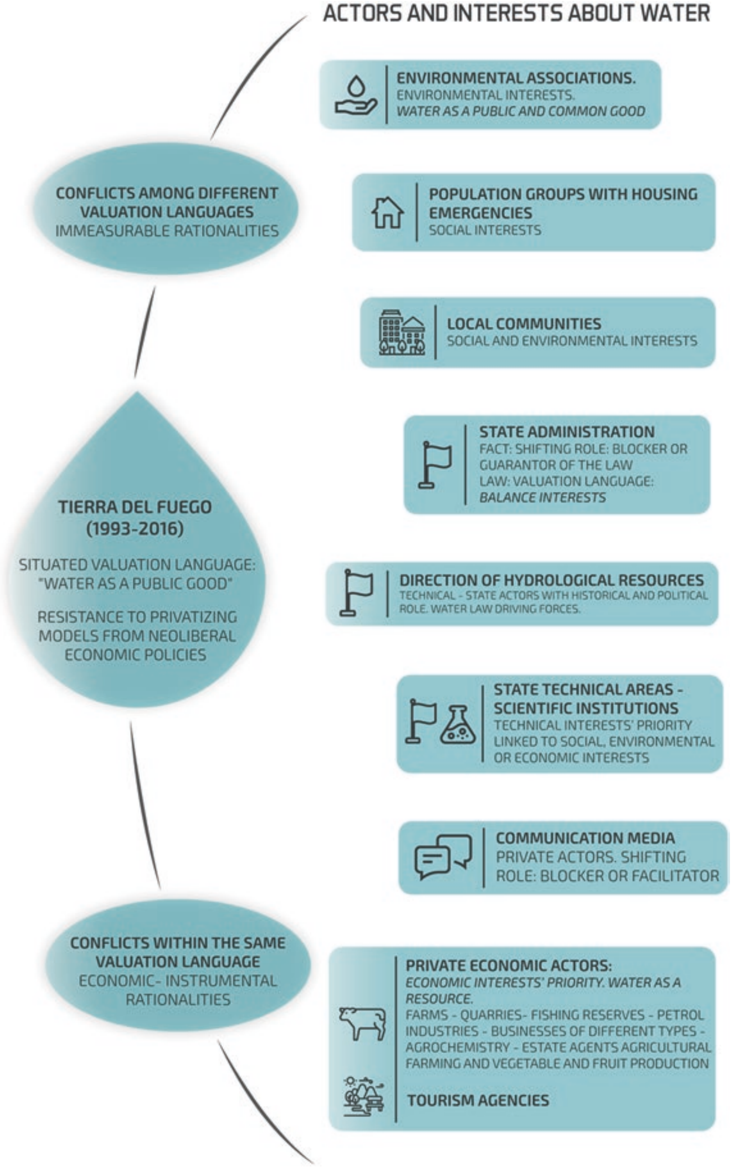

**Fig. 17.1** Water Law and water valuations in networks of daily sociability

## 4.2 Conflicts Due to the Lack of Specific Ruling Between 1993 and 2016

Environmental conflicts due to overlapping of several of the activities and interests connected to water uses in the Fuegian basins were aggravated by the failures in regulation during the 1993–2016 period (Urciuolo et al. 2017). These are commented below, from the perspective of the actors.

- *Conflicts Linked to Management*
- Most disagreements were initiated by struggles for water and the disorganization of shared usage of fountains. The lack of an enforcement authority for a Water Law caused other laws such as land, fishing, protected areas, ports, among others to partially regulate situations related to water management, although they lacked the appropriate technical criteria. At the same time, this obstructed users' participation mechanisms which brought about distrust from users on the ability of the public sector to deal with the issue. Some examples follow:
  - The construction of works in watercourses for different uses, without regulation, approval, registration, and/or authorization—that is, without granting permits and concessions. The actors, mostly private, involved in preexisting irregularities, had no interest in regulation enforcement.
  - Clandestine hydraulic works constructed in watercourses shared by several users without appropriate techniques, such as channels for irrigation of thousands of hectares of pastures, in some ranches in the Northern Zone.
  - It is impossible to encourage user participation in water management through the creation of consortia, basin organizations, and user organizations without a regulatory framework. An attempt to create user organizations for the Chico River in 2009 brought together ranchers against a potential Chinese petrochemical investment that would have required large volumes of water. However, despite many meetings and workshops held by the Water Resources Directorate, which promoted the law, most of the rural actors were reluctant to disclose their own water works, and the organization did not prosper.

  In relation to management in particular, the private sector argued self-efficacy and efficiency to the detriment of the public sector. Most private stakeholders accused the State of inefficiency in the implementation of controls, both in relation to its own fiscal assets and in its eagerness to control the private sector. In fact, rural actors asserted in their arguments a deep mistrust of the State's regulatory capacity and a concern about potential water fee payment. Furthermore, on grounds of sustainable production, private stakeholders highlighted the need to take care of the resource, thus subordinating environmental interests to economic ones. At the same time, they demanded that the State promote production, which eventually translated into some positive arguments regarding the State's capacity.

Actors from state sectors pointed out the historical character of rural actors as an economic, social, and political elite, which received long run benefits from the State, while being reticent about its regulatory capacity. Public actors also pointed out that for decades, both ranches and oil industries have extracted nutrients and resources from the land without making any contribution to society proportional to the returns obtained. Moreover, they argued that many farmers receiving easement from the oil industry, either divested in economically productive activities or reduced their activities in agriculture and livestock.

Provincialization was seen by the interviewees as a turning point, since it implied the strengthening of state control in the province. Actors from the Directorate of Water Resources stressed the need for workshops with stakeholders in order to provide information on both technical aspects and current regulations—as many environmental debates involved the knowledge and enforcement of current laws—in addition to the importance of a Water Law. Among others, these actors made reports and used the media to publicize the defense of water as a public good.

- *Conflicts Linked to the Conservation of Water Basins and the Protection of the Resource*

  These conflicts affected the current and future quantity and quality of available water. There are several activities in hydric basins in the province that threaten the conservation of wetlands such as peatland draining for mining extraction and the draining of meadows for cattle raising, which is a usual practice at some steppe farms. Considering peatlands, as peat is included in the Mining Code as a second category mineral, the lack of a water authority has hindered integrated management between water and mining authorities, causing mining interests to prevail (Iturraspe et al. 2021).
- *Conflicts Linked to Hydric Regulation and Codes*
- Among the conflicts many situations involved incompatible uses of water.[20] For example:

  - Use of water for aggregate washing and aquaculture downstream are incompatible. Aggregate washing in quarries located in areas of the Atlantic Coast Nature Reserve, a RAMSAR site, and hemispheric importance site of the Western Hemispheric Shorebird Reservation Network (WHSRN) (Finck and Giomi 2019).
  - Discharge of untreated effluents by informal housing settlements in nearby water sources for human consumption. The sanitation area had to potabilize a stream that until then was pristine, as recent monitoring indicated biological contamination. This required new water treatment plants which in turn

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[20] In relation to incompatible uses of water, beyond the time period of the case study, there is a growing presence of environmental groups in socio-environmental conflicts, which promotes the emergence of new languages for the valuation of nature. Likewise, although yet without a direct pronouncement on the Water Law, in recent conflicts, the participation of the Yagán indigenous people explicitly joined recent environmental struggle; for example, in rejecting the installation of salmon farms in the Beagle Channel. See: https://www.facebook.com/NoSalmoneras/

  required pumping water. In cold areas, the pumps break, causing water cutoff problems.

- Other conflicts have been identified as sources of resistance to the Water Law, most of them involving private sectors that for many years ran uncontrolled by state regulations.

  - One of the main conflicts due to lack of regulation involves the difficulty of granting access to public water on private lands. Some rivers and lakes have been enclosed to prevent citizens from the water. This is worse in situations involving a fishing reserve (Fishing Law No. 244). In these cases, local fishers were not allowed to access the water thus creating a historic problem in the province. The emblematic case of the Rio Grande was analyzed in detail by Urciuolo and Lesta (Urciuolo et al. 2009). Agreements were reached with landowners through a process of dialogue with local fishermen; however, the formalization of easements required a Water Law.
  - Preexisting situations of works, constructions, etc., in areas of public domain carried out using non-hydrological criteria are in an irregular situation, since the public domain is inalienable and imprescriptible. This has led to conflicts in areas of the provincial cadastre with private urbanization and real estate projects that were advancing with works on the public domain. Furthermore, new housing settlements have been advancing on the area through land filling, rendering necessary to establish and fix several lines with hydrological criteria: the river bank line (delimiting public and private domain), flood evacuation routes, and flood risk zones, in order to efficiently control the occupation of watercourses and surrounding areas.
  - Use of water for secondary recovery by oil companies, without specific regulations governing all aspects of its use: exploration and exploitation of groundwater for use by companies, agricultural settlements, and housing, without regulations for recovery and knowledge of the state of the aquifers, legal guarantees for public water users (producers, investors, others). In the absence of a concession, it was not possible to promote the productive use of water, since users were not guaranteed water availability in quantity and quality in the long term, nor the compatibility of uses in the future. The lack of a concession made it difficult to obtain productive credits, at the national level, or other investments of international capital that required formal authorization. The transitional provisions that were achieved in 2004 attempted to address these difficulties.
  - Need for a registry of concessioned water users in order to control the volumes consumed or the way water is used and to guarantee compatibility of uses in order to protect the resource. Water was used without authorization, without knowing the amount of water actually able to use, or the rules for the construction of works to be respected.
  - Need to agree on water rights with Chile. As for international basins, the absence of concessions meant that, in the negotiations carried out at the

  Foreign Ministry level, the rights granted in Chile (upstream) prevailed over those of Argentinean users, who only had access to 2-year provisional permits, as of the transitory regulations of 2004.
- The lack of a contravention system resulted in inadequate uses of water not being penalized. Need to charge for public water to large users. Although some of the uses were contemplated in the Tariff Law; the lack of a concession mechanism prevented charging for public water use (except for the oil industry use, at much lower rates than other provinces in Argentina).

In general, the actors of the Water Resources Directorate stand out not only for their technical capacity but also for their political action as historical promoters of the Water Law. For example, as a result of some transitory provisions achieved in 2004:

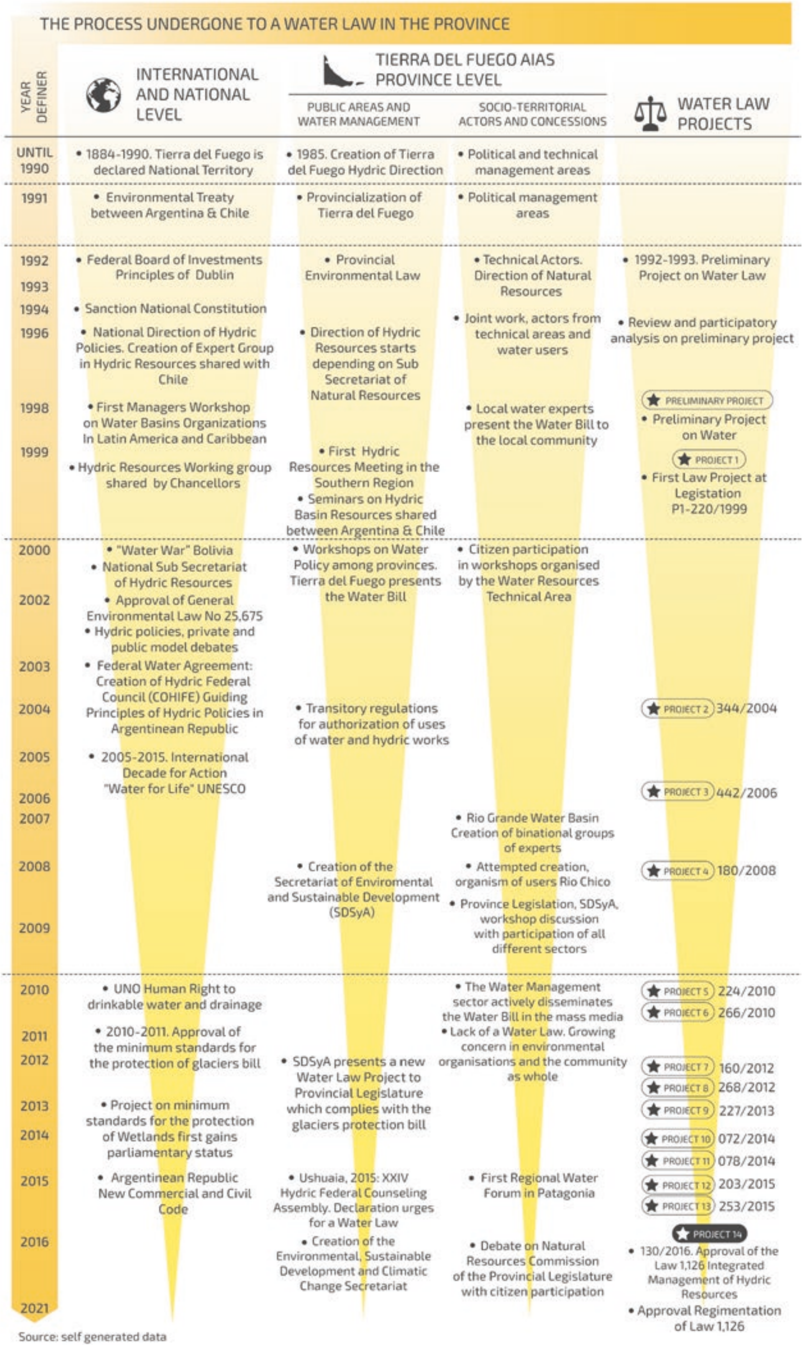
> We celebrated, for the first time we could regulate something, we had something in our hands to go and tell the users, for example, open this gate because this is not yours, you do not have anything granted that allows you to close this gate to use this water for yourself. In other words, we even went around the fields putting resolutions, saying, this gate must be open. It is very complicated, on the one hand to guarantee access to the river and on the other hand to guarantee that whoever was using the water was using it legally and not usurping it. And it was all over the province, and convincing the municipalities, the Sanitation department, all the other institutions, the Cadastre, convincing them all that we had to work thinking about water (Director of Water Resources 1994–2012).

## 5 The Process Undergone to a Water Law in the Province

### 5.1 The Timeline in the Process to a Water Law

Figure 17.2 shows the complex institutional process that leads to the implementation of a Water Law, which involved relationships on international, national, and local scenes. Different aspects of the process are detailed.

- *Pre Provincialization*
  During the existence of the National Territory of Tierra del Fuego, the Hydraulic Directorate—a predecessor of the Directorate of Water Management—was created in 1985. At that time interests were focused on water works but not on water management. The idea of a Water Law was brought about by field experts who migrated from other provinces. In order to develop a bill, it was first necessary to get acquainted with the hydrological characteristics of the territory. The first field studies date back to 1980 (Iturraspe and Urciuolo 2000).
- *Provincialization*
  In 1990, by Law No. 23775, the Province of Tierra del Fuego, Antarctica, and Southern Atlantic Islands were created, and the Provincial Constitution was enacted on June 1, 1991. It states that the general interest of water as a public resource must be regulated, protected, and guaranteed by the State (Art. 83).



THE PROCESS UNDERGONE TO A WATER LAW IN THE PROVINCE

| YEAR DEFINER | INTERNATIONAL AND NATIONAL LEVEL | TIERRA DEL FUEGO AIAS PROVINCE LEVEL |  | WATER LAW PROJECTS |
|---|---|---|---|---|
|  |  | PUBLIC AREAS AND WATER MANAGEMENT | SOCIO-TERRITORIAL ACTORS AND CONCESSIONS |  |
| UNTIL 1990 | • 1884-1990. Tierra del Fuego is declared National Territory | • 1985. Creation of Tierra del Fuego Hydric Direction | • Political and technical management areas |  |
| 1991 | • Environmental Treaty between Argentina & Chile | • Provincialization of Tierra del Fuego | • Political management areas |  |
| 1992 | • Federal Board of Investments Principles of Dublin | • Provincial Environmental Law | • Technical Actors. Direction of Natural Resources | • 1992-1993. Preliminary Project on Water Law |
| 1993 |  |  |  |  |
| 1994 | • Sanction National Constitution |  | • Joint work, actors from technical areas and water users | • Review and participatory analysis on preliminary project |
| 1996 | • National Direction of Hydric Policies. Creation of Expert Group in Hydric Resources shared with Chile | • Direction of Hydric Resources starts depending on Sub Secretariat of Natural Resources |  |  |
| 1998 | • First Managers Workshop on Water Basins Organizations In Latin America and Caribbean |  | • Local water experts present the Water Bill to the local community | ★ PRELIMINARY PROJECT<br>• Preliminary Project on Water |
| 1999 | • Hydric Resources Working group shared by Chancellors | • First Hydric Resources Meeting in the Southern Region<br>• Seminars on Hydric Basin Resources shared between Argentina & Chile |  | ★ PROJECT 1<br>• First Law Project at Legistation P1-220/1999 |
| 2000 | • "Water War" Bolivia<br>• National Sub Secretariat of Hydric Resources | • Workshops on Water Policy among provinces. Tierra del Fuego presents the Water Bill | • Citizen participation in workshops organised by the Water Resources Technical Area |  |
| 2002 | • Approval of General Environmental Law No 25.675<br>• Hydric policies, private and public model debates |  |  |  |
| 2003 | • Federal Water Agreement: Creation of Hydric Federal Council (COHIFE) Guiding Principles of Hydric Policies in Argentinean Republic |  |  |  |
| 2004 |  | • Transitory regulations for authorization of uses of water and hydric works |  | ★ PROJECT 2 344/2004 |
| 2005 | • 2005-2015. International Decade for Action "Water for Life" UNESCO |  |  |  |
| 2006 |  |  |  | ★ PROJECT 3 442/2006 |
| 2007 |  |  | • Rio Grande Water Basin Creation of binational groups of experts |  |
| 2008 |  | • Creation of the Secretariat of Enviromental and Sustainable Development (SDSyA) | • Attempted creation, organism of users Rio Chico<br>• Province Legislation, SDSyA, workshop discussion with participation of all different sectors | ★ PROJECT 4 180/2008 |
| 2009 |  |  |  |  |
| 2010 | • UNO Human Right to drinkable water and drainage |  | • The Water Management sector actively disseminates the Water Bill in the mass media<br>• Lack of a Water Law. Growing concern in environmental organisations and the community as whole | ★ PROJECT 5 224/2010<br>★ PROJECT 6 266/2010 |
| 2011 | • 2010-2011. Approval of the minimum standards for the protection of glaciers bill |  |  |  |
| 2012 |  | • SDSyA presents a new Water Law Project to Provincial Legislature which complies with the glaciers protection bill |  | ★ PROJECT 7 160/2012<br>★ PROJECT 8 268/2012 |
| 2013 | • Project on minimum standards for the protection of Wetlands first gains parliamentary status |  |  | ★ PROJECT 9 227/2013 |
| 2014 |  |  |  | ★ PROJECT 10 072/2014<br>★ PROJECT 11 078/2014 |
| 2015 | • Argentinean Republic New Commercial and Civil Code | • Ushuaia, 2015: XXIV Hydric Federal Counseling Assembly. Declaration urges for a Water Law | • First Regional Water Forum in Patagonia | ★ PROJECT 12 203/2015<br>★ PROJECT 13 253/2015 |
| 2016 |  | • Creation of the Environmental, Sustainable Development and Climatic Change Secretariat | • Debate on Natural Resources Commission of the Provincial Legislature with citizen participation | ★ PROJECT 14<br>• 130/2016. Approval of the Law 1,126 Integrated Management of Hydric Resources |
| 2021 |  |  |  | • Approval Regimentation of Law 1,126 |

Source: self generated data

**Fig. 17.2** Timeline of the process toward a Water Law in the Province of Tierra del Fuego, AIAS

During the following years, different laws on the management of the environment and natural resources were passed (Environmental Law No. 55, Law No. 105, Law No. 244). However, none of these included water regulations. In 1991, an Environmental Treaty between Argentina and Chile was signed which included a specific protocol about hydric shared resources.

- *First Preliminary Bill on Water Management*
  In 1992, the Directorate-General for Natural Resources sought to obtain a basic regulatory framework for all natural resources. In relation to water legislation, it requested the Federal Investment Council for technical cooperation to draft a preliminary Water Law. In 1993, a proposal was drafted by a renowned specialist in Environmental Law which, though never submitted to the legislative body, served as a useful basis for incoming drafts, as it required additional adaptation to the provincial water characteristics. Yet, neither the institutional nor the social agreements necessary to achieve a norm were in place. During the same year, the Federal Environmental Pact was signed, and the Federal Council for the Environment (COFEMA) was formalized. Both encouraged the inclusion of Article 41 of the Argentine National Constitution, which establishes the right to a sustainable environment. The preliminary draft thus required further modifications in order to comply with the reformed National Constitution.
- *Preliminary Bill Update According to the National Constitution of the Argentinean Republic Reform and the Incorporation of the Dublin Principles*
  The reform of the National Constitution in 1994 transferred the domain over natural resources to the provinces (Art. 124). In 1996, the Directorate of Water Resources moved from the area of Public Works to the Under-Secretariat of Natural Resources, which included the management of public water. That same year, the enacting of the Provincial Law of Lands promoted new productive activities which created different uses of water such as agricultural watering and aquaculture. At the same time the preliminary project on existing water management was reviewed under expert advice with the support of the National Hydrological Policies Directorate. Among other changes, new reforms are included based on the National Constitution and also the Dublin Principles (1992) (see Sect. 2.2).
- *Between 1996 and 1999, the Hydric Information Increased*
  Regarding the Environmental Treaty between Chile and Argentina (1991) in 1996, the National Directorate of Water Resources promoted the creation of a working group of experts in order to jointly obtain information systems and maps and a general planning for the use of the resources. By 1999, two of the five water basins were selected, Hua Hum River in Valdivia plus Rio Grande River in Tierra del Fuego, AIAS.
- *Between 1997 and 1999, the Agreement on Regulation Progressed*
  Over this period, the consensus process on regulations was strengthened. On the one hand, it was possible to advance intra-institutional consensus between technical areas of the national and provincial government (waterworks, land office, municipalities, others) that, on many occasions, were unaware of, or even resisted the purview of the regulations. For example, within the land office area, in rela-

tion to the delimitation of land ownership domains, conflicts of interest were detected since private projects (real estate, among others) were installed in areas considered of public domain according to the water criteria set forth by the law. On the other hand, agreements between technical areas and users advanced. Furthermore, presentations and workshops were held with private stakeholders and the community as a whole in order to agree on a Water Law.

- *First Hydric Resources Meeting in the Southern Region* (Ushuaia 1999)
  Along with this process and inspired by the First Water Basins Managers from Latin America and the Caribbean Workshop (1998), a similar meeting was held in Ushuaia, attended by directors of hydric resources and actors from different provinces sharing common interests. This federal event could be considered a precedent for what would later be known as the Federal Hydric Council (Consejo Hídrico Federal, whose acronym is COHIFE).
- *First Water Law Bill Presented to the Legislature and the Community*
  In 1999, the first bill was submitted to the legislative body and was presented to the community. During this period, the process for building up consensus continued. First approaches of private stakeholders explicitly intended to influence the law draft. Among these, since 2001 the Rural Society maintained a defensive position based on the vision that State control could affect their private interests. There was also a distrust regarding rising charges for water use and reluctance to allow citizen access to public waters on private lands.
- *The Federal Water Agreement* (*2001–2003*)
  During the 1990s, debates around private and public models for water management took place. In those days, the political and economic neoliberal hegemony multiplied the discourses and projects which proposed the commodification of water, following the model of Chile. The fact that Argentina is a federal country hampered the enforcement of the proposal since international agencies such as the World Bank had to agree with all the provinces.
  In this context, most actors responsible for water resources within the country supported the need for an integrated water management. For that reason, a solid legal basis was required to uplift environmental policy in a general way. In 2002, the General Environmental Law No. 25675 was enacted. The National Under-Secretariat of Water Resources held water policy workshops for all provinces to debate the principles for the management and uses of water in the country. In this way the Federal Water Agreement was reached in 2003. Within this framework, the Guiding Principles for Argentinean Hydric Policies were established, and at the same time the Federal Hydric Council (COHIFE) was created. Its main aim was to attain a regulation that could balance economic, environmental, and social interests. The challenge was that frequently the actions that these three interests promote could become antagonistic or could also exclude each other (Consejo Hídrico Federal 2003). The aforementioned principles connect water regulation with the environment and, in doing so, include an environmental perspective in all stages of planning, managing, and evaluating water policies. Tierra del Fuego, AIAS Province, signed the Federal Water Agreement and in 2004 updated the

bill that had been submitted to the legislative body based on these guiding principles which asserted that water is a good linked to life and emphasized its social and environmental value by proposing water to acquire economic value only when the needs linked to previous assessments have been met. This introduced differences with the international principles of Dublin (1992) which highlight the issue of water scarcity, emphasizing the dimension of economic value. On a federal level, the water paradigm as a public good was defended by experts in hydrological resources. This implied highlighting the role of the State in its regulation by limiting the access of markets to the country's water. The 2004 project lost parliamentary condition after two years without being treated, according to Law No.13,640.

- *Progress in the Regulation of Transitory Provisions* (*2004*)
  In 2004, transitory provisions for the regulation of water on a provincial level emerged from a proposal made by the director of hydric resources. Procedural regulations for the authorization of water uses and for the authorization of hydraulic works were set. The regulations covered some needs such as legal authorization of water use to economic actors that were applying for productive loans and also the development of different registries, among them a temporary registry of users. These were obliged to pay for water according to the Tariff Law. However, as there was no Water Law that could grant concessions, no money was actually paid except by oil companies. On the other hand, rising projects involving new actors who requested to use water generated more local participation in favor of the Water Law and growing interests for the regulation of the resource. Despite being a step forward, the new bill submitted in 2006 lost parliamentary status.
- *Bills by the Secretariat of Sustainable and Environmental Development* (*SDSyA*) (*2008*)
  In 2008, the Secretariat of Sustainable and Environmental Development of the province was created combining the areas of environmental management under an integrated management paradigm. The Executive Power sent a new Water Law Project to the Provincial Legislature, based on the existing one and taking into account the need to reinforce the ecosystemic focus on Integrated Water Resources Management and the principles established by the General Environmental Law No 25,675. Also in 2008, there was some progress on the debate about certain aspects of the Water Law. Regulation on sport fishing was a keystone, as one of the many conflicting activities. During this period the Provincial Legislature and SDSyA jointly organized a workshop about the Water Law Project, which was well attended by different stakeholders and expert advisors, and the debate gained visibility in local media. Nevertheless, the bill did not achieve legislative treatment. It became obvious that the difficulty in treating the law was rooted in social and territorial processes that lead to distinct interests and valuations of the water present in the conflict. In 2010 other parliamentary blocks presented alternative Water Law projects, yet these did not obtain parliamentary treatment.

- *A Landmark in 2010: UN States the Right to Drinkable Water and Sanitation*
  The importance of this declaration at global and national level had precedents in other struggles for rights, such as the Water War in Cochabamba, Bolivia. Although it did not have a direct impact on the province or the Water Law, its significance and social relevance must be understood as part of the process of water valuation construction from integrated perspectives in the local arena.
- *Growing Visibility of the Conflict*
  In 2012, the project was adapted by the SDSyA to concur with the new national legislation by including specific articles about glacier and wetland protection. The project was sent to the Legislature but was not treated and hence lost its parliamentary status in 2014. The submission of a new project the same year caused great expectations as general consent was growing. The issue gained momentum in mass media as well as in the discourses of environmental organizations. Nevertheless, there still were objections from the rural society about boundaries such as the tow path that weakened the agreements reached and again flawed the legislative treatment.
- *The Support from the XXIV Federal Water Council* (*COHIFE*) *General Meeting. Ushuaia, 2015*
  In 2015, the Federal Water Council held its XXIV General Meeting. The Provincial Executive Power asked COHIFE for a federal declaration of interest for a Water Law in the Tierra del Fuego, AIAS Province. The declaration was paramount, mainly because it placed the water law in the political and public agenda as a pressing issue regardless of political changes occurred in 2016. Also during 2015 a Regional Forum about Water in Patagonia Austral was organized by the provincial Secretariat of Sustainable and Environmental Development, the National Institute of Agricultural Technology (INTA), and the National University of Tierra del Fuego, AIAS (UNTDF, AIAS); the event further showed the need for a Water Law by helping to understand the water-related conflicts in the face of the instability of different economic, environmental, and social interests. The view of water as a public good was so being stressed.
- *New Civil and Commercial Code of the Argentinean Nation* (*2015*)
  The New Civil and Commercial Code of the Argentinean Nation (Código Civil y Comercial 2015) establishes all public goods,[21] among them the water[22] and also

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[21] Not all kinds of goods are considered public, but rather those that (unlike private goods) must be assigned to *public use*, directly or indirectly, and in the latter case, they must be *final goods* or *fixed goods* affected by common usage. There is a distinction between the legal regimes of property in the *public domain* and in the *private domain* of the State. The first refers to the *State as a Public Power* and the second to the *State as a Legal Person*. In the first case, the State, as a public authority, has a *guardian role* on these goods; in the second case, the State, as a legal person, can be the *owner* of the goods, in which case they are subjected to the ordinary rules of private property, except for some modifications (Marienhoff 2011).

[22] Public domain goods (waters): rivers, estuaries, streams, and other waters that run through natural channels, groundwater, lakes and navigable lagoons, glaciers, and the periglacial environment, all other water that has or acquires the ability to satisfy uses of general interests, groundwater. Territorial sea, inland waters, bays, inlets, gulfs, maritime beaches.

the regulation of such goods. Goods in the public domain, including water are inalienable (they cannot be sold and they are not included in commercial operations of any sort). They are also non-seizable (they cannot be guaranteed as part of any payment agreement), and they are imprescriptible (they cannot be lost over time) (Art. 237). According to this, when the State affects or incorporates goods into the public domain and allows their use by the private sector or the citizens, it is paramount that they never lose their condition, so rights are granted only through concessions (Art. 239). Additionally, Art. 240 provides consequences for environmental regulation and addresses limits to the exercise of individual rights based on compatibility with rights of collective incidence.

- *The Project Passing* (*2016*) *and Regimentation* (*2021*)

  Toward the end of 2016, the Executive Power, through the Secretariat of Sustainable and Environmental Development and Climate Change (SDSAyCC), submitted a new law project based on the existing one, with a strong aim toward the conservation of hydric water basins and the protection of water quality. In this way there were already three different law projects under treatment in the Provincial Legislature, taking into account the previous ones presented in 2015. The Draft of the Water Law was debated in the Natural Resources Commission of the Legislative body and later to the Provincial Council for the Environment (composed of representatives of Hydric Resources Sector, the National University of Tierra del Fuego, Southern Center for Scientific Research (CADIC), town halls, tourism sectors, environmental organizations among others).

  At that moment, actors from the SDSAyCC, Hydric Resources, Legislators from the Natural Resources Commission, and some other legislators joined in a meeting with the rural society. In this meeting, previous objections were sorted out, and an adaptation to the Project Tow Path of 1974 from the New Commercial & Civil National Code enacted since 2015 reduced the path from 35 to 15 meters. New debates were overruled due to the long process of negotiations and concessions established in previous years. Finally, by the end of a year seriously affected by an economic and social crisis, during the Legislative Session of November 30, 2016 (Poder Legislativo de la Provincia de Tierra del Fuego, AIAS 2016), practically without debate and unanimously,[23] the bill was passed and promulgated two days later.

  Preexisting irregular situations of water uses had to be reviewed from the perspective of the new law. The priority of economic interests, on part of the same private actors with strong lobbying power that had obstructed the promulgation of the law, also influenced in delaying its enactment. Only with a new change of the provincial government in 2021, the regimentation of Law No. 1,126 was achieved, thus enabling the chance for an adequate implementation.

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[23] Significantly, the arguments of the legislators present during the act of sanction made reference to a *historical debt* without there being a strong partisan appropriation, as different political party blocks had presented Water Bills: *Partido Justicialista* (1999, 2006)/*Partido Justicialista y Frente para la Victoria* (2016); *Unión Cívica Radical* (2004, 2013, 2015); *Coalición Cívica. Acción República de Iguales* (2008, 2010, 2012)/*Partido Social Patagónico* (2014, 2015). *Movimiento Popular Fueguino* (2010, 2012, 2014).

### 5.2 Main Contributions of Law No. 1,126 of Integrated Management of Water Resources

The current Water Law makes specific contributions for the intervention in the aforesaid conflicts:

- *Contributions Linked to Management*
  The present Water Law emphasizes the role of the State which is oriented by the Guiding Principles of Hydric Policies of the Argentinean Republic (Consejo Hídrico Federal 2003) through a Provincial System of Hydric Resources set by the technical agency responsible for water management. This system is also constituted by users and water basin agencies, understanding that "a regulation is a management tool associated with adequate management decisions" (Urciuolo et al. 2017).
- *Contributions Linked to Hydric Regulation and Codes*
  The conflicts associated with water resource assignments arose from the lack of a concession system which sought to cover temporary permits. In connection to this, the Water Law contemplates a register that deals with rights (unique register of public water users) associated with a thorough inventory of all water resources (wetlands, glaciers, groundwater flows; with information about actual water flows among other pieces of information). This information allows for the generation of criteria for granting permissions and concessions with appropriate control over common and special uses. Permits for special uses of public water consider rights as well as duties; they offer legal warrants and set incompatible uses for different existing usages. Regarding common uses, the Water Law allows formalizing the right of way to guarantee access to public waters in private lands, which was one of the main conflicts due to the lack of regulation. Also, the contravention system observed by the Water Law will penalize those users who do not fulfill their duties as established in the permits and concessions. Furthermore, the system of financial charges for the use of water takes into account incentives for efficient and rational use of water.
  The Law on Water includes delimitations allowing for the clearance of unruly occupied river banks, flood evacuation lines, and watering zones and also the constitution of water rights for special as well as for ordinary uses of water. Additionally, it will pave the way for the clarification of competences between the City Council and Provincial Land Registry Office, in order to set the boundaries for the concession of land and new housing developments. Moreover, the Water Law establishes the creation of a provincial strategic plan by the executive power elaborated with participatory mechanisms, as well as a provincial water plan. Planning instruments considered are: a control system, a system with information on water resources, an emergency committee, and a system for promoting the culture of water.
- *Contributions Linked to the Conservation of Water Basins and Water Protection*
  The Water Law makes integrated contributions to the conservation of hydrological basins through a strategic hydrological plan, which includes measures for the

protection of water quality and drinkable water reservoirs through monitoring and remediation programs and protection of water basins and watersheds, glaciers, and wetlands. It also foresees the regulation of activities in areas adjacent to glaciers, an integrated improvement of flooded areas, or areas which are likely to be flooded.

While protection of water quality was regulated by the Provincial Environmental Law No 55, the conservation of watersheds required a specific rule. The new law allows the government to control spills on rivers and streams and also to prevent pollution events which have caused claims for judiciary protection of the resources.

In sum, the law has the following general objectives: (1) to allow the solution of preexisting irregularities pertaining to the use and access to public waters; (2) to provide the necessary tools for the adequate regulation of the use of provincial public waters, as well as to provide legal guarantees to public water users; (3) to foster the sustainable, rational, and productive use of water as a tool for regional development; and (4) to support the conservation and protection of water and hydrological watersheds, glaciers, and wetlands.

## 6 Conclusions: Valuations of Water According to Water Law in Tierra del Fuego, AIAS, and the Defense of Water as a Public Good

Our analysis linked institutional and socioterritorial processes involved in the materialization of the Water Law. At an institutional level, Merlinsky (2013a, b) employs the category of *conflict productivity* in terms of how new laws and institutions are generated due to citizen organization processes in environmental conflicts. Such a category may contribute to the comprehension of some of the processes partially analyzed here, especially the ones that led to the institutionalization of the Federal Water Council (COHIFE). On the other hand, many studies analyze the influence of elites on public institutions through pressure groups or lobby actions in order to hamper the enacting of laws that affect their interests. However, that focus is centered on the analysis of the power of elites and would have obstructed the visualization of other actors who have fostered the implementation of a Water Law via political mediation and everyday negotiation practices. In terms of the socioterritorial level, the *political underground or basement* is a category that belongs to the sociohistorical and the political-cultural analysis developed by Tapia (2008), which shows the political actions present in several practices such as negotiations, collective actions, relationships, and experiences.

On the other hand, the categories borrowed from the Political Ecology field constituted tools that allowed the articulation of the aforementioned levels and the visualization of the disputes between water valuation languages that become intertwined. Generally speaking, water valuations may be territorialized in different time-space levels, with different correlations of forces and in diverse contexts, thus requiring situational analysis. In particular, over the period analyzed in the province, between 1993 and 2016, the discourse of "the defense of water as a common good" was part of the resistance language against the neo-liberal economic policies and water markets. In other words, water valuation as a common good should be understood not only as a legal motto that has driven the law to be passed, but as a valuation language that led to the articulation of environmental, social, and economic interests over water in a specific sociohistorical and territorial configuration disputing other possible valuations of nature. In this regard, the valuation of water as a common good has rooted in Tierra del Fuego, AIAS, through everyday social practices, thus integrating the set of identifications of the population who associate this motto with many other local environmental struggles.

Environmental valuations of water have generated the greatest tension and re-signification in traditional forms of the modern public-private dichotomy. Clearly, the environmental criteria or hydrological criteria do not respond to a legal abstraction, neither to private interests, nor to preexisting ownership titles. For instance, in the progressive link between water and environment, collective environmental rights allow the restriction of private rights, or the rights of certain people when there is an imbalance toward the preponderance of an economic interest. Therefore, obstacles to the Water Law cannot be reduced to the legal dimension of the conflict. On the contrary, they have expressed a long-term environmental conflict which had been muzzled for many years in this territory. In this sense, Political Ecology exceeds the classical liberal political vision that emphasizes the legal language and the judicial vision in the resolution of political conflicts. Political Ecology has made it possible to highlight the political dispute over nature and water, unveiling the growing relevance of environmental criteria in the public debate on water management, in connection with other social or economic criteria.

It should be noted that there were no homogeneous positions in sectors or in interviewed actors; therefore, the predominant arguments are summarized in a general way. Among the obstacles to the law, the main argument outlines the predominance of an economic interest in the uses of water by the private sector, mainly from ranches, to the detriment of environmental considerations or social criteria as well as some failings of the State itself, including partisan disputes. The latifundia sector acted, given its historically consolidated elite power, as a pressure group to prevent the State from exercising its public power to authorize, control, regulate, and collect the canon for the use of water. At the same time, the State did not constitute a homogeneous sector. Conditioned by the political struggles in the territory and with different relations of power in the multiple mediations, the State, with relative autonomy, was able to act (either by omission or by action) contradictorily, as an adversary or guarantor of the law. Adversary, in that it did not give legislative treatment to the Water Law for almost two decades, favoring the interests of the territorial elites. Guarantor, in that it sanctioned the Water Law, although it still has to guarantee the effective exercise of the use and enjoyment of water by all inhabitants.

In sum, the defense of water as a public good constituted the main valuation by the promoters of the Water Law in the territory, who through various political actions, regardless of political affiliation, had the same single purpose: *to force the State to guarantee a right*. Among the promoters of the Water Law, there were technical and water management specialists responsible for water management in governmental areas, who negotiated and built consensus with institutions and multiple stakeholders in the territory. In fact, they were the ones who promoted management models that aimed at institutional changes—mainly addressed at management and land planning—and cultural changes in terms of water valuations that involve the notions of public and common good. In this sense, the guiding criteria behind the water management policies that these actors promoted in the province were the need for a knowledge-based dialogue between technical-scientific arguments and other popular disciplinary and cultural knowledge of water, whether for political decision-making or management, conservation, and conflict resolution practices among others. Eventually, the latter translated into multiple actions by the aforementioned actors during the analyzed period that included: attempts to create user and basin organizations, the will to expand the participation of various stakeholders in technical roundtables for the elaboration of environmental reports, and actions for the dissemination of environmental issues in the media and social networks. In relation to the latter processes, it is worth noting the growing strength of environmental groups that articulate with other social organizations, scientific and educational institutions, and the community in general, to promote environmental education actions and other projects along with the incipient presence of other valuation languages (environmental justice, rights of nature, ecofeminism, decoloniality, and others).

In conclusion, it has been demonstrated that the different ways of prioritizing interests regarding water respond to competing valuations of water sustained by the various stakeholders in different practices. The conflicts between opposing valuations were part of the conditions that hindered the feasibility of the legislative treatment of a Water Law between 1993 and 2016, in the province of Tierra del Fuego, AIAS. For all the above, it can be understood that although the Water Law is currently a fundamental tool for water management, many of the socioterritorial conflicts between water valuations remain open and are part of the territoriality dynamics in Tierra del Fuego, AIAS.

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# Chapter 18
# Reaching Out for the UNDP Sustainable Development Goals in Patagonia

**Miguel A. Pascual, Gabriela Mataloni, and Rubén D. Quintana**

## 1 Introduction

During the Sustainable Development Summit held at the United Nations Headquarters in New York in 2005, the 193 Member States adopted a new agenda entitled 'Transforming Our World: The 2030 Agenda for Sustainable Development'. The Agenda includes 17 Sustainable Development Goals (SDGs) that provide a roadmap for joint action by world countries to end global poverty, build a life of dignity for all, and leave no one behind. Through the SDGs, human well-being was tightly linked to environmental preservation, much in the same way as the Millennium Ecosystem Assessment did before (MEA 2005), but with a clearer and more specific focus on people's problems, including poverty, health, climate change, institution building, gender inequality, and women's empowerment. Post-2015 Development Planning kick-started action in the form of financial and partnership assistance and specific programmes and projects.

Although the 17 SDGs are inextricably linked and are all relevant to freshwater and wetland ecosystem integrity and management, one of them directly focuses on freshwater, namely, SDG 6 on Clean Water and Sanitation. Because water scarcity affects more than 40% of people worldwide, a figure projected to rise as temperatures do, this SDG proposes to ensure availability and sustainable management of water and sanitation for all. The specific targets and actions include achieving

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universal and equitable access to drinking water and sanitation through investment in infrastructure and sanitation facilities, pollution and wastewater reduction, increase in water-use efficiency, and protection and restoration of water-related ecosystems, as well as promoting good water governance through education, integrated management, cooperation, and community involvement and participation.

Other proposed goals also include different water and sanitation targets. For instance, SDG 3 on good health and well-being considers the importance of clean water supply to prevent and reduce illnesses; SDG 11 on sustainable cities and communities takes into consideration the risks posed by water-related disasters; SDG 12 on responsible production and consumption remarks the importance of the environmental management of chemicals and wastes as they affect freshwater; and SDG 15 on life on land considers the importance of protection, restoration, and sustainable use of inland freshwater ecosystems and their services, including the management and control of exotic species.

The International Convention on Wetlands (Ramsar Convention) urges Contracting Parties to integrate into their policies and programmes the conservation and wise use of wetlands in accordance with the SDGs as a way to meet national commitments (International Convention on Wetlands 2018). On a global scale, the SDGs ‘improvement of water quality’, ‘sustainable management’, and ‘efficient use of natural resources and sustainable food production’ were considered priorities in order to achieve sustainable development of these ecosystems (Jaramillo et al. 2019).

The SDGs are about fulfilling human needs and expectations, with equity and with environmental sustainability, wherein the preservation and restoration of freshwater ecosystems has an important role. From a biophysical point of view, Patagonia is very well furnished to meet such a challenge. It is a large, ecologically diverse region (Chap. 1), a large producer of energy and foodstuff, with a bounty of freshwater and associated benefits to people (Chaps. 9 and 10) and with a low population density. As an example, Patagonia produces over 20% of the hydropower and 95% of the oil and gas of Argentina and contains over 80% of the native forests and over 10% of the forest plantation of Chile (INFOR 2022), in an area equivalent to 30% of Chile and Argentina combined, while hosting a mere 6% of the total population of both countries. Patagonia is also a world-class tourist destination. On top of these continental natural goods and services, Patagonian marine areas sustain important fisheries and lavish marine wildlife refuges in coastal communities.

The true challenge therefore is not related to the availability of resources, but to the wise use of them, collectively visualising desirable scenarios and trajectories for the region’s future and setting appropriate courses of action. Visions and actions can only be construed based on a realistic assessment of challenges and opportunities, backed by good science and strong governance. In this chapter we present our view of the state of the art of science and governance in the region, highlighting the strengths and the weaknesses to reach the UNDP Sustainable Development Goals.

## 2 The Ecosystems Perspective for Studying Freshwaters of Patagonia

Freshwater and wetland ecosystems are natural sinks for different processes that involve the water-land-atmosphere interface and, often, the first at a territorial scale to show symptoms of environmental degradation (Dudgeon et al. 2006). They also constitute ecosystems of great natural, social, and cultural value; human health and well-being depend directly on them, so their deterioration is quickly projected onto society. As the genesis of water problems typically has an ecosystem scale, it is unlikely that we could adequately interpret water-related problems or find lasting solutions without an analysis that contemplates the spatial and functional dimensions responsible for their integrity and functioning (Likens and Bormann 1974). The promotion of collective actions in favour of the health of aquatic and wetland systems also requires perspectives that reveal their connections with the complexity of the territorial matrix that contains them. The ecosystem services paradigm provides a conceptual and instrumental framework that allows considering the problems of these ecosystems at the appropriate structural, functional, and socio-environmental scale (Pascual et al. 2022). But before elaborating on this paradigm, it is relevant to consider the scale of change typically affecting freshwater and wetland ecosystems of Patagonia.

### *2.1 Drivers of Change in Patagonia*

Freshwater and wetland ecosystems are affected by different drivers of change that are, in general, classified into two categories (Pascual et al. 2022): climate (climate variability, extreme events, climate change) and land use/land cover (urbanisation, deforestation, reforestation, desertification, intensification of agriculture, and farming). Land use/land cover is also itself affected by engineering works (dams, irrigation, water transfers), disturbances, such as fires, and natural resource exploitation such as mining. These drivers of change can significantly affect various ecosystem services through their effect on watershed components and on the ability of ecosystems to physically regulate the hydrological cycle of a region (Chap. 11). For example, climate and land use/land cover modulate water production, hydrological regulation, edaphic storage, aquifer recharge, the intensity and frequency of surface runoff phenomena, and the capacity of environments to retain sediments, nutrients, and contaminants.

Drivers related to climate change are the ones that generate the greatest impacts on the global hydrological cycle. Meteorological observations show that Patagonia experienced temperature increases in the order of 1 °C between 1960 and 2010 (CIMA 2014). In the northern region of the Patagonian Andes, rainfall had a significant negative trend during the same reference period (CIMA 2014), in accordance with the negative trends recorded in seasonal flows of Atlantic rivers of northern

Patagonia (Pasquini and Depetris 2007; Vich et al. 2014; Chap. 9); the same trends are verified in central Chile, which shares the same climate signal (Vicuña et al. 2020). Climate change projections for the end of the century estimate temperature increases as large as 3 °C for parts of central Patagonia. Regarding precipitation, for the scenarios with the highest emission of greenhouse gases, a decrease between 10% and 20% is projected toward the end of the century over the west of northern and central Patagonia (CIMA 2014), which will significantly modify the hydrological cycle in these regions (Pessacg et al. 2020; Chap. 4).

Regarding changes in land use/land cover, different local and regional processes could modify patterns of evapotranspiration, water reserves, sediment retention, and organic and inorganic compounds, with direct effects on ecosystem services, which can intensify or weaken the impact of climate change. In turn, climate change can induce land use/cover change processes. At a regional level, the most profound changes are those produced by the desertification of large areas of the Patagonian steppe in Argentina and the deforestation in Chilean Patagonia, particularly on the northern end. Desertification has been particularly intense in the north-central section of eastern Patagonia as the result of a long history of unsustainable grazing in combination with change towards a warmer and drier climate (Therburg et al. 2019; Chap. 11). From 2001 to 2020, Chile lost 2.10 Mha of tree cover, equivalent to a 11% decrease in tree cover since 2000 (Global Forest Watch 2022). These two drivers reduce the capacity of land to retain water and sediments, increasing the torrentiality of rivers and their erosion potential. The hydrological consequences of both drivers are changes in the timing and extent of peak flows, flooding, and elevated turbidity and nutrients in waters. At a smaller scale, changes in land use/land cover significantly affecting the quantity and quality of freshwaters of Patagonia include dams, irrigation, urbanisation (Chaps. 10 and 11), and introduction of exotic species (Chaps. 13 and 14).

## 2.2 Concepts and Applications of Ecosystem Service Perspectives

The ecosystem service (ES) paradigm is based on a simple idea: human well-being – physical, emotional, and spiritual – depends directly on the multiple benefits that societies obtain from nature (Pascual et al. 2022). These benefits have been originally termed ‘ecosystem services’ (MEA 2005) and, alternatively, ‘Nature’s contributions to people’ (NCP) by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al. 2018).

Whereas the basic tenet that human well-being depends on nature integrity has been mostly out of discussion, the application of the paradigm to real-world problems has been dotted by arguments, claims, and criticisms (Pascual et al., 2022).

Following the publication of the Millenium Ecosystem Assessment (2005), the ES approach became one of the most widely used to interpret human relations with nature, having an important influence both in academia and in the fields of policy design. The evolution of the approach, the conceptual frameworks, and their definitions and classifications have been the subject of significant scrutiny and debate (Hermelingmeier and Nicholas 2017). Even among its adopters, there has been a multiplication of sub-approaches and platforms, with different visions (Díaz et al. 2018; Kenter 2018; Masood 2018).

The aims of the approach are very ambitious: to capture the pillars of human well-being as related to the state of nature and, at the same time, establish measurable state variables and dynamic socio-ecological ecosystem models to operationalise the concept when conducting specific analyses. Yet, the paradigm does not offer a 'turnkey' programme, but there are innumerable decisions and modalities in its application. In fact, most criticism relate to perceived oversimplifications or biases at the time of application regarding, for instance, the specific ecosystem services considered, the weight that biodiversity preservation is given in the analyses, and, most importantly, the way the provision of ecosystem services is valued, including the level of attention given to the traditions and expectations of the various social groups that are typically involved in particular scenarios (Pascual et al. 2022) frequently characterised by disparate valuation languages (Chap. 17).

Nevertheless, beyond these arguments, several features of the approach make it unique (Pascual et al. 2022). The ecosystemic perspective provides a distinctive framework to analyse the land-water-atmosphere interface in a dynamic way and to evaluate the consequences of different human interventions and remediation options for aquatic, terrestrial and wetland environments. The flexibility of the approach is one of its main strengths, since it allows to articulate sectors, visions, and disciplines in the construction of public policies in the face of complex socio-environmental problems. And because the ES concept is multidisciplinary in essence, its application very naturally promotes the articulation between scientific-technical disciplines (McDonough et al. 2017). In the specific case of freshwater and wetland ecosystems of Argentina and Chile, the perspective of the ES can provide a bridge between disciplines that have had a long independent and largely uncoordinated tradition in their views of water (Pascual et al. 2022), such as hydrology, agronomy, limnology, wetland ecology, engineering, and social sciences. Finally, the ES perspective promotes an environmental view of water security problems. The traditional view is dominated by the so-called grey engineering, aimed at actions in the places where problems are experienced and focused on the application of solutions based on classic infrastructure. The ecosystemic vision of water security problems fosters a spatially and functionally broader view of river basins, focusing on the origin of the problems, promoting nature-based solutions on an equal footing with classic infrastructure-based solutions (Pascual et al. 2022).

## 2.3 General Challenges in Applying the ES Perspective: Science and Governance

Bringing the ecosystem services paradigm to be an operational management approach comes with a sizable price tag in terms of application requirements (Beaumont et al. 2018). The very same good properties of the approach (ecosystem based, geographically explicit, multiple ecosystem services considered at multiple scales and from the point of view of multiple stakeholders) soon translate into significant technical demands: multidisciplinary research, considering processes operating at multiple scales and across the land-water boundary, within a socioecological framework. Moreover, the actual adoption of the approach by decision making systems also requires governance structures that are science-based, with strong consultation mechanisms and with the participation of multiple stakeholders, and able to deal with the uncertainties associated with multi-scale scientific results and with the intrinsic unpredictability of freshwater ecosystems. With these requirements as a backdrop, in the next section we reflect on the state of the art of science and governance in Patagonia to reach the UNDP Sustainable Development Goals in relation to freshwaters.

### Freshwater and Wetland Science

As early as in the nineteenth century, Francois Forel coined the term 'limnology' to define a new branch of science devoted to the integrative study of continental waters, wherein different disciplines such as geology, hydrology, physics, chemistry, and biology inform each other (Vincent and Bertola 2014). Contrarily to the prevailing view of the time, he regarded lakes as complex systems highly interdependent with surrounding ecosystems and with the human presence, which he recognised to have 'a powerful influence on the natural environment and all living things around him' (Forel 1904). In Argentina, pioneering limnological research was carried out mainly by biologists, particularly ichthyologists (Espínola et al. 2012), and therefore early studies of freshwater ecosystems were carried out mostly within water bodies with well-defined boundaries. It was only later that connections with other systems acquired relevance in interpreting freshwaters and that also a new type of water-dependent ecosystems – the wetlands – was identified (e.g. Neiff et al. 1994; Malvárez 1999; Brinson and Malvárez 2002) (Chap. 2). This led to a delayed onset of the ES approach for the study and valuation of freshwaters that continues to the present day. In the last decade, a few pioneer surveys focused on ES of wetlands at different spatial scales were carried out in Argentina (Kandus et al. 2011; Oddi and Kandus 2011; Orúe et al. 2011; Carreño et al. 2012). As for Patagonia, although the ES approach has nurtured a considerable quantity of high quality research in Patagonia (Peri et al. 2021), this has focused mainly on terrestrial socio-ecological ecosystems. Exceptions are provided by Liberoff et al. (2019), and Pessacg et al. (2020), and related publications. Nevertheless, the lack of an equal progress in this direction regarding aquatic and wetland ecosystems has not been identified as one of the pending challenges for the comprehensive application of ES at a regional level (Nahuelhual et al. 2021).

## Governance of Water Resources

Argentina is organised as a Federal Republic composed of 24 jurisdictions (23 provinces plus the Autonomous City of Buenos Aires, seat of the Federal Government). The water governance system could be defined as polycentric, with federal boundaries (Berardo et al. 2013; Trimble et al. 2021a, b). The 1994 reform to the Argentine Constitution granted subnational governments (provinces) ownership and management of natural resources, including surface water and groundwater, but the Federal Government is trusted with the protection of the environment and the ability to set standards for its use and conservation (Chaps. 9 and 17). The main water agency at the federal level is the Infrastructure and Water Policy Secretariat, dependent on the Ministry of the Interior, Public Works, and Housing. Other federal dependencies also influence water management decisions, such as the ministries of the Environment and Sustainable Development (regarding sustainable water use), Agriculture (regarding water for irrigation), and Foreign Affairs, International Trade and Worship (regarding international basins).

Argentina does not have a ‘water law’ per se, but water guidelines are present throughout multiple laws and codes. Among these, the Environmental Water Management Regime of 2002 recognises the watershed as an indivisible unit of environmental management; establishes minimum environmental protection requirements to preserve water, ensuring its sustainable use; and mandates the creation of basin committees for inter-jurisdictional watersheds to foster collaboration (Berardo et al. 2013). This legislation led to much debate during the early 2000s between provincial and federal governments, where the central issue was the definition of the actual level of decentralisation that water management should have. This debate led to the elaboration of the ‘Guiding Principles for Water Policy’ as an agreement between the provinces and the federal government. This set of 49 principles defines technical, social, economic, legal, institutional, and environmental aspects of water management and was the basis for the creation of the Federal Water Council (Consejo Hídrico Federal - COHIFE), which is the only formal instance for water policy coordination at the federal level (see also Chap. 17). The principles were meant to facilitate a new, more comprehensive water law, although this new law has not been sanctioned yet. The ‘Guiding Principles’ are conceptually very advanced since they embrace integrated water resources management (IWRM), recognising the environmental integrity of river basins and call for society’s participation in goal setting of water plans, decision-making processes, and management control. In practice, there is a wide variation in the design and implementation of institutional arrangements at the provincial level for dealing with water issues (Trimble et al. 2021b). For instance, not all provinces grant water resources the same recognition in their constitutions, some explicitly mention mechanisms for user participation in decisions regarding water or environmental issues, whereas others do not even contain the word ‘water’ in their text (Pochat 2005).

Argentinian provinces participate in basin organisations (committees and authorities) for dealing with water management issues in 16 inter-jurisdictional basins, three of them in Patagonia. These basin committees, as well as COHIFE, are structures that act mainly as venues for policy coordination among provinces and other governmental organisations, sometimes with some level of control over water allocation, dam management, and monitoring programme implementation (i.e. AIC, Trimble et al. 2021b, Chap. 9). On the other hand, a key aspect of IWRM, such as forums involving all relevant stakeholders – including water users – in the decision-making process, is rare. Trimble et al. (2021b) found that Argentina was the least advanced when compared with Uruguay and Brazil in terms of user and social participation in water management. The Chubut Province of Patagonia provides one exception by being one of a handful of districts in the country that have set up internal basin committees. In 2013, to overcome the fragmentation of water governance in the province, a provincial law created a basin committee for the entire Chubut River basin, as well as for other basins in the province (Trimble et al. 2022). The Chubut basin committee is run by the provincial water authority and the Ministry of the Environment and gathers deputies from each municipality in the basin, as well as representatives from the irrigation, hydropower and human consumption sectors and from research agencies active in the region. In practice, however, the committee has served mainly as a space for light coordination among members. A heavy dependence on the will of the water authorities for calling meetings and influencing policy decisions, paired with changes in provincial administrations, has curtailed the authority of the committee in recent years.

The Chilean model of regulating the use and consumption of continental waters is unique in the world. Installed in 1981, within the framework of the neoliberal reforms of the Military Regime, the Water Code states that a ‘water right’ integrates the use, enjoyment, and disposal of water into the patrimony or domain of its owner (Donoso 2018; Chap. 9); ‘individual property’ regards water as separate from land, thus permitting the holder of a water right to transfer it. Although waters remain national goods for public use, in Chile, more than in any other country in the world, water policies are governed by the free market, wherein water is managed as a commodity subjected to the forces of supply and demand (see also Chap. 17). Water User Associations (WUAs) play a key role in water management in Chile since they are in charge of the distribution of the resource as well as the resolution of conflicts. Despite some success at performing their administrative functions, multiple basins and rivers have not yet succeeded organising around a WUA, and most of those that are organised do not have the active participation of their users (Blanco et al. 2020). Also, there is a lack of representation of all sectors involved, usually leaving hydroelectric, mining, and environmental uses aside.

The public administrative structure of water governance in Chile, on the other hand, is relatively simple. Although there are multiple central authorities involved in the formulation and regulation of water policies and the judicial system, the main one is the ‘Dirección General de Aguas’, a technical body of the Ministry of Public

Works (MOP), whose powers and attributions include water allocation, hydrological planning and monitoring, water research, WUAs supervision, managing the national public water registry, and water use monitoring (Donoso 2021).

The management model of Chile allows the concentration of ownership exacerbating monopolistic situations, hindering an optimal allocation of water resources and producing overprices in water services, with serious impacts on social equity regarding access and water security. Moreover, the water administration is carried out at the level of specific uses and specific river sections and not at the level of basins, which seriously limits the possibilities of controlling environmental problems and resolving conflicts. Undoubtedly, this model does not provide adequate tools to develop any integrated basin management to ensure sustainable water use; the lack of integrated management has led to over-extraction, depletion of multiple aquifers and conflicts, thus leading to the Tragedy of the Commons (Blanco et al. 2020): individual users with open access to a resource, without other governance rules, act following their own self-interest and contrary to the common good (Hardin 1968). In fact, Chile has had a poor record when it comes to the implementation of IWRM in general or the involvement of all relevant stakeholders in the decision-making process in particular (Bennison et al. 2019). Some participatory processes for sustainable water management at the basin level are emerging, such as in the Rapel basin (Rojas et al. 2020).

## 3 Are We on the Right Path?

The United Nations 2030 Agenda for Sustainable Development recognises water as the lifeblood of ecosystems, vital to human health and well-being and a precondition for economic prosperity. That is why SDG 6, the availability and sustainable management of water and sanitation for all, has strong links to all other Goals (UN-Water 2018). The initiative also recognises that water scarcity is becoming more commonplace, aggravated by climate change, while pollution is increasing, and natural ecosystems are under growing pressure. Successfully managing water resources is a long-term, unceasing commitment. At the 1992 Earth Summit, countries agreed that the best way to meet that challenge was to adopt integrated approaches for the management of water resources. The importance of IWRM was underscored in passing years and considered central to the United Nations 2030 Agenda for Sustainable Development, as it provides an essential framework to achieve not only SDG 6 but all Sustainable Development Goals. Therefore, SDG target 6.5 calls for implementing by 2030 IWRM at the national and subnational levels, including through transboundary cooperation as appropriate (UN-Water 2018).

The degree of implementation by individual countries is measured by the 2030 Agenda through indicators in four different dimensions: the existence of an *enabling environment* of policies, laws, plans, and arrangements; *institutional frameworks* in place that ensure cross-sectoral coordination, the participation of the private sector and other stakeholders and the fulfilment of gender objectives; the existence of *management instruments and programmes* for informed decision-making, covering water availability monitoring and sustainable water use, pollution control, water-related ecosystems and disasters, and data and information sharing; and the level of *financing* for investments, including infrastructure, recurring costs, and revenue raising.

The 172 countries included in the 2018 report were individually scored for each of these four dimensions, as well as for an overall IWRM score; scores for each of these dimensions and for the overall IWRM then fall in one of six categories regarding the extent of their implementation, from very low to very high (Fig. 18.1). As noted by Bennison et al. (2019), Chile ranked poorly in terms of the overall IWRM score (159th out of 172 countries, percentile 8). Argentina also ranks below the median (105th out of 172 countries, percentile 39). These overall scores result from scores within the 'low' class and the 'medium-low' for all four dimensions for Chile and Argentina, respectively. Considering that the first two dimensions are more related to water governance and the third to science and technology applied to water research and management, a direct conclusion is that both countries are not doing very well regarding either science or governance. As a reference, Brazil significantly outscores Chile and Argentina, ranking well above the median for the overall IWRM score (68th out of 172 countries, percentile 60).

Interestingly, the better scoring of Brazil is based largely on dimensions more related to governance. This comes as no surprise; as early as the 1960s, severe problems of water resource degradation led to a shift in perspective on the ways of dealing with water appropriation and management in Brazil (Trimble et al. 2021b). A series of discussion forums involving different actors were created during the 1980s that concluded that a national system of water resources was needed (Pagnoccheschi

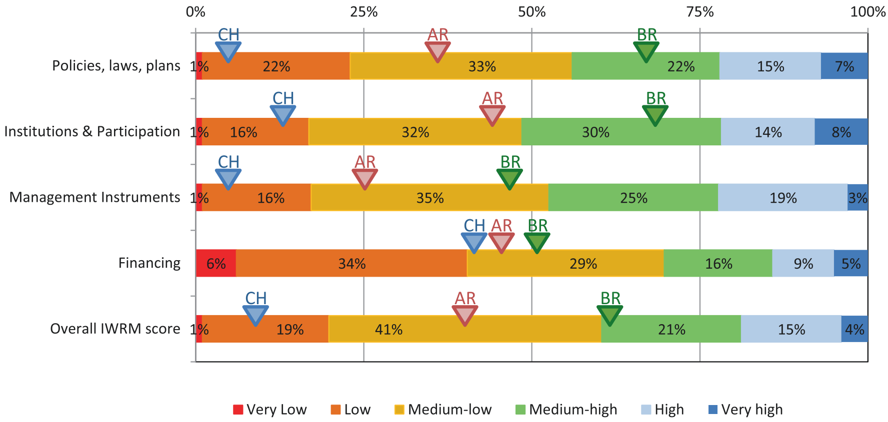

**Fig. 18.1** Degree of implementation of integrated water resource management (IWRM) by 172 countries (SDG 6 Indicator 6.5.1) according to UN-Water. The bars show the percentage of countries in each of six implementation categories (from 'very low' to 'very high'), across the four main dimensions of IWRM and an overall implementation. Specific implementation scores for Chile (CH), Argentina (AR), and Brazil (BR) are shown. (Adapted from UN-Water (2018))

2003). These recommendations were eventually included in the 1988 Constitution and led to the promulgation of the Water Law in 1997. Today Brazil has a governance system that accommodates the federal logic of unifying principles and procedures, with the specific realities and expectations registered at the subnational level. The management of freshwaters recognises river basins as the territorial units for planning and implementing policies (thus indirectly including land-use management) and is backed by basin committees with the participation of multiple stakeholders, basin plans, and financial instruments to conduct those plans (Trimble et al. 2021b, 2022).

On the other hand, the three countries do not score that differently in terms of financing. This suggests financial constraints are not the central obstacle for the implementation of IWRM in Chile and Argentina, but limitations are more related to the legal, regulatory and institutional frameworks and the existence of specific management instruments and programmes. Main reasons for this are, in Chile, the free-market control of water policies (Blanco et al. 2020) and, in Argentina, the hyper-federalised political system, with weak federal norms and guidelines for freshwater management (Berardo et al. 2013). In fact, participatory governance structures for freshwater management are rare in Chile and Argentina. Water management has historically followed a technocratic approach based on the prediction and control paradigm (Trimble et al. 2021b), highly dominated by an engineering view directed at providing mere hydrologic and hydraulic solutions to mitigate the effects of environmental deterioration, and within a tradition of natural resource management as a process that is top-down, closed-door and bureaucratic.

In Chile, the elaboration of a new Constitution is under way, providing a unique opportunity to advance freshwater management. The need to rethink how freshwater is conceived and managed is a key aspect considered by one of ten commissions that make up the Constitutional Convention. Proposals incorporate the role of the State to guarantee water and sanitation as an inalienable right and its responsibility in protecting watersheds. The proposal is in line with a bill approved in 2021 by the Senate that limits the private control of water (Salgado and Fuentes 2021). It will be interesting to analyse the specificities finally introduced in the constitution regarding freshwater and to document to what extent and how federal legislation percolates to the quality of governance at the basin level.

In Argentina, there is no clear perspective of changes in legislation at the federal level that could produce a shift in paradigm such as the one that may take place in Chile, or that could contain some of the good properties of water governance of Brazil. Particularly important needs in this sense are the provisions for a national system that norms aspects of the governance of individual basins, such as participatory rules, or the elements that a basin plan should have, as well as its administration and funding schemes. Yet, those few basins wherein participatory processes involving stakeholders at all levels – one of the key pillars of IWRM – are in place, such as the Chubut River Basin (Trimble et al. 2022), offer opportunities for scrutinising and mainstreaming experiences, both positive and negative. Thus, practical examples and developments in selected basins could light the way for the adoption of new paradigms in Patagonia and Argentina in general.

As proposed in previous sections, the ecosystem service approach, in any of its variations, has a great potential to build bridges between disciplines and between science and freshwater management, both in Chile and Argentina (Pascual et al. 2022). Is the adoption of an ES approach compatible with the prevailing IWRM approach? And, more importantly, are there any conceptual or operational advantages in incorporating ES perspective to prevailing views?

IWRM and ES are the two conceptual approaches that presently dominate water resource research and management at a global level and that have independently evolved into nearly identical concepts (Cook and Spray 2012). Although IWRM is a water management approach and ES is a general ecosystem management approach, both have some pivotal aspects in common: the recognition of the importance of land-water interactions, the consideration of the well-being of recipient communities with equity, and the overarching importance given to ecosystem sustainability for freshwater and wetland conservation and management. Also, both recognise the complexity of human-environment relationships and consider that competing interests must be integrated in order to successfully implement policies. From a practical point of view, while IWRM is more paradigmatical, ES allows for normative products and applications, by explicitly quantifying the flow of ecosystem services, with a geographic recognition of provision areas and reception areas, and considering the trade-offs among ecosystem services, among groups of stakeholders and among geographies within a basin. Therefore, the assimilation of the ES approach can strengthen IWRM, reinforcing the environmental axis in the management of water resources; broadening the analysis of the links between terrestrial, aquatic, and wetland ecosystems; providing technical tools to map and quantify the flow of ecosystem services; setting up a common platform for information stemming from multiple disciplines; and fostering the objective evaluation of nature-based solution as an alternative to solve water problems (Pascual et al. 2022). Promoting green infrastructure solutions is a particularly important goal, of great conceptual importance. Green infrastructure can be regarded as the boundary product between healthy ecosystems and human health. As such, it serves not only SDG 6, but it can clearly advance SDG 3 (clean water and human health), SDG 11 (sustainable cities and communities) and SDG 12 (responsible production and consumption), and it is central to SDG 15 (protection, restoration and sustainable use of inland ecosystems and their services).

Meanwhile, the science and technology systems of both Chile and Argentina must generate the kind of science required to provide a robust technical background to freshwater management under the IWRM and the ES approaches. In Argentina, a scientific network for the evaluation and monitoring of aquatic systems (REM. AQUA) has been recently created by the National Council for Scientific and Technical Research (CONICET) and the Ministry of Environment and Sustainable Development (CONICET 2022). Initiatives of this kind require a considerable effort over an extended time lapse and thus must be regarded as state policies, with a continuing commitment from government agencies at municipal, provincial, and national scales towards financing their permanence and applying their findings to nature-based management alternatives.

In summary, the current pace of climate change and human impacts on our planet provides a very well justified sense of urgency for the adoption of renewed, bold decision systems to meet the challenge posed by SDG 6 – the availability and sustainable management of water and sanitation for all – and other water-related SDGs. This is particularly true in countries that, like Chile and Argentina, are naturally rich in freshwater resources but clearly lagging behind in the adoption of more integrative views for their management. The challenge requires both reviewing the freshwater science we do and rethinking the governance of natural resources in our countries. Integrating the applied products of the Ecosystem Services paradigm to the Integrated Water Resources Management would be a major step in the right direction to do so.

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