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Chapter 13

Wetlands in the *Campos Sulinos*: Diversity, Functions, and Threats



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13.1 Introduction

Wetlands are among the most productive ecosystems on the earth, with enormous ecological and social importance. Wetlands in their natural state provide substantial benefits to the sustainability of a region (Batzer and Sharitz 2014). Water storage and purification, flood protection, aquifer recharge, grain production, energy and recreation are some of the various functions these ecosystems provide for humanity. Costanza et al. (2014) estimated that services and goods provided per unit area of wetlands (US\$14,785 ha⁻¹ year⁻¹) exceed the values provided by the world's forests (US\$232 ha⁻¹ year⁻¹).

Wetlands are ecosystems of high biological diversity (Batzer and Sharitz 2014). Estimates indicate that 10% of the identified species of the planet (130,000 species) occur in continental wetlands, although these habitats cover less than 1% of the earth's surface (Dijkstra et al. 2014). Such high species diversity is a consequence of the different wetland types that exist in the world, from marine to coastal to continental, including the human-made ones (Cowardin et al. 1979). Furthermore,

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habitat variations such as depth, aquatic plant types, size, and hydroperiod also contribute to the high species diversity of these ecosystems (Guadagnin and Maltchik 2007; Rolon and Maltchik 2006; Stenert and Maltchik 2007).

The diversity of wetland types makes it difficult to develop a single definition that encompasses all wetland ecosystems (Maltchik et al. 2018). Therefore, researchers suggest that the basic definition of wetlands is related to their key attributes: presence of water, hydromorphic soils, and water-adapted biota (Cowardin et al. 1979). In Brazil, Junk et al. (2014) proposed a definition based on those attributes. In addition, the authors proposed the first hierarchical classification of Brazilian wetlands. This classification holds as a key element the fluctuation of the water level, which essentially discriminates the large wetlands of the Amazon and Pantanal from the small-sized ones of the southern region of Brazil (of the South region). Despite their small surface areas, southern Brazilian wetlands provide similar ecosystem functions and services as the large, tropical wetlands. Indeed, global meta-analysis covering over 300 wetlands across 51 countries indicated that small wetlands provide a greater range of ecosystem services than large wetlands (Chaikumbung et al. 2016).

In southern Brazil (Rio Grande do Sul state), most wetlands are small-sized, although accurate inventories are lacking. Maltchik et al. (2003) identified 3441 wetlands across the state, and approximately 40% of inventoried wetlands had surface areas smaller than 5 ha (Maltchik et al. 2003). Across the *Campos Sulinos*, the general characteristics of the wetlands strongly vary between the Atlantic Forest and Pampa regions (“biomes” according to IBGE 2019) that comprise this region. In the Atlantic Forest (highland grasslands), wetlands are smaller, shallower, more isolated, mostly temporary, and with more than one dry period over a year (Knauth et al. 2018; Pires et al. 2018). In the Pampa (Pampa grasslands), wetlands are larger, deeper, non-isolated, with a strong reduction of surface water in summer and – when intermittent – with only one drought per year (Knauth et al. 2019; Bertuzzi et al. 2019; Bacca et al. 2021). Species diversity of several taxa is high in wetlands across both regions.

Despite their high biological diversity and the multitude of ecosystem services provided, wetlands in the *Campos Sulinos* are strongly threatened. Although detailed data on wetland loss are scarce, earlier estimates suggested that over 90% of the existing wetlands are either lost to or impacted by anthropogenic activities, especially conversion to agriculture (rice fields), urban areas, and tree plantations (Maltchik et al. 2003). Rice fields have a strong impact on the diversity and community structure of many wetland species, e.g., invertebrates, plants, tadpoles, and waterbirds (Moreira et al. 2016; Maltchik et al. 2017), and so do pine plantations (Rolon et al. 2011; Stenert et al. 2012). In terms of conservation, this situation is difficult to reverse due to the economic importance of both activities in the region. The state of Rio Grande do Sul contributes with over 65% of the total Brazilian rice production, while forestry activities have been strongly sponsored since the 1960s and remain in expansion in the region (see Ribeiro et al. 2020). However, the overall number of protected areas in the *Campos Sulinos* is low (less than 3% in the case of the Pampa), and even smaller when wetland protection is considered. Currently, two

Ramsar wetland sites are legally protected by Brazilian environmental laws in the Pampa grasslands: Lagoa do Peixe National Park and Taim Biological Station. Both Ramsar sites are of enormous importance to the aquatic biodiversity of the Pampa grasslands, especially endemic and migratory waterbird species. Ribeiro et al. (2020) showed that the inner areas of both Ramsar sites are well conserved in terms of natural cover and are currently not much subjected to anthropogenic pressures.

In this chapter, we cover most of the recent knowledge produced in relation to the wetland ecosystems of Pampa and South Brazilian highland grasslands, i.e., the *Campos Sulinos* region. We review the main drivers of the biodiversity of small wetlands; the contribution of specific land-management practices to wetland-dependent wildlife; the importance of dispersal processes to the biodiversity of isolated wetlands; the occurrence of endemic and endangered annual fish species; and the importance of restoration initiatives in impacted wetlands. Finally, we provide some suggestions that provide relevant information to wetland conservation and management in the *Campos Sulinos*.

13.2 Drivers of the Biodiversity of Small Wetlands and Wetland-Dependent Wildlife: The Hydroperiod Leads, but Space and Traits Also Matter

Considering the threatened wetland status in the *Campos Sulinos* (Maltchik et al. 2003), understanding of the drivers of the biodiversity of the wetland-dependent wildlife is crucial to conservation policies in the region. The knowledge of the drivers of such high wetland biodiversity is incipient. This is because most previous research works on wetland species diversity are limited to inventories of specific taxa that either cover the range of the state of Rio Grande do Sul (RS) (Maltchik et al. 2009, 2010) or are embedded within biodiversity reviews of specific sub-regions (Becker et al. 2006; Bond-Buckup 2008). Studies that address the biodiversity drivers and the ecological dynamics of the small-wetland biota in the *Campos Sulinos* are more recent.

Hydroperiod (i.e., the length of the wet phase) is the main driver of wetland biodiversity (Dodds and Whiles 2020). The differences in local species composition commonly observed between permanent and temporary wetlands is attributed both to the role of drought (that filters out species unable to cope with dry periods) and to the varying environmental conditions (e.g., habitat structure, water chemistry) and biotic interactions (e.g., predation rates) that occur in wetlands along different hydroperiods (Wellborn et al. 1996). The climate in southern Brazil is characterized by hot summers. Surface water of smaller wetlands tends to totally or partially disappear in the warmer periods of the year, and thus most freshwater wetlands alternate flooded (usually from fall to spring) and dry periods (summer). This creates a mosaic of temporary wetlands with different hydroperiods coexisting with larger, permanent wetlands across the landscape (Maltchik et al. 2004). In accordance with

the general expected pattern, earlier space-for-time studies evidenced that the taxonomic composition of multiple taxa (e.g., algae, macroinvertebrates, plants) differed between temporary and permanent wetlands (Maltchik et al. 2010; Rolon and Maltchik 2006; Stenert and Maltchik 2007). Nevertheless, beyond the coarse comparison between temporary and permanent hydroperiods, it has been increasingly recognized that biodiversity patterns in temporary wetlands can vary according to more fine-grained changes in hydroperiod. In specific, temporary wetlands differ in relation to various hydrological aspects, e.g., the number of dry and wet months, inundation timing, and filling rates. Subtle variations in those aspects of the hydrological regime can lead to important changes in wetland biodiversity (Jeffries et al. 2016).

In the highland grassland region, wetland habitats vary according to the lengths of their flooded periods, onset of their wet phases, and filling rates, leading to a landscape composed of short- to long-hydroperiod (i.e., permanent) wetlands (Pires et al. 2021) (Fig. 13.1a). Recent studies showed that such differences in hydroperiod length were important drivers of the richness and composition of animal communities in the region (Knauth et al. 2018; Pires et al. 2021; Stenert et al. 2017). Specifically, higher diversity of microcrustacean assemblages (Cladocera) hatching was detected in the sediment of medium- and short-hydroperiod wetlands compared to long-hydroperiod ones, suggesting that reduced hydroperiods stimulated the production of dormant stages (Stenert et al. 2017). Long-hydroperiod wetlands supported higher richness of dragonfly larvae (Odonata, Insecta), while larvae with life-history traits related to survival to dry periods (e.g., Lestidae) were more common in shorter hydroperiods (Pires et al. 2019).

There was also strong spatial and seasonal variation in the local environmental conditions (e.g., vegetation structure, water chemistry) and predator presence among hydroperiods (Pires et al. 2021). The hydroperiod gradient thus likely drove the strength of biotic interactions that underlie the observed patterns in assemblage structure of wetland invertebrate and vertebrate taxa in highland grasslands. For instance, the complexity of the surrounding vegetation was directly associated with longer hydroperiods and explained the higher diversity of spider assemblages in wetlands with more complex habitat structure (Ávila et al. 2017). Fish presence also co-varied with hydroperiod (fishes were absent in reduced hydroperiods). The potential stronger predation pressure exerted by fishes was suggested as the driver



Fig. 13.1 (a) Small temporary wetland in the highland grassland region; (b) wetlands in the Pampa region; and (c) wetland fragmented by rice field expansion. (Photos: Leonardo Maltchik)

of the lower tadpole richness found in the long-hydroperiod wetlands (Knauth et al. 2018) as well as the distinct taxonomic and functional compositions of macroinvertebrates reported between fish and fishless wetlands (Boelter et al. 2018). Finally, there is strong evidence that changes in hydroperiod length interfered in the meta-community dynamics of aquatic taxa in the region. Spatial location and environmental variables had varying influences on the composition of aquatic macroinvertebrates in each hydroperiod over the seasons: while environmental variables explained the overall metacommunity structure in colder seasons, spatial variables prevailed in the spring. However, active- and passive-dispersing macroinvertebrates were distinctly influenced by spatial and environmental variables in each season. This suggests that hydroperiod length differently affects the metacommunity dynamics of passive- and active-dispersing macroinvertebrates (Pires et al. 2021).

In the Pampa grasslands, a region warmer and less humid than the highland grasslands, small wetlands coexist with much larger remaining wetland fragments (Junk et al. 2014) (Fig. 13.1b). In the coastal region of the Pampa, previous studies showed that intermittent dune wetlands supported lower richness and distinct composition of macroinvertebrate and macrophyte taxa compared to permanent ones. Specifically, permanent wetlands supported a higher proportion of predatory macroinvertebrate taxa and hydrophyte species, while no species discriminated the intermittent wetlands (Rolon et al. 2008; Stenert and Maltchik 2007). This suggests that wetlands with longer hydroperiods favor the occurrence of long-living species with adaptations to wet conditions, while taxa inhabiting intermittent wetlands also occur in permanent ones. Wetland hydroperiod was also important to explain anuran distribution in the Pampa. Although richness and composition of adult and tadpole anurans also differ between temporary and permanent wetlands, again, as in the case of the highland grassland wetlands, incorporating finer-grained information on hydrological aspects of wetlands highlights a more complex scenario of biodiversity structuring (Moreira et al. 2010). Specifically, the number of dry months was an important predictor of anuran composition and adult abundance; permanent and temporary wetlands also differed in relation to anuran diversity over the seasons. Overall, this suggests that anuran biodiversity is differently structured in temporary and permanent wetlands, likely through the role of adult movement patterns.

Considering the dependency of wetland hydrology and of the life cycles of many wetland-dependent organisms (e.g., plants and ectothermic animals) on precipitation and temperature regimes (Jackson et al. 2014), climate patterns could be viewed as an additional driver of wetland biodiversity through impacts on hydroperiod and species phenology. So far, most evidence of the potential influence of those regional drivers on biodiversity of wetlands in the *Campos Sulinos* stem from studies in the coastal wetlands of the Pampa. For plants, seasonal variation in precipitation leads to local extinction and/or dormancy of submerged and free-floating species in drier periods of the year (Rolon et al. 2008). Conversely, wetland plant communities in areas with higher seasonality in precipitation are increasingly dominated by amphibious species, more tolerant to longer dry-period lengths (Bertuzzi et al. 2019). Invertebrate composition also shifted according to climate characteristics of the coastal wetlands of the

Pampa grasslands. Specifically, areas with warmer and less seasonal variation in temperature showed higher dissimilarity in spider and insect composition compared to the colder and more seasonal regions (Ávila et al. 2020; Bacca et al. 2021). This is likely because species with phenological and ecophysiological traits more associated with higher temperatures (e.g., spiders with restricted movement patterns and insects with low thermoregulation capacity) were less frequent in the wetlands with colder and less-seasonal temperatures (Pires et al. 2018; Ávila et al. 2020).

In addition, given the “discrete” nature of freshwater wetlands, the spatial positioning and the isolation of wetlands are believed to be strong predictors of their biodiversity (Jeffries et al. 2016). Higher richness of waterbird species was detected in coastal wetland fragments located in landscapes with greater connectivity and matrix permeability than isolated wetlands located in unsuitable landscapes (Guadagnin and Maltchik 2007). Increasing isolation of wetland fragments was also negatively related to macrophyte richness (Rolon et al. 2012) and higher dissimilarity in the composition of aquatic insects in the Pampa grasslands (Bacca et al. 2021; Pires et al. 2018).

The key ecological factors driving the wetland biodiversity across *Campos Sulinos* include the length of hydroperiod (which underlies a series of local environmental modifications and biotic interactions within wetlands), the local climate regimes associated with wetland location, and the degree of isolation of each wetland. Although there seems to be few unique species of temporary and permanent wetlands, variations in wetland hydroperiod seem to drive the local biodiversity of the wetland-dependent wildlife by favoring the predominance (rather than species turnover) of different species along a hydroperiod gradient and, most important, through important changes in the ecological dynamics of single wetlands. In this context, species functional traits related to dispersal mode, tolerance to dry periods, and adaptation to local climate patterns play an important role in determining the number and the identity of the species present at a given moment in each wetland across the *Campos Sulinos*.

13.3 Wetlands and Land Management Practices: Did You Make the Best Choice?

Small freshwater wetlands that dry up for part of the year are often viewed as wastelands and easily overlooked. Given the small surface water area (<1 ha), these freshwater habitats are tightly bound to surrounding terrestrial environments. Historically, grasslands in the *Campos Sulinos* region have been shaped by grazing and fire. Nowadays, these grasslands are facing a rapid transformation to grow food crops and other land uses, with deleterious effects on biodiversity. Research published over the past 5 years makes the warning clear (Staude et al. 2018; Moreira et al. 2021). As grasslands are lost and biodiversity erodes, so will the quality of the wetlands in the *Campos Sulinos* and the associated ecosystem services (biological productivity, water quality maintenance).

Modification of surrounding land for human use, be it agricultural or urban, has an inherent effect on each wetland type (and associated biodiversity). So, it is unsurprising that pristine areas support higher levels of biodiversity than agro-pastoral lands. In Southern Brazil, small wetlands are commonly surrounded by extensive livestock farming, intensive crop-based agriculture (Fig. 13.1c), and afforestation. Reducing the impact of such land uses is a priority, not only for ethical reasons but also to integrate conservation and food production on the same land (i.e., land sharing). Relative benefits and drawbacks of rural land uses to wetland biodiversity largely depend on management practices. However, reconciling biodiversity conservation with economic gains is not straightforward. Synergetic effects of climate change and land use are already drying out many wetlands in Brazil (Project MapBiomass 2021a). But there is an opportunity to manage such issues. Progress can be made by involving strategies of land sharing and land sparing.

Aside from floodplains in the coastal region, wetlands in the *Campos Sulinos* typically exist as small remnants embedded in a larger matrix of farming. Roughly 40% of native grasslands remain in the portion corresponding to the Pampa, and the situation of highland grasslands in the Atlantic Forest is even worse (Project MapBiomass 2021b). Many of the remaining grasslands are used for extensive domestic animal grazing. Steered correctly, this land use promotes wildlife habitat and can preserve native vegetation. However, with sharply increasing prices for commodities (such as soybean, corn, and rice) during the last decade, the remaining grasslands of the Pampa are under substantial pressure for agricultural development. Evidence already shows that the conversion of natural grasslands into short-term crops promotes biodiversity erosion in its different dimensions (i.e., species richness, functional and phylogenetic diversity). Such erosion is consistent across primary producers, terrestrial consumers, and key consumers in wetlands (Stauder et al. 2018; Moreira et al. 2020; Saccol et al. 2022). In a closer look at amphibians, phylogenetic relatedness and reproductive modes mediate most responses to land use intensification (Moreira et al. 2021; Saccol et al. 2022). Species that deposit eggs directly in the water, without the protection of foam nests, are the most sensitive to grassland modification (Fig. 13.2a). Currently, there is growing consensus that extensive cattle grazing over large native pastures may provide an alternative



Fig. 13.2 (a) Frog species that deposit eggs directly in the water, without protection of foam; nests are susceptible to grassland modification (Photo: Leonardo F.B. Moreira); (b) annual fish embryo in dormant stage (Diapause III) (Photo: Vinicius Weber); and (c) *Austrolebias gymnoventris*, an annual fish species of temporary wetlands of Pampa grasslands. (Photo: Pedro Hoffman)

more compatible with conservation than short-term crops in wetlands of *Campos Sulinos* (e.g., Fontana et al. 2016; Staude et al. 2018; Moreira et al. 2020). After the conversion of natural ecosystems around wetlands to croplands, alterations of hydrological regimes and vegetation homogenization are the immediate impacts on freshwater communities. As time goes by, the footprint of agrochemicals tends to increase because many wetlands serve as long-term sinks on the landscape. But we should consider that the wet and dry phases of temporary wetlands are closely intertwined. A dry pond bed does not make it biologically dead. Even a dry bed deserves attention and conservation.

It is no secret that the production of desiccation-resistant forms is crucial for community dynamics in freshwater wetlands, especially temporary ones. Many wetlands have diverse propagule banks (e.g., seeds, dormant zooplankton, and killifish eggs) that can be sources for biodiversity recovery once habitat quality improves. As always, uncertainty remains. The contribution of this “storage” effect is dependent on the length of the dry phase, stressors originating on-site or off-site, and the ongoing climate change. Although studies often focus on how organisms respond to agrochemical application during the aquatic phase (Stenert et al. 2018; Azambuja et al. 2021), it is important to acknowledge that intensive land management can also impact wetlands during the dry period. While the studies considering how wetland-dependent wildlife is modulated by land management are in their infancy in Southern Brazil, they already point lower thermal tolerance – and other deleterious effects – in dormant propagules, larval stages, and adults (Babini et al. 2015; Zebral et al. 2018). Going forward, we must remember that the compounding impacts of heat events and droughts are expanding dry periods in temporary wetlands. Landowners and the scientific community should jointly take action to understand which land uses have more pervasive impacts and which uses help to supplement biodiversity and boost revenues.

There is still much to know about the consequences of the transformation of freshwater wetlands to grow food crops, but rice fields are sometimes considered supplementary habitats for many aquatic species in farmland landscapes (Guadagnin et al. 2012; Maltchik et al. 2017). In the absence of any environmental guidance or recognition of the flow of ecosystem services, farmers often jeopardize essential regulating services to increase the supply of provisioning services, such as rice. So, how can we do better? The conversion of wetlands to rice fields in the *Campos Sulinos* region occurred with little consideration of their impacts, yet some management practices provide promising models. Organic farming, wet fallow, and maintenance of original vegetation around field edges are important tools that help to mitigate the impacts of wetland conversion to rice cultivation (Machado and Maltchik 2010; Moreira and Maltchik 2014). Still, strong bottlenecks impair the adoption of more sustainable land management practices, and biodiversity-friendly agriculture has been occupying a gray zone between science and politics. Criteria for loan concessions, tax subsidies, and organic food choice by the public are tricky matters when producers decide to adopt better land management practices (Schiesari et al. 2013). Two additional factors associated with conventional irrigated rice command attention. The soybean crop area in rotation with irrigated rice increased

threefold in the last 10 years (as a consequence of drainage practices; IBGE 2021). And the use of early and very early-cycle rice cultivars is increasing rapidly. These tendencies indicate a decrease in soil moisture and mosaics of microhabitats favorable to different wetland-dependent species. If the current scenario holds, not only habitat services (as open water in rice fields is critical, both as spawning and foraging site for many amphibians and fish) but also regulating services like local climate and maintenance of soil fertility would be compromised.

Land mismanagement has transformed many grassland environments, and associated wetlands, into exotic forest plantations. In the last 35 years, all states in southern Brazil have increments (between three to ten times) in the area used for tree monocultures (Project MapBiomass 2021b). Exotic trees of *Acacia*, *Eucalyptus*, and *Pinus* cover now around 5400 km² of grasslands only in the Pampa. Further, tree plantations favor the spread of invasive exotic species into the landscape – a dire reality in the coastal plain, even inside protected areas. It is well established that afforestation of open habitats triggers not only landscape changes, principally by shading, but it can also change hydrological cycles through an increase in the evapotranspiration rates (e.g., Jackson et al. 2005). Thus, exotic tree monoculture systems have a huge impact on many wetland species because of their thermal requirements and mobility. Animals like amphibians, reptiles, and aquatic insects require heat exchange from the environment to fulfil several body functions. In addition, they typically have small home ranges and move smaller distances than mammals or birds. Reports of the effects of tree plantations on the wetlands from the *Campos Sulinos* region have increased in the last decade (Stenert et al. 2012; Saccol et al. 2017), and negative outcomes include reduction in specie richness, changes in community structure, and local extinctions in several environments (Stenert et al. 2012; Kellermann et al. 2021). Today, we hear frequent debates about ideas such as the creation of conservation units – not just to protect mega-diverse fauna and flora but to enhance the integrity of watersheds – and indeed, communities and governments are emerging in the Brazilian states with this as a response mitigating to the effects of warming climate. Yet, it is sobering to see how crops and plantation forests occupy large areas inside the few integral protection areas encompassing grasslands and wetlands of southern Brazil (Ribeiro et al. 2021). The challenges ahead are formidable, but the future of wetlands in the *Campos Sulinos* passes through particular landowners and if they are willing to maintain grasslands, i.e., an intact surrounding of wetlands, or not. Of course, land-sparing strategies and formally protected areas are also important tools in ways forward. Every bit of avoided conversion matters.

Although extensive livestock farming seems to be the best choice for land sharing in the *Campos Sulinos*, there is still direct nitrogenous waste input into ecosystems and damage by trampling, besides changes in vegetation structure (Dala-Corte et al. 2016). Thus, grazing regimes and stocking densities are key features in the relationship between livestock and small wetlands. One common challenge for wetland restoration projects is figuring out what effects of land use (isolated or synergistic) they are even working with. Sustainable land management practices are always a work in progress, with answers that are not always definitive. Grazing by

cattle and horses has shaped South Brazilian grasslands since the seventeenth century, and thus the *Campos Sulinos* can be seen as a cultural landscape. Cultures change, and today, it's hard to acknowledge that there are two contradictory trends. Many landowners lack any cultural connection to the *Campos Sulinos*, and the values of this landscape, or its traditions. Today, we strive to change the culture of land use to be less intensive and to value the carefully managed cultural landscape. Science has provided a growing body of evidence and practical applications to wetlands and local land management in southern Brazil. Policies that foster enabling conditions are still a long way off, however. In short, one needs to consider that our land use dilemma doesn't have to be a devil's bargain.

13.4 Zoochoric Dispersal Promotes Biological Exchange Between Grassland Wetlands

Dispersal is the process of movement of organisms between habitats in the landscape. It is essential for the colonization of species in new habitats and for the dynamics of metapopulations (Brown and Lomolino 2006). The success of dispersal depends both on intrinsic features of the organism, such as its ability to move, and on landscape structure, such as the distance between favorable habitats and permeability of the matrix (Bowler and Benton 2005). Wetland organisms with high capacity for self-locomotion, such as birds and mammals, tend to have their dispersal facilitated among wetland habitats, even if they are isolated from each other. On the other hand, sessile aquatic species such as plants and some invertebrates depend on passive dispersal to move across the landscape.

Passive dispersal occurs when whole organisms or their diaspores are transported by a vector (Green et al. 2016). When the wind is the dispersal agent, the process is called anemochory, and it is more evident in plant species with seeds adapted to soar in the wind (Soomers et al. 2013). Dispersal by anemochory in wetland systems tends to present a random behavior, since it will depend on the diaspore characteristic, such as size and shape, and on the intensity and direction of the wind. In hydrochory, the water acts as a dispersal agent, and this process is relevant both for whole organisms and for their propagules (Soomers et al. 2013). Zoochory (Fig. 13.3) occurs when the transport vector is an animal. This process is traditionally divided into synzoochory, when for different reasons the vector intentionally transports the dispersed organism; epizoochory (or ectozoochory), when propagules or whole organisms are transported unintentionally attached to the external parts of the vector; and endozoochory, when this dispersal occurs through the animal's digestive tract (Green et al. 2016). Several species of animals act as vectors of passive dispersal in aquatic environments, and although there is indirect evidence for the role of amphibians (Vanschoenwinkel et al. 2008) and mammals (Vanschoenwinkel et al. 2011), waterbirds have special relevance (Green et al. 2016).



Fig. 13.3 Representation of the zoochory process promoted by waterbirds. Typically, a whole organism or its diaspore ends up attached to the outside of the animal or is ingested by the bird, being released and transported from area A to area B

Darwin (1859) regards some considerations about how waterbirds may act as agents of dispersal both by epi- and endozoochory, and from the mid-twentieth century onward, several studies, notably those carried out in the northern hemisphere, launched new perspectives on the role of birds in plant dispersal (Figuerola and Green 2002; Silva et al. 2021a, b). In South America, the first studies related to dispersal by endozoochory in wetlands were on waterbirds and in the Pampa grasslands (Silva et al. 2018, 2019, 2021a, b), demonstrating the importance of zoochory as a dispersal process among wetlands. Silva et al. (2018) found that whole plantlets of the angiosperm *Wolffia columbiana* (Araceae) survived the passage through the digestive tract of birds of white-faced whistling-duck (*Dendrocygna viduata*). This finding broadened the spectrum of how whole plants can be dispersed in wetlands of Pampa grasslands. Silva et al. (2021a) identified 2066 intact diaspores from 40 different plant taxa, including seeds of 37 angiosperms and diaspores of Lycophyta (*Isoetes* cf. *maxima*), Pteridophyta (*Azolla filiculoides*), and Charophyceae in fecal samples of five waterfowl. The authors found that diaspores of native amphibious and emergent aquatic plants were dominant in the fecal samples of waterbirds, but aquatic floating and terrestrial plants were also present.

Endozoochory by waterbirds also is particularly relevant to the dispersal of a wide range of non-flying aquatic invertebrates. Silva et al. (2021b) found 164 invertebrate propagules in fecal samples of waterbirds in the Pampa, including eggs of the Temnocephalida and Notonectidae, statoblasts of bryozoans, and ephippia of Cladocera. The authors identified 12 waterbird species contributing to the dispersal of invertebrates in the Pampa, and these numbers likely will be higher if the number of studied bird samples and bird species were higher. Recently, Barboza et al. (2022)

found that snails are also potentially dispersed by endozoochory by waterbirds in the Pampa grasslands.

Fish dispersal by waterbirds has often been an alternative that explains how some fish species colonized isolated waterbodies (Emmrich et al. 2014). Epizoochory of eggs historically always was cited as a possible mechanism to fish dispersal (Darwin 1859), although no studies provide solid empirical evidence validating such a process (Hirsch et al. 2018). Recently, one study raised the internal transport of eggs as an alternative to fish dispersal in wetland system. Silva et al. (2019) showed that eggs of some killifish species (Rivulidae, Cypriniformes) of the Pampa can pass through avian gut, with a rate of 1% of survival.

In the Pampa grasslands, large populations of teals, rails, storks, herons, ibis, and gulls make seasonal migrations or regional movements that can reach hundreds or thousands of kilometers (Blanco et al. 2020). The factors that regulate these movements are not clear, and although some species have well-defined annual migration periods, most of them probably migrate according to the variations in the regional hydroperiod, wetland availability, and food supply. Such characteristics can make the dispersal in wetlands of the Pampa less directional than that observed in temperate regions (Silva et al. 2021a). Considering the high ability of waterbirds to cover hundreds of kilometers in short periods and their capacity to disperse a wide spectrum of aquatic organisms by zoochory, these studies show the importance of waterbirds in the distribution of wetland species in the Pampa grasslands.

13.5 Annual Killifishes: The Endemic and Endangered Grassland Fish Species

Annual killifishes are small-sized and short-lived organisms that inhabit temporary wetlands in South American subtropical grasslands (Volcan and Lanés 2018). Annual fish developed adaptations to ensure the survival of their eggs in the sediment during dry periods (Furness et al. 2015). One of the most important adaptations is when fish embryos enter the dormant stage (diapause) and adjust their development accordingly to environmental conditions (Godoy et al. 2021) (Fig. 13.2b). Annual life cycles in fish are unique to two families: Nothobranchiidae (Africa) and Rivulidae (Neotropical Region).

Many annual killifish species from the Rivulidae family occur exclusively in temporary wetlands spread in landscapes highly impacted by livestock and agriculture (Lanés et al. 2018; Fig. 13.1a). The low dispersal ability, specialized life cycle, and restricted geographic distribution turn several annual killifish species critically endangered (Volcan and Lanés 2018). Therefore, annual killifishes are target-group and flagship species for conservation of temporary wetlands in South America subtropical grasslands.

In Rio Grande do Sul state, 40 species of Rivulidae can be found (Lanés et al. 2021; Volcan et al. 2021), belonging to three killifish genera: *Austrolebias* (34

species) (Fig. 13.2c), *Cynopoecilus* (5 species), and *Atlantirivulus* (only 1 species, unique with non-annual life cycle). The *Campos Sulinos* region thus is considered the world center for diversity and endemism of annual killifish, mainly of the genera *Austrolebias* and *Cynopoecilus* (Lanés et al. 2018). However, true diversity should be greater, since inventories and studies based on phylogenetic and molecular analyses indicate a high number of potentially new species not yet been described by science (Garcez et al. 2020): the existence of at least a dozen new species of annual killifishes not yet described in RS can be expected.

The annual killifish species found in the *Campos Sulinos* tend to be endemic to this region, although some species are shared with neighboring countries such as Uruguay and Argentina (Lanés et al. 2014). Species distribution is concentrated mainly in the Pampa grasslands, in wetlands belonging to the Laguna Patos-Mirim hydrographic system, and the Uruguay River basin. Among annual killifish species, only three (*Austrolebias botocudo*, *Austrolebias nubium*, and *Austrolebias varzea*) were found in highland grasslands.

Lanés et al. (2018) conducted the first broad scale study to understand annual fish distribution patterns in the *Campos Sulinos*. They showed that the species assemblage was strongly spatially structured, and many individual species formed discrete units. Furthermore, annual killifish occurrence was negatively associated with altitude at the landscape scale, and negatively associated with water depth and presence of predatory fish at the local scale. The authors also found that assemblage composition is spatially structured, and differed among hydrographic regions, with high influence of altitude, temperature, and monthly precipitation.

13.6 Restoring Biodiversity and Ecological Services in Degraded Wetlands: Now Is the Time

We are at the beginning of the United Nations Decade on Ecosystem Restoration (2021–2030), which is an international appeal to the urgent need to restore natural ecosystems degraded by human activities. In the current and growing scenario of environmental degradation, wetland systems have been widely lost all over the world. Studies estimate a loss of wetland area of around 64% worldwide (Davidson 2014).

Faced with this alarming scenario, many public policies aimed at wetland restoration have emerged in recent decades (Mitsch and Gosselink 2007; Moreno-Mateos et al. 2012). For example, wetland restoration is already an established concept in national and global climate change mitigation strategies (Society for Ecological Restoration 2004) and was listed as one of the relevant topics at international conferences on wetlands. Strassburg et al. (2020) identified global priority areas for ecosystem restoration across all terrestrial biomes, and they estimated their benefits and costs. These authors showed that among the different types of ecosystems (forests, natural grasslands, shrublands, and arid ecosystems), wetland restoration has

the highest relative importance for biodiversity conservation. In North America, more than \$70 billion has been spent restoring 3 million ha of wetlands over the last decades (Copeland 2010). In China, a total of 1369 km² of wetlands were restored from agricultural areas between 1990 and 2010 (Mao et al. 2018).

The earliest wetland restoration activities were focused on restoring a specific ecosystem function. Nowadays, restored wetlands are intended to reestablish a variety of ecological attributes including community structure (species diversity and habitat) and ecosystem processes (energy flow and nutrient cycling), and the broad suite of goods and services to the landscape (Spieles 2022). However, wetland restoration is challenging because it must be guided by an understanding of the main ecological processes that structure the aquatic communities (Spieles 2022). There is a wide variety of different types of wetlands, with distinct characteristics, hydrology, vegetation, and soils, such as floodplains, tidal marshes, peatlands, depressional wetlands, mangroves, forest wetlands (Junk et al. 2014), which makes the restoration of these ecosystems even more challenging. Therefore, the successional trajectories and attributes can be highly variable among wetland restoration projects.

Restoration efforts related to wetlands have been initiated around the world for different purposes such as water quality improvement, biodiversity enhancement, floodwater control, carbon sequestration, and aquifer recharge (Comín et al. 2014; Craft 2016). However, little focus has been given to the recovery of functional processes in wetlands (Cabezas et al. 2009), such as primary production, predator-prey interaction, decomposition, and herbivory (Español et al. 2015). Wetland recovery activities have been achieved using native species and eliminating the causes of degradation (Hughes et al. 2018), but the recovery capacity is greatly affected by abiotic factors such as the size of the restored ecosystems, connectivity to other aquatic systems, and climate (Moreno-Mateos et al. 2012).

A common approach to restoring wetland ecosystems is to introduce species important for restoring ecosystem functions. Some methods used to accelerate the succession and ecosystem development of restored wetlands consist of adding soil nutrient and organic matter and vegetation propagules (seeds, fragments – rhizomes, seedlings) (Craft 2016). Among the methodologies used in wetland restoration, the translocation of sediment containing seed banks and dormant eggs of aquatic invertebrates from a “donor wetland soil” has been shown to be very efficient in the recovery of plant and invertebrate communities in ecosystems degraded by human activities (Richter and Stromberg 2005).

The surface layer of natural wetland sediment (5 cm) can contain 2000–50,000 viable seeds of different aquatic plant species per 1 m² (Burke 1997) and 10³–10⁶ eggs per 1 m² of zooplanktonic species (Hairston 1996). De Stasio (1990) found between 150,000 and 400,000 resistance eggs of the copepod Calanoida *Diaptomus sanguineus* per 1 m² per year. Thus, the seed and egg banks existing in the sediment of natural wetlands constitute a fundamental ecological and evolutionary reserve for the recovery of biological communities in these ecosystems (Brendonck and De Meester 2003; Jenkins and Boulton 2007).

Recently, two initiatives to restore degraded wetlands were carried out in the Pampa grasslands (Vendramin et al. 2021; Silva et al. 2022). In both studies,

techniques of topsoil translocation from natural wetlands were used to test the zooplankton recovery in sediments of rice fields and wetlands degraded by mining activities. Both studies analyzed if the increase of topsoil addition from natural wetlands in the sediment of cultivated (rice field) and mined wetlands (clay mining) influenced the hatching of zooplanktonic organisms from their dormant stages. For practical purposes, the results showed that some natural wetlands may not be good donors of topsoil sediment to restore areas subjected to mining activities (Vendramin et al. 2021), that is, the quality of donor wetlands (their topsoil, water, and habitat structure) should be evaluated before to start restoration projects. Sediments from natural wetlands with a more diversified and abundant bank of zooplanktonic dormant stages should be preferred for use in restoration projects.

An important goal of wetland restoration projects developed in the Pampa grasslands was to try to find the optimal amount of sediment needed to be removed from natural wetlands to restore degraded wetlands. This information is extremely important to reduce costs and encourage initiatives by producers in restoration projects. Silva et al. (2022) showed that 30% of topsoil addition from donor natural wetlands can lead to significant changes in the zooplankton composition in areas degraded by rice fields in a short-term experiment. The topsoil addition of 30% in the rice fields made the zooplankton composition resemble the composition of the reference wetlands. These results showed that the transplant technique of zooplankton dormant stages associated with topsoil can be a promising method to be used in the recovery of wetlands in the Pampa grasslands. Due to the effects of wetland habitat quality on the surrounding landscape, restoration of a wetland will also require restoration of adjacent ecosystems. Unfortunately, ecological restoration of grasslands in the *Campos Sulinos* region still is very much at the beginning and still limited in terms of available techniques (e.g., Guerra et al. 2020; Porto et al. 2023). Both more research and more implementation in practice are clearly necessary, also at the interface of terrestrial and wetland ecosystems.

13.7 Importance of Wetland Conservation in the *Campos Sulinos*

The wetlands of the *Campos Sulinos* present high biological diversity for different groups of organisms (invertebrates and vertebrates) and provide many benefits for the human population through ecosystem services (provision, support, regulation, and cultural). The conservation of the wetlands will guarantee the maintenance of a large part of the region's biological and genetic diversity and will provide natural resources to local human population (water to drink and to produce energy and grains, fish, firewood, clay to make tiles and bricks, and many others). In addition, the conservation of wetlands will provide leisure and recreation areas for the urban population, and minimize the negative impacts that floods bring to large- and medium-sized cities.

Studies related to wetland inventory and classification are necessary to support wetland conservation in the *Campos Sulinos*. These studies will provide maps of the exact location of the different types of wetlands present in the region and their conservation status. In addition, studies related to ecological restoration and environmental education will strongly contribute to the conservation of the wetlands of the *Campos Sulinos*.

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