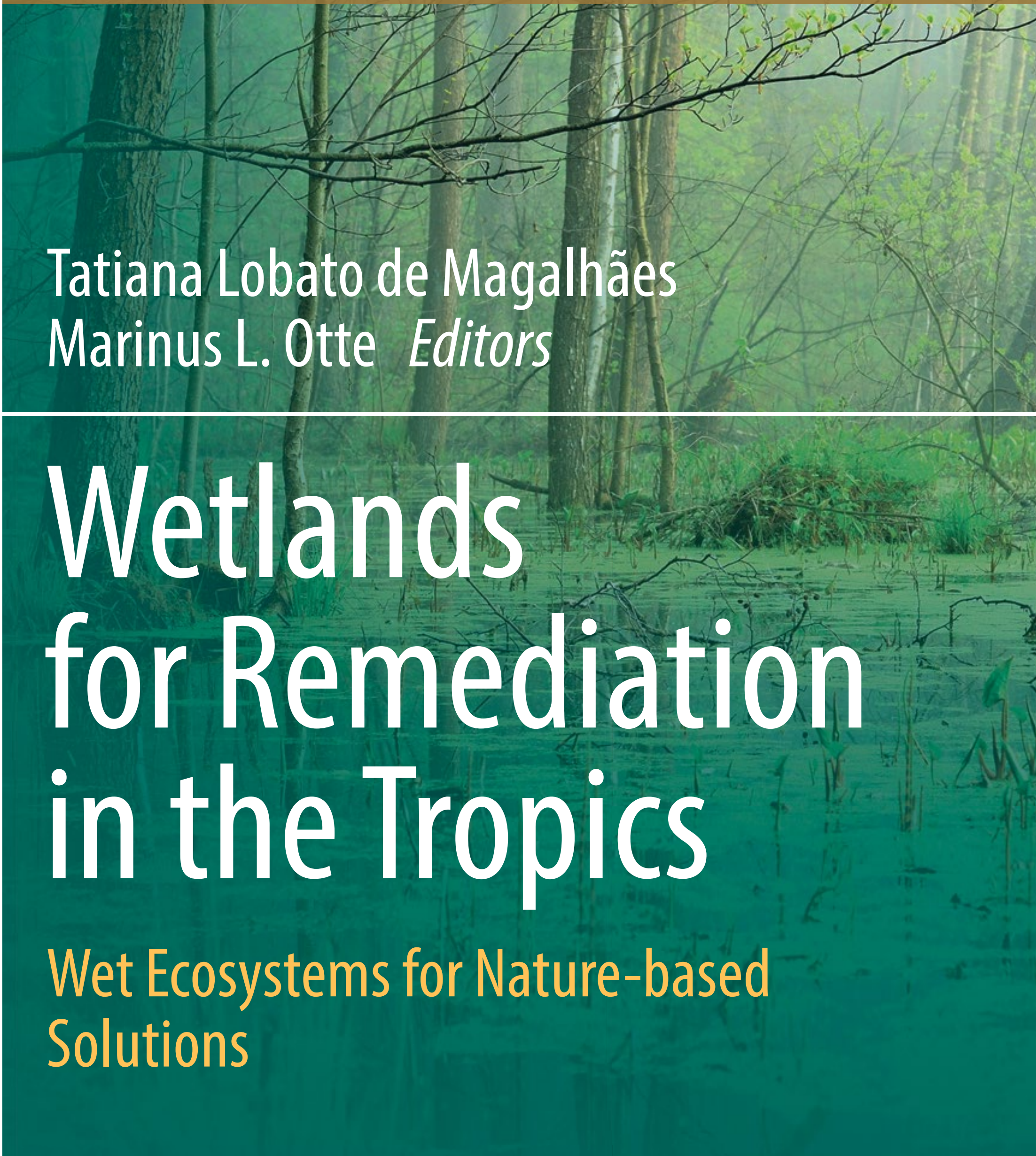
Wetlands: Ecology, Conservation and Management 9



Tatiana Lobato de Magalhães
Marinus L. Otte *Editors*

Wetlands for Remediation in the Tropics

Wet Ecosystems for Nature-based Solutions

Springer

Wetlands: Ecology, Conservation and Management

Volume 9

The recognition that wetlands provide many values for people and are important foci for conservation worldwide has led to an increasing amount of research and management activity. This has resulted in an increased demand for high quality publications that outline both the value of wetlands and the many management steps necessary to ensure that they are maintained and even restored. Recent research and management activities in support of conservation and sustainable development provide a strong basis for the book series. The series presents current analyses of the many problems afflicting wetlands as well as assessments of their conservation status. Current research is described by leading academics and scientists from the biological and social sciences. Leading practitioners and managers provide analyses based on their vast experience.

The series provides an avenue for describing and explaining the functioning and processes that support the many wonderful and valuable wetland habitats, such as swamps, lagoons and marshes, and their species, such as waterbirds, plants and fish, as well as the most recent research directions. Proposals cover current research, conservation and management issues from around the world and provide the reader with new and relevant perspectives on wetland issues.

Tatiana Lobato de Magalhães • Marinus L. Otte
Editors

Wetlands for Remediation in the Tropics

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Preface

Although the use of aquatic and wetland plants for phytoremediation of pollutants in wastewater is widely known in temperate zones of the world, information on the application in the Tropics is generally lacking. The book ***Wetlands for Remediation in the Tropics*** addresses this issue by providing a scientific review of remediation approaches utilizing aquatic and wetland macrophytes in the Tropics, including some subtropical regions. It covers theory, provides some case studies, and identifies gaps in our current understanding. It highlights the reasons why the Tropics differ from temperate regions in this context, particularly concerning differences in climate and species diversity and abundance.

The idea for this book arose from the following considerations:

(i) Wetlands for remediation are being used in the Tropics, but the large majority of the literature is from Temperate climates. Studies from the Tropics are usually underreported.

(ii) Most regions in the Tropics suffer from a lack of funding for environmental improvements. Because of the relatively low construction and maintenance costs, sustainable approaches to addressing environmental problems, such as wetlands for phytoremediation, are attracting increasing attention. The potential for widespread applications in the Tropics is enormous.

(iii) Water resources in the Tropics are under immense anthropogenic pressure, not just threatening the supply of usable water but also in terms of habitat for organisms. In many regions, wetlands are the last remaining strongholds of biodiversity. Increasing the use of wetlands for remediation in the Tropics therefore not only serves as a sustainable alternative for technological/industrial solutions for improvement of water quality but also compensates for habitat losses. This added benefit to ecological services is now weighed favorably in cost/benefit analyses. This, too, makes this volume of interest not only to scholars and academics but also to practitioners and government officials.

The book's first chapter focuses on the history of phytoremediation and the role of wetlands and aquatic plants in cleaning freshwater environments. The second chapter addresses questions about the differences between tropical and temperate biomes that will affect phytoremediation, to better apply lessons learned from the temperate regions to the Tropics. The subsequent chapters provide information on the principal aquatic plant species used for remediation in the Tropics, followed by a review of wetland applications for remediation in urban, rural, and industrial environments. Some of these contributions focus on specific geographical regions (e.g., the Americas, Africa, Pakistan, Taiwan, and other regions). The book concludes with one chapter providing a critical overview of the following steps needed to advance the use of wetlands for remediation in tropical regions through its costs and benefits. The book's authors comprise an international set of highly qualified scientists – from Argentina, Australia, Brazil, Canada, Colombia, the USA, the Netherlands, Mexico, Pakistan, and Taiwan – with substantial research experience using wetlands for remediation.

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About the Editors

Dr. Tatiana Lobato de Magalhães is a research professor at the Universidad Autónoma de Querétaro, QRO, Mexico, and has specialized in wetland research and education for more than 10 years, with a strong focus on ecology, genetics, and distribution of aquatic plants. Her work includes research projects and teaching across the Americas, including the Amazon and Southern Brazil Highlands. She was awarded the 2021 Academic Merit Medal by UAQ, the 2019 José Mariano Mociño (1757–1820) Medal by the Mexican Society of Botany, and the 2018 Research Fellow Wetland Ambassadors by the Society of Wetland Scientists. She is an associate editor of an international book series and scientific journals (*Wetlands*, *Aquatic Botany*, *Wetland Ecology*, *Management*, and *Conservation*), and chapter chair at the Society of Wetland Scientists. Tatiana is recognized by the National System of Researchers in Mexico (2020–2024) and certificated as a professional wetland scientist by PCPSWS (2020–2025). Details of her academic and professional career can be found at https://lobatomagalhaes.weebly.com/.

Dr. Marinus L. Otte is a professor at North Dakota State University, Fargo, ND, USA, and has specialized in wetlands research and education for more than 30 years, with a strong focus on pollution and phytoremediation. His research, teaching, and outreach on water and wetlands have involved projects around the world, most recently in Mexico as a Fulbright Specialist. He is a lifetime member of the Society of Wetland Scientists and has been the editor-in-chief of the scientific journal Wetlands since 2012. He published about 100 peer-reviewed scientific articles and contributed to many conferences worldwide.

Chapter 1
A Brief History of Phytoremediation Using Wetlands

Marinus L. Otte

Abstract Academic interest in wetlands and the use of plants for mitigating environmental pollution gained interest, particularly from the 1970s, but the use of wetlands and plants for addressing pollution goes back at least one century before that. Most wetlands used for the improvement of water quality are artificial and have been constructed around the world. Most of those are not the subject of scientific studies and therefore the number of scientific publications does not reflect the number of wetlands used for the purpose of environmental remediation. However, if scientific publications are an indication of the prevalence of wetlands around the world, it is striking that only 14% of all reports are from the tropics. This is despite the fact that the tropics are particularly suited for the use of wetlands for remediation of environmental quality, because of the relatively high and stable temperatures and availability of water. An explanation may be a general lack of legislation, funding, and education regarding wetlands in the tropics.

Keywords Nature-based solutions · Constructed wetlands · Treatment wetlands · Bioremediation · Pollution · Anthropocene · Tropics

1.1 Introduction

As the concerns about our environment grew during the second half of the twentieth century, highlighted by publications such as Rachel Carson's "Silent Spring" (Carson 1962) and the Club of Rome's first report "Limits to Growth" (Meadows et al. 1972), new directions of research emerged in order to find answers to questions, such as: How polluted is our environment? and How can we fix it? We now know that there are very few places in the world that are not polluted. Even the

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website of the Norwegian Polar Institute (2021) starts with the sentence, "The levels of pollutants in Antarctica are, in general, lower than elsewhere in the world." Lower, not absent, in what is arguably the remotest place on Earth. In addition, pollution from the era of the Roman Empire is detectable in ice cores from Greenland (McConnell et al. 2018). That island is not as remote from where the Romans once roamed as is Antarctica from everywhere else, but it shows that pollution from human activities has been spreading around the globe as long as humans have walked the Earth. It therefore can be stated that there are no places left on earth that are not affected by human impacts and that pristine ecosystems no longer exist. As we have come to realize how widespread alteration of our environment really is in this "Anthropocene" and that we cannot solve these issues with technological solutions, interest has shifted to "Nature-Based Solutions" (Anonymous 2017; Eggermont et al. 2019; Seddon et al. 2020). The idea to mimic natural processes in order to address environmental problems is not new but has gained momentum, particularly during the past 50 years. The 1970s saw the rise of interdisciplinary approaches to nature-based solutions, including the fields of phytoremediation and wetland science.

1.2 Phytoremediation

Phytoremediation is defined by Merriam Webster (2021) as: "the treatment of pollutants or waste (as in contaminated soil or groundwater) by the use of green plants that remove, degrade, or stabilize the undesirable substances (such as toxic metals)" and mentions that the first use of the term was in 1991. However, what is now understood to be research and practice related to phytoremediation started long before 1991.

The Italian philosopher Andrea Cesalpino in his book *De plantis libri* XVI, published in 1583 (Brooks 1998; Stephenson and Black 2014) has been credited with being the first to observe that only certain species of plants were associated with serpentine soils. The chemical composition of serpentine soils is very different from other soils, including the presence of high concentrations of metals like Ni, Co, and Cr but low concentrations of nutrients, so plants living on those soils have unique adaptations.

During the mid-twentieth century, plants that were able to grow on metal mine wastes started to attract attention, as it was observed that only a few species were able to grow on the typically nutrient-poor but metal-rich substrates. Pioneers in the field were researchers such as Prof. Tony Bradshaw (e.g., Bradshaw 1952; Gregory and Bradshaw 1965) and Prof. Wilfried Ernst (e.g., Ernst 1974). Bradshaw and Ernst led decades of research in metal tolerance in plants and applications in phytoremediation, well into the twenty-first century. They, as well as the people they trained and collaborated with, carried out groundbreaking work on the mechanisms of metal tolerance and population genetics (Antonovics and Bradshaw 1970; Ernst et al. 1992), hyperaccumulators (Ernst 2000; Van Ent et al. 2013) and ecological restoration (Bradshaw 1996, Lewis et al. 1996; Dobson et al. 1997; Ernst 2005). They showed that metal tolerance in plants is species as well as metal-specific, that metal tolerance comes at a cost, rendering metal-tolerant plants less competitive than non-tolerant ones of the same species, and that metal-tolerant plants can be used to revegetate metal-contaminated soils.

Metals were of particular interest because, being elements, they cannot be degraded. In addition, cases of extreme pollution with metals such as Cd causing "Minamata Disease" (Mcalpine and Araki 1958) and Pb affecting the development of children (Mitchell 1932; Millichap et al. 1952) had attracted much attention.

Other pollutants, including synthetic organic compounds, attracted attention much later, partly because their uses and their negative impacts really started increasing from the 1970s onwards and because analytical techniques to accurately assess them had yet to be developed. The 1970s saw rapid growth in research on pollutants in the environment, with a marked shift in focus to address not just direct impacts on humans but on entire ecosystems through a new field termed "ecotoxicology" (Truhaut 1977).

What Ernst (1974) called "Schwermetallvegetation" (heavy metal vegetation) was limited to plants growing on metal-rich substrates, such as natural outcroppings and mine sites that were above water. Nowhere in that book did the author mention wetlands. However, the work on plants associated with serpentine soils, as well as with mining and mine wastes was part of a wider interest within the scientific community regarding disturbance and stress and their effects on ecosystems, propelled by the work of Prof. Phil Grime who revolutionized the field of plant ecology with his ideas about plant strategies in responses to environmental conditions (Grime 1974, 1977). As the work was on the effects of pollutants on plants, the focus on the effects of disturbance and stress in plants was first on the vegetation of dry environments, but as research progressed, those ideas also carried through in the ecology of wetland plants (Hills et al. 1994).

Pollution of water and wetlands gained interest when it was realized that enormous amounts of pollutants were being transported down rivers and into the sea (see, e.g., Salomons and Förstner 1984). Soon, many reports followed on pollutants in sediments, including organic compounds (e.g., Moldenhauer 1996). Then during the late 1990s, it became clear that Ernst (1974) and other researchers from that period had overlooked an important group of plants: wetland plants turned out to be quite special regarding tolerance to high metal concentrations. In dryland (as opposed to wetland) plants, perhaps 1:1000 species were known to be tolerant to high concentrations of metals, but in wetland plants, metal tolerance appeared to be the rule rather than the exception (Ye et al. 1997a, b; McCabe and Otte 2000; McCabe et al. 2001; Matthews et al. 2004a, b, 2005a, b; Otte et al. 2004). It was also around that period that wetlands started gaining traction in the field of phytoremediation—the first issue of the International Journal of Phytoremediation contained a paper with the title "A Review of Processes Responsible for Metal Removal in Wetlands Treating Contaminated Mine Drainage" by Sobolewski (1999).

1.3 Wetlands Clean Our Water

Without humans realizing it, wetlands have been cleaning pollutants from water as long as wetlands have existed, and it took a long time before we started to realize the importance of all the benefits of wetlands. In ancient times, wetlands were perceived as good hiding places during wartime and as providers of medicinal plants, but also as smelly places that were not good for one's health (de Klerk and Joosten 2019). The most negative attitudes towards wetlands persisted throughout human history, but over the past century, perceptions began to change. Possibly the earliest scientific treatise on the benefits of wetlands for improvement of water quality was by Seidel (1966). Her paper titled "Reinigung von Gewässern durch höhere Pflanzen" (Cleaning of waters by higher plants) described not just the mineral contents of wetland plants but also their ability to remove organic, aromatic compounds from water. She already held two patents from 1964 to clean sludge (Seidel 1964a) and water (Seidel 1964b).

The United Nations Ramsar Convention of 1971 in particular raised awareness around the world of wetlands as habitats for birds (Ramsar 2014). That convention, as well as the ramifications of the implementation of the Clean Water Act of the United States in 1972 (US EPA 2021), led to a surge in research on wetlands. A search of the Web of Science (WoS) for the term "wetland*," the asterisk indicating a wild card for any character (Fig. 1.1), found the earliest record to be by Schluter (1910). A rapid increase starting from about 1970 continues to this day. As research

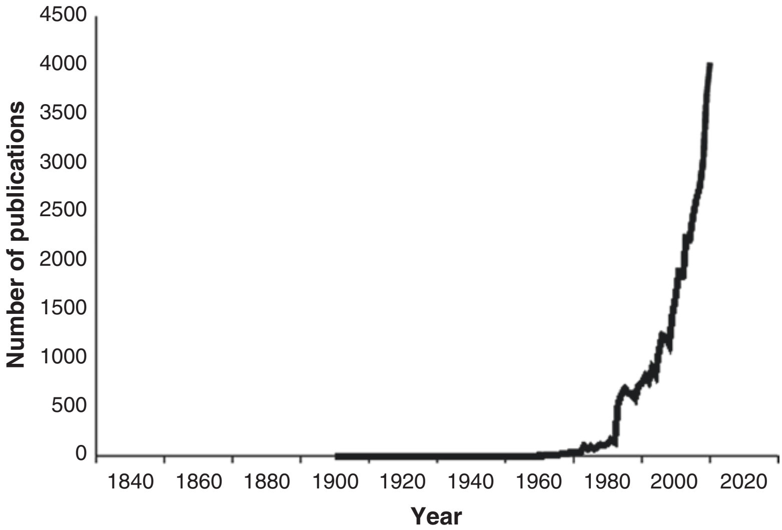

Fig. 1.1 Results from a search for "wetland*" in the title of articles in the Web of Science (WoS) on 6 October 2021. Cumulative number of publications between 1900 and 2020. The first article recorded in the WoS using the term wetland in the title is Schluter (1910), but the rate of publications with "wetland" or "wetlands" in the title increased noticeably from the 1970s

on wetlands expanded, it was further realized that natural wetlands were outstanding in their ability to remove pollutants from water (Kivaisi 2001; Otte and Jacob 2006; Zhang et al. 2015) and that they could therefore be constructed for that purpose as well as for providing other ecosystem services, such as flood control and habitat for biota. As research on wetlands in general increased after the 1970s, so did research on constructed wetlands, also referred to as treatment wetlands (Fig. 1.2).

The realization that wetlands clean our water started well before the 1970s though. As Brix (1994) mentions, it has been well known that ancient Egyptian and Chinese cultures used natural wetlands for disposal of wastewater. In fact, this is probably true for all ancient cultures because settlements tended to develop near shorelines and coasts for easy access to water, and wastewater would naturally drain to the lowest point in the landscape. However, constructing wetlands specifically for the purpose of treating wastewater is more recent. Harty and Otte (2003) describe what may be the first record of a wetland constructed for the purpose of cleaning wastewater, dating back to at least the 1880s, when the British Army constructed a system in Curragh Camp, County Kildare, Ireland. It was based on a treatment system originating from Germany, which converted domestic waste to fertilizer (Costello 1993). Brix (1994) referred to an account of a constructed wetland in Australia dating back to 1904.

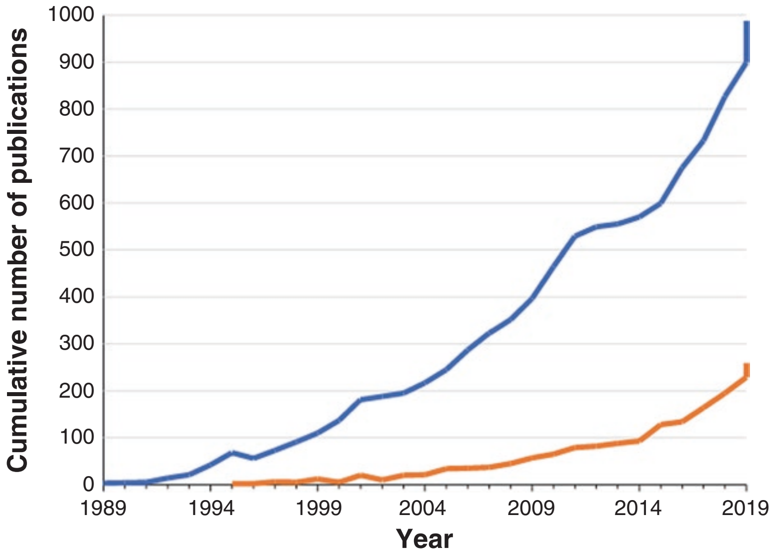

Fig. 1.2 Results from a search for "constructed wetland*" or "treatment wetland*" in the title of articles, review articles, or abstracts in the Web of Science (WoS) on 27 December 2021 (all records, blue) and 3 January 2022 (only for tropical countries, see Table 1.1, orange). Cumulative number of publications between 1989 and 2020. The first records are from 1989, two abstracts by Noordin et al. (1989) and Laudon and Wildeman (1989) and an article by Wieder (1989). The first record from a tropical country was from India by Billore and Das (1993)

The discrepancy between what is recorded in academic databases such as the WoS and anecdotal historic observations highlights the fact that wetlands had been constructed for all kinds of purposes already decades before the first mention of them in the scientific literature. This is because by far most projects to construct wetlands are carried out by entities whose main focus is not to produce scientific data but to achieve outcomes related to environmental issues, such as cleaning up water. By the 1980s, hundreds, if not thousands, of constructed wetlands already existed around the world. However, rigorous scientific research only started when more were being constructed, as questions about their designs and efficacy arose from the need for regulation by local, regional, or national authorities, for example, for the use of constructed wetlands as treatment of wastewater.

Over the past 50 years, more than 200 review papers have been written about constructed wetlands, one of the earliest by Gearheart (1992). Over the period January to September 2021 alone, 34 review papers on the subject had been published (WoS, 6 October 2021), including, Addo-Bankas et al. (2021), Parde et al. (2021), Varma et al. (2021), Vymazal et al. (2021), Zhang et al. (2021), and Zitacuaro-Contreras et al. (2021). Many books have been written about constructed wetlands too, including Kadlec and Wallace (2009) and Dotro et al. (2017).

The literature not only illustrates how widespread the use of constructed wetlands for water quality improvement is globally but also highlights the very wide range of substances that are effectively removed, from plant nutrients to metals to organic compounds, including hormones, as well as pathogens.

1.4 Wetlands for Phytoremediation

The definition of phytoremediation by Merriam-Webster cited above states that it involves "green plants that remove, degrade, or stabilize the undesirable substances," suggesting a direct role of plants in the processes that led to the removal of pollutants. However, in wetlands, their most important role is indirect, as they introduce oxygen into the substrate and provide the organic matter (and habitat) needed for microorganisms (see also Chap. 10) that drive the biogeochemistry involved in the removal and degradation of pollutants (Vymazal 2011). Uptake by plants typically accounts for only a small fraction of the total mass balance (Otte and Jacob 2006; Ventura et al. 2021). Plants may also volatilize organic compounds (Lin and Terry 2003; Limmer and Burken 2016), but it is unlikely that that route is a significant contribution to the overall mass balance in most cases (Otte and Jacob 2006).

The importance of plants in constructed wetlands raises questions about how performance is affected by seasons. Particularly in the northern hemisphere, winters can be cold and long. However, this does not mean that wetlands lose their ability to remove pollutants from water. Rates may slow down, and the cold may affect different pollutants in different ways, but many examples exist of wetlands, natural and constructed, efficiently removing pollutants also during the cold of winter (Mander and Jenssen 2003; Wang et al. 2017). So, what then makes wetlands for phytoremediation in the tropics special?

1.5 Wetlands for Phytoremediation in the Tropics Compared to Non-tropical Regions

Of course, low temperatures do occur in the tropics, but freezing temperatures that would significantly decrease the metabolic activity of plants and microbes and therefore the efficacy of constructed wetlands in the removal of pollutants occur on average at about 5000 m (Harris et al 2000). Very few people in the tropics live at elevations where freezing occur regularly, but there are exceptions. La Paz, Bolivia, with a population of about 770,000 is the highest city in the world at 3650 m above sea level and experiences below-freezing temperatures at night for a few hours from May to August (https://weatherspark.com/y/27348/Average-Weather-in-La-Paz-Bolivia-Year-Round, accessed 1 January 2022). Wetlands in La Paz would therefore experience climate conditions similar to those in temperate regions. However, about 40% of the world's population, so around three billion people, live in the tropics (https://worldpopulationreview.com/country-rankings/tropical-countries, accessed 7 January 2022), and the majority of those, by far, live at low elevations with the highest population densities (Cohen and Small 1998). Therefore, as anywhere in the world, the need for constructed wetlands for improvement of water quality in the tropics is highest at lower elevations where temperatures cold enough to affect the efficacy of the wetlands do not occur. The potential for constructing wetlands in the tropics is enormous (Kivaisi 2001; Zhang et al. 2015) and yet the number of reports on constructed wetlands from tropical countries in English is low compared to those from temperate regions (Table 1.1).

An analysis through the WoS of the articles in English with titles that contained the term "constructed wetland*" or "treatment wetland*" and that could be linked to particular countries returned a total of 14,127 results from 128 countries.

Table 1.1 Number of articles in English per country or region, according to the Web of Science (WoS) on 29 December 2021, for articles and reviews in all fields that contained the term "constructed wetland*" or "treatment wetland*," over the entire period covered by the WoS, 1800–present

Origin of articles	Number of countries	Number of articles	Articles per country
Global	128	14,127	110
Non-tropical/temperate	82	12,136	148
Tropical	46	1991	43
Tropical – Africa	19	248	13
Tropical – Asia	14	1131	81
Tropical – Latin America	13	612	47

Which country is tropical, and which is not is somewhat arbitrary because some countries are partly in and partly out of the tropics, for example, Mexico and India. For the purpose of this analysis, the countries listed by https://worldpopulationreview.com/country-rankings/tropical-countries were considered tropical. Some countries within the tropical zone that were not listed by that website because no data were available, but that clearly are in the tropical zone, were included as tropical, such as South Sudan, Cote d'Ivoire, Yemen, and Oman

On average, all countries globally published 110 articles per country, but in the non-tropical countries, the average per country was well above that average, 148, whereas tropical countries published much less than the global average, between 13 and 81 articles per country. The total number of articles globally from the tropics was 1991 articles, which is 14% of the global total. The average of 81 articles per country for Asia is strongly skewed by India, from which 443 articles originated, about 39% of all articles arising from Asian tropical countries, 22% of all tropical countries. India ranked 8th in the world ranking of articles per country (Table 1.2) and Brazil 11th. All other countries are not tropical, with the P.R. China and the USA leading the board. Of course, it must be emphasized that the WoS only includes papers written in other languages if an abstract in English is provided in them. It, therefore, excludes the vast body of published works in other languages.

There are many possible explanations why these differences in rates of publications on constructed/treatment wetlands between the tropical and non-tropical/temperate countries exist:

Table 1.2 Top 20 ranking of countries based on the number of articles and reviews per country published in English according to a search of the Web of Science (WoS) on 29 December 2021

Rank number	Countries/regions	Record count
1	P.R. China	2711
2	USA	2207
3	Germany	496
4	Canada	490
5	Australia	486
6	Spain	480
7	England	466
8	India	443
9	Italy	377
10	France	355
11	Brazil	293
12	Poland	253
13	Denmark	249
14	Netherlands	205
15	Sweden	204
16	Czech Republic	196
17	South Korea	187
18	Ireland	184
19	Portugal	172
20	Greece	165

The search was for all fields containing the term “constructed wetland*” or “treatment wetland*,” over the entire period covered by the WoS, 1800–present

- Countries in temperate zones tend to be richer and politically more stable than tropical countries. Economic stability and equity favor successful measures to address environmental issues (Berthe and Elie 2015; Islam 2015).
- Countries that have clear laws or regulations to protect wetlands also lead in scientific research and output. As mentioned previously, the Clean Water Act of 1972 led to a surge in research and publications about wetlands in the USA. China established the ambitious China National Wetland Conservation Action Plan in 2000, which aimed to establish 713 wetland reserves—with more than 90% of natural wetlands effectively protected by 2030 (State Forestry Administration 2006; Wang et al. 2012). China has just (December 2021, Lan 2021) adopted a wetland protection act, to be enforced from June 2022, which will further the need for scientific research. China surpassed the USA, by far, in terms of publications about wetlands. The search of the WoS mentioned above not only returned 2711 records from China, while 2207 articles were found to have arisen from the USA, but almost all publications from China were from the year 2000 or later, while publications from the USA started appearing two decades before that.
- The regulations pertaining to wetlands in countries like the USA and China not only have stimulated the availability of funding and research activities but have also led to better education about wetlands. This is not only directly, through formal education, but also indirectly, because the regulations and laws mean that the media pay more attention to the subject. In addition, countries like China have developed so-called wetland parks that provide means for public education through visitor centers and nature trails. All this has helped raise awareness of issues relating to water and wetlands.
- The search in Table 1.1 was biased towards articles written in English, while much research has also been published in other languages, for example, Spanish, French, Portuguese (Latin America and Africa), and Chinese (Asia).

Wetlands for the purpose of phytoremediation are being used in the tropics, but the bulk of the literature, by far, is from temperate climates. Also, most regions in the tropics suffer from a lack of funding for environmental improvements. Because of the relatively low construction and maintenance costs, sustainable approaches to addressing environmental problems, such as wetlands for phytoremediation, are attracting increasing attention, and the potential for widespread applications in the tropics is enormous (Kivaisi 2001; Zhang et al. 2015). Water resources in the tropics are under huge anthropogenic pressure, not just threatening the supply of useable water, but also in terms of habitat for organisms. In many regions of the world, wetlands are the last remaining strongholds of biodiversity. Increasing the use of constructed wetlands for phytoremediation in the tropics therefore not only serves as a sustainable alternative for technological solutions for improvement of water quality but also compensates for habitat losses. This is an added benefit to ecological services, which is now weighed favorably in cost/benefit analyses (Masi et al. (2018)).

1.6 Conclusions

The use of constructed wetlands for the purpose of phytoremediation in the tropics has great potential as a sustainable nature-based solution, but, going by the existing literature, that potential is not commonly utilized. The tropics are ideal for the creation of wetlands because the general lack of freezing temperatures means that efficacy is less variable compared to wetlands in temperate regions. In addition to their function of remediating pollution, wetlands provide a host of other ecosystem services, including habitat for organisms. In this epoch, the Anthropocene, constructed wetlands will mitigate some of the loss of wetlands over the past centuries (Otte et al. 2021).

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Chapter 2
Phytoremediation Using Tropical Wetlands: Are Temperate Treatment Wetlands Sound Models?

Daniel Campbell

Abstract The use of wetlands to phytoremediate polluted waters is a practice that developed in temperate regions. This chapter explores the opportunities and pitfalls that may exist when transferring these approaches to the tropics. In temperate regions, phytoremediation initially focused on natural wetlands, but constructed wetlands are now used because of regulatory pressures to conserve natural wetlands and the ease of constructing treatment wetlands. In tropical regions, lax regulations still favor using both natural and constructed wetlands for phytoremediation, but caution is needed to avoid impacts on wetland goods, services, and dependent wildlife. The tropics have stable annual temperatures, unlike temperate regions, so phytoremediation processes will not be influenced by temperature shifts. However, the tropics exhibit strong seasonality in precipitation and wetland hydroperiods, unlike temperate treatment wetlands, therefore, emulating natural hydroperiods and using local vegetation types is appropriate. Greater diversity of plant functional types should be considered in tropical treatment wetlands, including woody plants, which are prevalent where seasonal flooding occurs. Given the richness of tropical wetland plants, local plant species must be favored in tropical treatment wetlands. Once these socioeconomic and biophysical differences are recognized, phytoremediation approaches from temperate treatment wetlands can be adapted to tropical regions.

Keywords Constructed wetland · Assimilation wetland · Wetland hydroperiod · Wetland vegetation · Pollution · Environmental management

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2.1 Introduction

Humans have discharged their wastes into rivers and deltas for millennia and have, perhaps inadvertently or perhaps through tradition, used wetlands to purify the water. Even until the early 1980s, over 300 natural wetlands in the United States were receiving wastewater discharges (Kadlec and Knight 1996). But the specific design of wetlands for the phytoremediation of polluted water is recent. Seidel (1966) published her foundational papers on the use of bulrushes to treat waters in Germany only in the 1960s. The use of wetlands to treat polluted waters has advanced tremendously over the past half-century.

We now use wetlands to phytoremediate non-point source pollutants (Braskerud 2002) and diverse point-source effluents from municipal sewage plants (Kadlec and Wallace 2009), agriculture (Rozema et al. 2016; Wang et al. 2018), stormwater (Malaviya and Singh 2012), mining (Mayes et al. 2009; Opitz et al. 2021), and other industries (Vymazal 2014; Jain et al. 2020). Several broad treatment wetland designs are now in operation, including free-water surface wetlands, horizontal subsurface flow wetlands, vertical flow wetlands, floating wetlands, and hybrid systems (Kadlec and Wallace 2009; Vymazal 2011a; Fonder and Headley 2013; Colares et al. 2020). Treatment wetlands can now achieve high removal efficiencies for BOD and diverse organic compounds, suspended solids, nutrients, metals and metalloids, and even pharmaceuticals (Kadlec and Wallace 2009; Garcia et al. 2010; Marchand et al. 2010; Li et al. 2014; Vymazal 2014; Jain et al. 2020). Using wetlands to phytoremediate polluted waters is now as much within the domain of engineers as it is within the domains of environmental scientists and wetland ecologists.

However, with notable exceptions (see Zhang et al. 2015), the use of treatment wetlands to phytoremediate polluted waters has developed and been applied in industrialized countries within temperate regions. It is tempting to directly transfer this large body of literature and detailed guidelines from temperate regions (e.g., Kadlec and Wallace 2009) to tropical regions. But we must recognize that large differences exist between these regions, not only in climate, ecology, and biogeography but also in culture, economics, and regulatory environments. This chapter reflects on some major differences that would be relevant to this transfer. Given the global diversity of wetlands and their human contexts, this big-picture exercise is fraught with the dangers of oversimplification. But the difficult question remains: What pitfalls, if any, impede or bias our transfer of wetland phytoremediation approaches from temperate to tropical regions?

2.2 Natural Versus Constructed Treatment Wetlands

In temperate regions, many natural wetlands were used to filter out pollutants until the turn of the century if natural wetlands were conveniently located nearby (Kadlec and Knight 1996). A few examples illustrate the range of natural wetlands used to treat polluted waters in North America: freshwater marsh (Mudroch and Capobianco

1979), salt marsh (USEPA 1984), cypress forest domes (Ewel and Odum 1978), forested wetlands (Day et al. 2004), and northern peatlands (Kadlec and Bevis 2009). All these natural wetlands received secondarily treated municipal wastewater. They are sometimes called assimilation wetlands (Sloey et al. 2021). Forested assimilation wetlands in the southern United States continue to remove nutrients even after a half-century of operation (Day et al. 2004), although data gaps remain regarding long-term impacts and performance (Sloey et al. 2021).

Natural wetlands in temperate regions are now rarely considered for the phytoremediation of polluted waters. In fact, the second edition of Treatment Wetlands (Kadlec and Wallace 2009) only covers constructed treatment wetlands. The reasons behind this shift reflect an evolution in the conservation of wetlands and treatment wetland regulations and parallel evolution in wetland treatment designs. It is useful to review why this occurred in temperate regions in order to apply the best practices to tropical regions.

We now realize that a large fraction of the wetlands in temperate regions has been converted or lost (Davidson 2014). Furthermore, we now realize that wetlands play a disproportionate role in the broader ecosystem. They provide key ecosystem goods for humans, including food, freshwater, fibers, fuels, and medicines, and they also provide key ecosystem services, which benefit humans, such as water storage and flood control, water purification, storm protection, carbon capture, along with providing recreational, aesthetic, and even spiritual opportunities (MEA 2005). But wetlands are not only instruments in the broader landscape; they also offer habitat to diverse wetland-dependent flora and fauna (e.g., Godfrey and Wooten 1979–1981; Minello et al. 2003; Holopainen et al. 2015). Although some may only provide modest utilitarian value to human society, they all have intrinsic value. For instance, the endangered whooping crane (Grus americana) and Mississippi gopher toad (Lithobates sevosus) are wetland-dependent North American species for which wetlands have been conserved (Niering 1988). As wetlands have been increasingly used to phytoremediate polluted waters, it has become clear that these treatment wetlands also attract wildlife. If the polluted waters have elevated levels of toxic compounds such as heavy metals or pesticides, the wetlands can act as ecological traps, deleteriously affecting wildlife (Sievers et al. 2018).

The extensive loss of wetlands, their roles within the broader ecosystem, their habitat provision, and the potential for deleterious effects of some pollutants on wetland-dependent wildlife have spurred extensive conservation efforts as well as legislation and increasingly strict regulation for natural wetland protection in temperate countries (e.g., Votteler and Muir 1996; Ramsar Convention on Wetlands 2018). There has also been increasing regulation on wastewater treatment and on water quality of receiving waters (WWAP 2017). Together, these trends have led to restricted permitting and strict oversight where natural wetlands are used for phytoremediation.

In parallel with this increased conservation and legislation of temperate wetlands, there have been tremendous advances in understanding how to construct wetlands (e.g., Biebighauser 2015) and specifically how to design wetlands to phytoremediate polluted wastewaters (Kadlec and Wallace 2009; Vymazal 2011a).

Wetlands can now be constructed at relatively low cost to specific engineered designs to maximize phytoremediation of specific pollutants. The construction of treatment wetlands also increases wetland area in the landscape, which mitigates wetland loss. Together, the pressure to conserve wetlands and the relative ease of constructing treatment wetlands have both favored the construction of wetlands instead of using natural wetlands to phytoremediate polluted waters.

Wetland conservation, policy, and legislation remain in their infancy in many tropical countries (e.g., Junk et al. 2014; Gardner 2018), in stark contrast to many temperate countries. The use of natural versus constructed wetlands does not yet have any strict separation in policy or practice. For instance, both natural and planted mangrove wetlands are used to phytoremediate waters from shrimp farming in Asia (Ahmed et al. 2018). However, the same arguments for wetland conservation are as valid in tropical regions as they are in temperate regions. Again, wetlands provide key ecosystem goods and services to humans (MEA 2005; Ricaurte et al. 2014), and again, wetland loss is high in the tropics (Davidson 2014). Moreover, rates of human population growth are highest in the tropics (Kummu and Varis 2011), so human pressure and dependency on natural wetlands will continue to grow. Tropical wetlands also provide habitat to an even broader suite of wetland-dependent species (Junk et al. 2006) because the tropics have much higher species richness, species density, and restricted-range endemic species than temperate regions (Kreft and Jetz 2007; Jenkins et al. 2013). And once again, the quality of their habitat could be impacted if wetlands are used to phytoremediate certain classes of pollutants.

Wetlands are being constructed in tropical regions for the phytoremediation of polluted waters (Zhang et al. 2015). Constructing treatment wetlands in the tropics remains the most sustainable approach. However, the nascent conservation concerns and the lax regulatory systems in most tropical countries suggest that using natural wetlands to phytoremediate polluted waters will remain an important water quality management tool, especially where wetlands are prevalent in the landscape. Certainly, the use of natural assimilation wetlands to remove excess nutrients (e.g., Day et al. 2019; Sloey et al. 2021) is more appropriate than using natural wetlands to phytoremediate more complexly polluted waters.

The main lesson to be taken from temperate regions is that using natural wetlands for phytoremediation may alter ecosystem goods and services or wildlife habitat availability and quality in those wetlands. As such, care should be taken to review potential impacts on wetland goods and services and on habitat availability and quality for wetland-dependent wildlife before using natural tropical wetlands for phytoremediation.

2.3 Temperature and Its Seasonality

Consider the obvious differences in climate of a few representative temperate and tropical regions where wetlands exist (Fig. 2.1). Because of more direct solar radiation, tropical regions have higher mean annual temperatures than temperate regions,

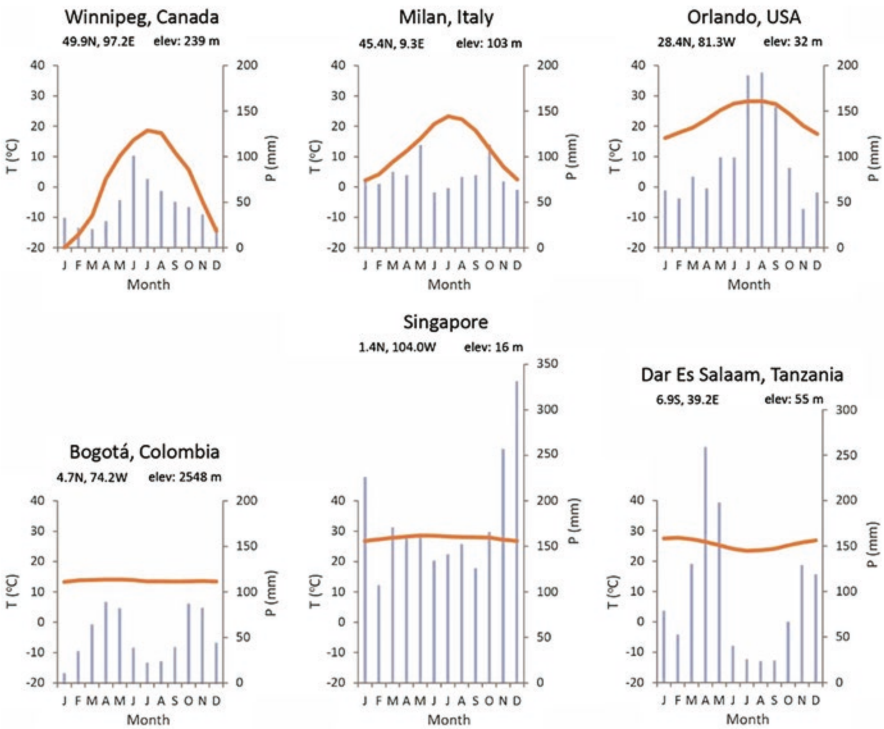

Fig. 2.1 Climate diagrams for selected cities in temperate regions (top) and tropical regions (bottom) showing monthly mean temperature (orange line) and mean precipitation (blue bars). Data: https://climatecharts.net/

except at elevated altitudes (i.e., Bogotá, Colombia). However, the most striking difference is the seasonal fluctuation in temperature that occurs in temperate regions. Seasonal fluctuations are more obvious at higher temperate latitudes, where freezing conditions occur during winter (i.e., Winnipeg, Canada), but seasonal fluctuations also occur under warm, lower temperate latitudes (i.e., Orlando, Florida, USA). Such seasonal fluctuations in temperature are hardly or not found in tropical regions.

The temperature of water in wetlands generally follows air temperatures (Kadlec and Reddy 2001; Kadlec 2009). Both the mean annual temperature and seasonal fluctuations in temperature have profound influences on wetland processes, including seasonal freeze-thaw dynamics at higher latitudes, seasonal cycles of evapotranspiration, the seasonal extent of oxygen dissolution into surface waters, seasonal sorption, redox, and other chemical reactions, seasonal cycles of plant growth and uptake, as well as seasonal cycles of microbially-mediated transformations such as decomposition, BOD removal, ammonification, nitrification, and denitrification (Kadlec and Reddy 2001; Kadlec and Wallace 2009). One consequence is that the rates of pollutant removal by treatment wetlands in temperate climates also fluctuate with the seasons and are generally reduced during winter (Kadlec and Reddy 2001; Kadlec and Wallace 2009). Designers and managers of treatment wetlands in temperate regions have devised a suite of strategies to overcome these performance challenges under seasonally variable temperatures and especially under winter operating conditions, such as over-sizing of treatment wetlands, reducing loading rates, favoring subsurface flow wetland systems, encouraging plant thatch as insulation, or the aeration of treatment inflows (Mæhlum and Stälnacke 1999; Werker et al. 2002; Ouellet-Plamondon et al. 2006; Wang et al. 2017). But even under warm temperate climates, seasonality in temperature can influence wetland phytoremediation processes. For instance, a seasonal temperature shift from 26 to 14 °C, which is typical of a warm temperate climate, had profound effects on microbial communities in epiphytic biofilms and reduced their abilities to reduce total nitrogen, ammonium, total phosphorus, and COD (Mu et al. 2021).

Tropical wetlands do not experience these seasonal fluctuations in temperature and, as such, will not suffer temperature-induced performance shifts if they are used to phytoremediate polluted waters. Tropical wetlands could even be viewed as providing simpler baseline conditions under which to evaluate wetland phytoremediation. It is quite conceivable that tropical wetlands used for phytoremediation of polluted waters may yield important lessons to apply back to temperate treatment wetlands. For instance, the performance of different treatment wetland designs or the removal of specific pollutants may be more easily evaluated under the longer-term stable temperature regions found in the tropics. Constructed wetland designs could be evaluated at different tropical elevations, allowing to test wetland phytoremediation under different mean annual temperatures but without any seasonal fluctuations in temperature. Other research questions could be addressed on the removal limits of different pollutants, the effects of different vegetation types on wetland treatment, and the long-term evolution of treatment wetlands. Researchers in tropical treatment wetlands should consider this larger role that their studies could play.

2.4 Hydroperiod and Soil Saturation

Hydroperiod refers to the duration of flooding and soil saturation and its flip-side, exposure, and consequent aeration (Rasmussen et al. 2018). Hydroperiod is a critical component of wetlands, determining not only soil aeration and saturation but also redox conditions, microbial transformations, vegetation, and other biotas. In temperate regions, natural wetlands have a broad range and patterns of hydroperiods (Mitsch and Gosselink 2015). In contrast, wetlands constructed for the phytoremediation of polluted waters generally do not. Free water surface treatment wetlands and horizontal subsurface flow wetlands are designed to generally maintain constantly saturated soil conditions, despite seasonal fluctuations in temperature, precipitation, evapotranspiration, and snowmelt (Kadlec and Wallace 2009; Fonder and Headley 2013). Vertical flow wetlands are the only class of constructed wetlands designed for intermittent soil saturation, which is a function of the dosing rates (Kadlec and Wallace 2009; Fonder and Headley 2013).

In contrast, precipitation follows strong seasonality in many tropical regions, with a strong rainy season followed by a drier season (Figs. 2.1 and 2.2). Consider first the wetlands of the arid Lake Chad region, which are among the world's largest wetlands (Keddy et al. 2009). Water levels vary between years, but annual fluctuations are relatively low as a result of the vast flat floodplain, despite strong seasonality in precipitation, with an amplitude of only 1 m between high and low water levels (Olivry et al. 1996). But many tropical wetlands have wide fluctuations in water levels and are periodically exposed. Contrast the Lake Chad example with the Amazon Basin, another of the world's largest wetlands, and an example station at Leticia, Colombia (Fig. 2.2). The mean annual amplitude of water levels in the Amazon River is over 10 m between maximum and minimum river stages, which inundates extensive floodplains for several months at a time, followed by exposed conditions for several months. These seasonal pulses of inundation/exposure, and consequently saturated/aerated soils, are common in floodplain wetlands in the Amazon basin and profoundly affect biogeochemical conditions, vegetation, fauna, and food webs (Parolin 2009; Junk et al. 2011). Other tropical regions have less dramatic fluctuations in inundation patterns (e.g., Congo River basin; Lee et al. 2011), but they nevertheless have marked hydroperiod cycles.

Consideration should be given to working with hydroperiod cycles that are more typical in the tropics, instead of trying to impose stable water levels found in temperate region treatment wetlands, unless they are locally prevalent. Constantly flooded and saturated soil conditions are not the norm in most tropical regions. In regions such as the Amazon basin, managers will have no choice but to adapt to the dramatic seasonal pulses of water when using wetlands to phytoremediate polluted waters.

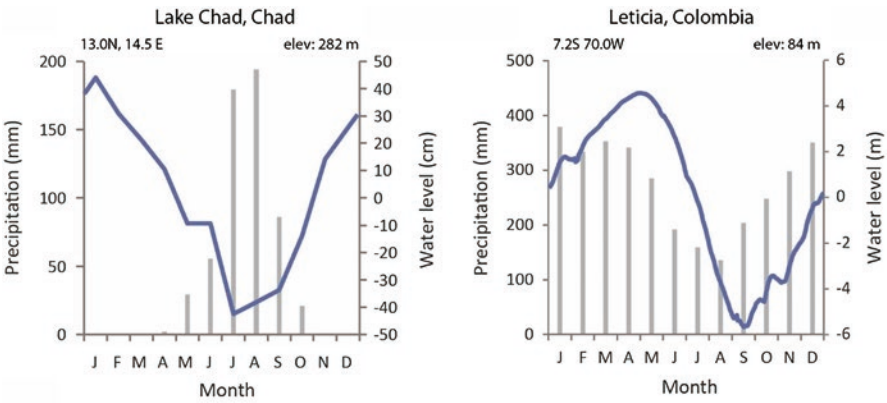

Fig. 2.2 Mean monthly precipitation and mean daily water levels at Lake Chad (right) and on the Amazon River at Leticia, Colombia (left). Water levels are shown relative to the mean annual water level. Data from Olivry et al. (1996), https://climatecharts.net/ and http://www.ore-hybam.org/index.php/eng

2.5 Low Diversity Marsh Vegetation?

Diverse wetland plant communities are desirable for phytoremediation (Coleman et al. 2001; Kadlec and Wallace 2009; Zhang et al. 2010), yet wetlands constructed for phytoremediation in temperate regions are rarely diverse; only a narrow selection of plant functional types and plant species dominate (Kadlec and Wallace 2009; Vymazal 2011b; Table 2.1). Free water surface treatment wetlands predominantly use emergent macrophytes, along with floating-leaved macrophytes, free-floating macrophytes, and submersed macrophytes, while horizontal subsurface flow and vertical flow treatment wetlands almost exclusively use emergent macrophytes (Kadlec and Wallace 2009; Vymazal 2011b; Fonder and Headley 2013). The emergent macrophyte species are often tall species, typical of productive marsh (referred to as herbaceous swamp in Europe), and include cattail (*Typha* spp.), bulrush (*Scirpus* spp. *sensu lat*o), rushes (*Juncus* spp.), reed canary grass (*Phalaris arundinacea*), and common reed (Phragmites australis; Kadlec and Wallace 2009). Often, they exist almost as monocultures. In the tropics, these same genera of emergent macrophytes are targeted in wetlands constructed for phytoremediation (Zhang et al. 2015), along with others such as papyrus (*Cyperus papyrus*); free-floating plants such as water hyacinth (*Eichhornia crassipes*) are also used (Rezania et al. 2015).

Some of the common plant species used in temperate treatment wetlands can also become invasive in natural wetlands, displacing native vegetation. Examples include the emergents *Typha* x glauca, *Typha angustifoli*a, *Phragmites australis,*

Table 2.1 Dominant species found in constructed treatment wetlands in the United States (Kadlec and Wallace 2009), organized by plant functional type

Scientific name	Plant functional type
Typha spp.	Tall emergent
Phragmites australis	Tall emergent
Phalaris arundinacea	Tall emergent
Scirpus spp. *sensu lato*	Tall to short emergent
Juncus spp.	Tall to short emergent
Pontederia cordata	Short emergent
Bacopa caroliniana	Short emergent
Sagittaria spp.	Short emergent or Floating-leaved
Hydrocotyle spp.	Floating-leaved
Nuphar luteum	Floating-leaved
Lemna spp.	Free-floating
Limnobium spongea	Free-floating
Eichhornia crassipes	Free-floating
Ceratophyllum demersum	Submersed
Potamogeton spp.	Submersed
Elodea spp.	Submersed

and *Phalaris arundinacea* in North America (Zedler and Kercher 2004; Rodriguez and Brisson 2015; Bansal et al. 2019). A great deal of effort is sometimes expended to avoid or replace invasive species in treatment wetlands (Kadlec and Wallace 2009).

These points beg a few questions. Must the vegetation of tropical treatment wetlands be dominated by marsh vegetation, with low diversity emergent macrophytes and perhaps floating-leaved, free-floating, and submersed macrophytes, as they are in temperate treatment wetlands? Are there also dangers of spreading invasive species? Instead, should managers not target regionally representative vegetation, wetland plant functional types, and species?

Tropical wetlands have diverse herbaceous, shrub-dominated, or forested wetland plant communities (Giesen 2018). Although extensive marshes occur, as they do in the Lake Chad region (Olivry et al. 1996), diverse forested wetlands predominate in many tropical regions, such as large river floodplains, peatlands, and coastal mangroves (Bwangoy et al. 2010; Melack and Hess 2010; Giri et al. 2011; Ribeiro et al. 2021). For instance, over a thousand species of trees occur in Amazonian wetlands (Parolin 2009). Surely this vegetation diversity, including woody species, could be exploited in wetlands used to phytoremediate polluted waters in tropical regions. There is certainly no reason to introduce non-native wetland species to phytoremediate polluted waters in tropical regions.

It is useful to examine more closely why low-diversity marsh vegetation prevails in temperate treatment wetlands. First, emergent macrophytes, along with floating-leaved, free-floating, and submersed macrophytes, tolerate the permanently saturated, anaerobic conditions that are often prescribed in temperate treatment wetlands (Kadlec and Wallace 2009). Emergent and floating-leaved macrophytes have air channels in their stems that oxygenate their roots, allowing them to grow in permanently anaerobic soils (Nakamura and Noguchi 2020). Woody plants generally lack adaptations to permanently flooded conditions, so they only occur in wetlands that have periodically exposed and aerated soils (Kozlowski 1984). Only a few temperate woody plants can aerate their roots when flooded, such as bald cypress (Taxodium distichum; Martin and Francke 2015) and some willows (*Salix* spp.; Randerson et al. 2011), allowing them to survive over the long term under flooded conditions.

Second, temperate treatment wetlands are now mostly constructed and hence successionally young. It is relatively easy to establish a cover of emergent, floating-leaved, free-floating, or submerged macrophytes in newly constructed wetlands, either by seed or by plantings, given suitable moisture conditions (Mitsch et al. 2012; Biebighauser 2015). Even if no plants are introduced, plants will colonize newly constructed wetlands if hydrological conditions are suitable, especially if parent populations are found nearby (Mitsch et al. 2012). Sometimes, diverse species are planted but fail to establish because of unsuitable hydrology or excess pollutants, leaving low-diversity emergent vegetation to dominate (Kadlec and Wallace 2009). Often, only a few emergent species are selected for planting or seeding (Calheiros et al. 2009; Vymazal 2011b), perhaps because managers are familiar with their performance for phytoremediation.

Third, monocultures of tall emergent macrophytes occur because of competitive hierarchies among wetland plants. Treatment wetlands generally receive elevated

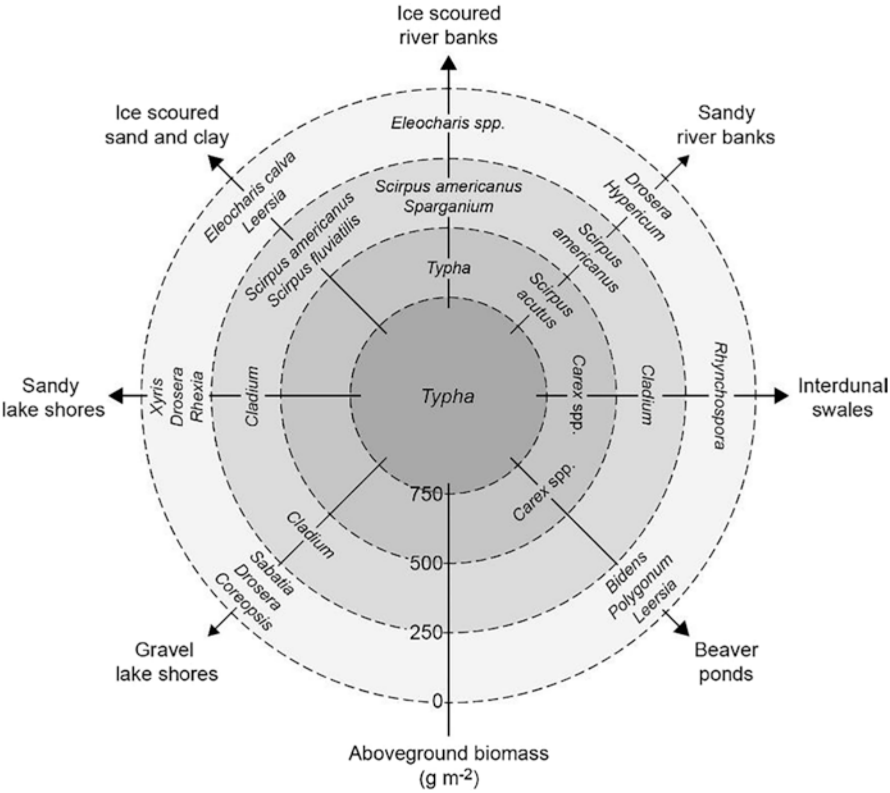

Fig. 2.3 Competitive hierarchy which occurs in temperate herbaceous wetlands between diverse low-productivity vegetation and low-diversity high-productivity vegetation. (Modified from Moore et al. 1989; used with permission). Nutrient additions push diverse temperate wetland vegetation toward the center of this diagram

nutrient inputs (Kadlec and Wallace 2009). With sufficient nutrients, a few tall emergent macrophytes become most productive, then outcompete shorter-stature species for light until they dominate (Fig. 2.3; Moore et al. 1989). For instance, the vegetation in a natural wetland used for phytoremediation in the north-central USA shifted from a nutrient-poor peatland dominated by bryophytes and short ericaceous shrubs to 2 m tall Typha marsh vegetation within a decade of receiving secondarily treated municipal wastewaters. This was likely a result of competitive dominance, although a shift toward more saturated soil conditions may also have played a role (Kadlec and Bevis 2009). Similar shifts to herbaceous monocultures have been observed among wetland plants in a tropical region receiving wastewater discharges (Kent et al. 2000).

Woody vegetation has not been a focus of treatment wetlands in temperate regions, although Kadlec and Wallace (2009) consider this as an oversight. Some examples do exist. Natural forested wetlands in the southern United States have been successfully used to polish secondarily treated municipal wastewaters; understorey and tree species showed marked increases in productivity (Day et al. 2019). In a constructed horizontal subsurface flow wetland in Australia, young Melaleuca trees had high rates of removal of suspended solids, turbidity, BOD, total nitrogen, and ammonia from primary municipal effluent sewage (Bolton and Greenway 1999). Shrub thickets of willow (*Salix* spp.) are also now included in temperate-constructed treatment wetlands (Lachapelle-T et al. 2019).

The use of herbaceous wetlands in the tropics will likely continue to play an important role in the phytoremediation of polluted waters (e.g., Zhang et al. 2015). However, given the seasonal fluctuations of precipitation and soil saturation in many tropical wetlands and the consequent prevalence of woody plants, and given their diversity within and across tropical ecoregions, it would be prudent to consider shrub-dominated or forested wetland systems for the phytoremediation of polluted waters in the tropics. Some wetland woody plants, such as mangroves, are traditionally incorporated into shrimp farm effluent treatment systems in the tropics (Ahmed et al. 2018). Other tropical woody plants may be candidates for wetland treatment systems, especially if periodic exposure can be built into their flooding regimes. Consideration could be given to intermittent hydroperiods, perhaps using alternate basins or treatment systems, to foster the growth of woody species.

Given the high diversity of plants in tropical regions, practitioners should be less fixated on a small suite of species in wetlands for the phytoremediation of polluted waters and should embrace the local native diversity of wetland vegetation, including woody species. Emphasis should be placed on selecting suitable functional types for phytoremediation, instead of focusing on specific species. Using local species will also limit the dangers of spreading potentially invasive non-native species into tropical wetlands. Perhaps local wetland species in decline or at risk could even be incorporated into treatment wetland designs.

2.6 Conclusions

Water quality challenges exist in many tropical regions (WHO 2014; WWAP 2017), and the use of wetlands to phytoremediate polluted waters in these regions shows promise. Treatment wetland systems require low maintenance and are cost-efficient as compared to conventional treatment options. Many of the same approaches and designs of wetland treatment systems from temperate regions will be applicable to tropical regions, but not all will be. Broad differences exist in history, culture, and regulations, but also in climate, water fluctuations, and vegetation (Table 2.2), which together suggest that although lessons can be borrowed from temperate regions, distinct approaches should be developed in tropical regions for the phytoremediation of polluted waters.

Constructing treatment wetlands adapted to tropical climates with locally representative water level regimes and local vegetation, including woody species, shows the potential to harness tropical wetlands for the phytoremediation of polluted

Table 2.2 Summary of main differences between temperate and tropical regions relative to the use of wetlands for the phytoremediation of polluted waters

Attribute	Temperate regions	Tropical regions
Seasonal temperature fluctuations	High to moderate	Low to none
Seasonal water level fluctuations	Variable, often low to moderate	Variable, often high
Biodiversity	Low to moderate	High
Natural wetland forms	Marsh, peatlands, forested wetlands	Forested wetlands, marsh, some peatlands
Wetland loss	High	Moderate, increasing
Environmental regulation and enforcement	Strict	Generally lax
Research and development on treatment wetlands	High	Low
Phytoremediation using natural wetlands	Legacy approach; assimilation wetlands	Unclear
Phytoremediation using constructed wetlands	Many examples, dominant approach	Few examples
Predominant treatment wetland vegetation	Emergent macrophytes	Emergent macrophytes or floating plants; potentially woody plants

waters. Where natural tropical wetlands are used for phytoremediation, care will be needed to verify the effects of polluted effluents on wetland goods and services and on wetland-dependent species and the quality of their habitat.

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Chapter 3
Tropical and Subtropical Wetland Plant Species Used for Phytoremediation in Treatment Wetlands

Hernán Ricardo Hadad and María Alejandra Maine

Abstract Treatment wetlands (TWs) provide a highly applicable nature-based solution for water quality problems across a range of scenarios. The high diversity of aquatic plants in tropical and subtropical regions of the world provides great potential for uses in TWs. However, the abundant literature about TWs is scarce regarding aspects such as the plant role, ecology, bioaccumulation efficiency, and the search for native alternative species. Additional studies are needed with a focus on the role of plants, particularly those that have not been studied in detail, their responses to different pollutants, their tolerance to high levels of contaminants, and their ability to accumulate pollutants in tissues. The aims of this chapter are to characterize the plant species used in TWs from tropical and subtropical regions, to describe the role of plants and their ecological dynamics in TWs, and to assess some alternative species that potentially can be used in TWs.

Keywords Macrophytes · Wetland systems · Pollution · Nature-based solutions

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3.1 Introduction

The high diversity of aquatic and wetland plants in tropical and subtropical regions of the world provides great potential for uses in phytoremediation techniques, ecotoxicological studies, and biomonitoring programs. Phytoremediation refers to pollutant removal using terrestrial plants to remediate soils or aquatic plants to remediate wastewaters. As a phytoremediation technique, treatment wetlands (TWs) provide a highly applicable nature-based solution for water quality problems across a range of scenarios. In these systems, plants are the main component, and natural processes are optimized to improve water quality.

Due to the fact that aquatic and wetland plant species are diverse and abundant in tropical and subtropical regions, and the environmental conditions are highly favorable for their growth, TWs are a very suitable technology (Rodriguez-Dominguez et al. 2020; Wijekoon et al. 2022). However, the abundant literature about TWs is scarce regarding studies about the plant role, ecology, toxicological responses, bioaccumulation efficiency, and the search for native alternative species. Therefore, we think that additional studies focused on TW plants are needed.

The aims of this chapter are to characterize the plant species used in TWs from tropical and subtropical regions, describe the role of plants and their ecological dynamics in TWs, and to assess some alternative species that potentially can be used in TWs. In this chapter, we describe these issues and present literature on the subject.

3.2 Plant Species Used in TWs from Tropical and Subtropical Regions

In natural wetlands in tropical and subtropical regions, abundant and varied vegetation develop. In their natural habitat, plants occur in shallow lakes, swamps, streams, etc. (Fig. 3.1).

A few species of wetland and aquatic plants have reached wide distributions in these regions using a combination of natural dispersal mechanisms, usually involving zoochory or hydrochory. Others have had their world range increased by human intervention, whether deliberate or inadvertently. In addition, strategies such as asexual reproduction often allow them to colonize and reproduce quickly and successfully in new regions. Due to their dominant nature and their ecological plasticity, some macrophytes become weeds (Hofstra et al. 2020).

Macrophyte life forms are usually classified by where the majority of their photosynthetic tissue occurs in relation to the water surface and whether or not they are rooted in the sediment or attached to other substrates: for example, to rock surfaces in the case of the Podostemaceae (“riverweeds”) which occur in fast-flowing tropical rivers. The categories commonly recognized are emergent, free-floating, submerged, and rooted with floating leaves. Figure 3.2 shows examples of plant species

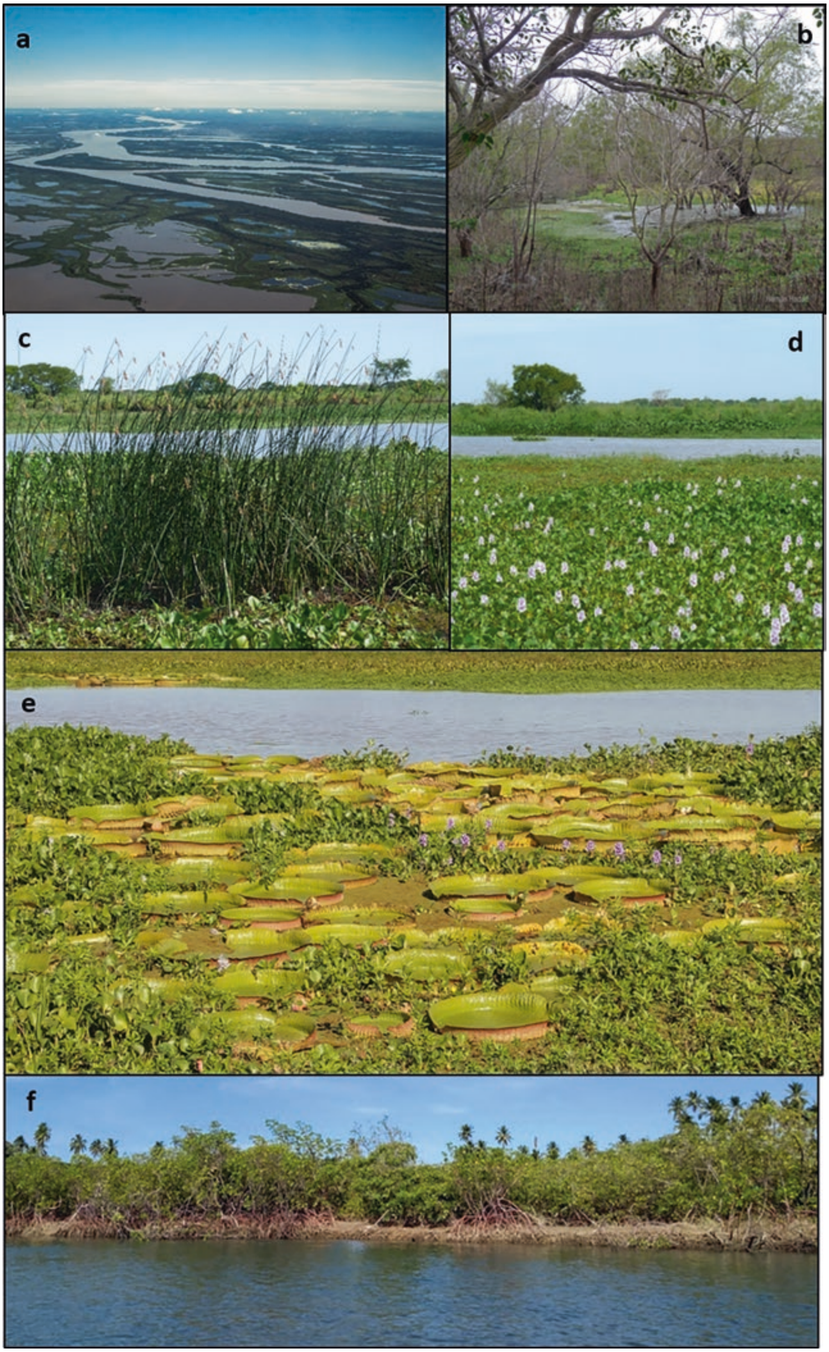

Fig. 3.1 Examples of natural wetlands. (**a**) Middle Paraná River floodplain, Argentina (aerial view, streams, temporary lakes, marshes). (**b**) Swamp with woody and herbaceous species. (**c**) Marsh dominated by *Scirpus californicus*. (**d**) Marsh dominated by *Eichhornia crassipes*. (**e**) Marsh dominated by *Victoria cruziana*. (**f**) Mangrove swamp (Photos **a–e**: H.R. Hadad, **f**: E. Nocetti)

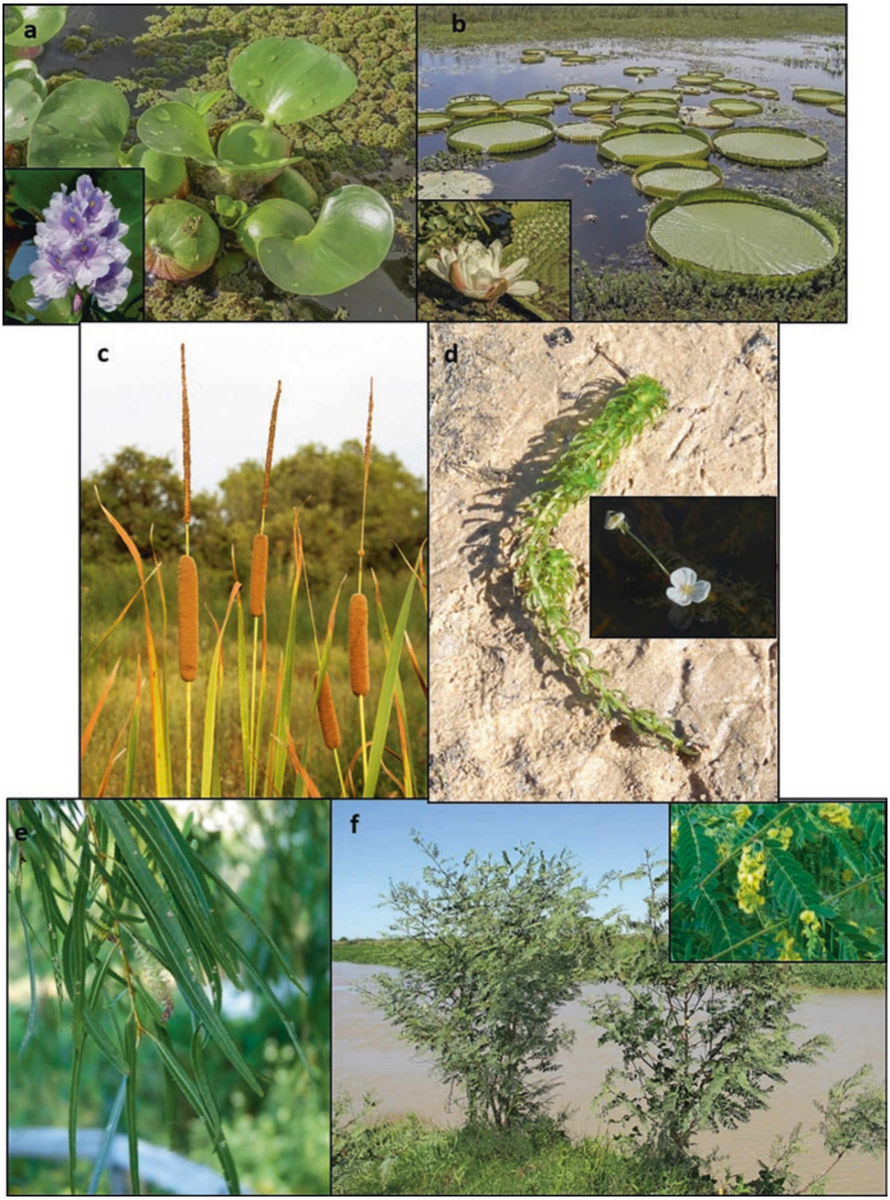

Fig. 3.2 Life forms of aquatic plants. (**a**) Free-floating, *Eichhornia crassipes.* (**b**) Floating leaves, *Victoria cruziana.* (**c**) Emergent, *Typha domingensis.* (**d**) Submerged, *Egeria najas*. (**d**) Wetland tree, *Salix humboldtiana.* Wetland shrub*, Sesbania virgata* (Photos by H.R. Hadad)

belonging to each life form. Emergent species are rooted plants that develop on the edges of water bodies and are capable of tolerating unflooded conditions (Campbell et al. 2016), separating them from wetland species that are less tolerant of continuous flooding (Campbell and Keddy 2022). The floating leaf-rooted species develop in deeper areas than the emergent plants, they are rooted to the bottom and their leaves float on the water surface. Free-floating macrophytes inhabit any part of the water body, and in rivers, they can drift downstream in the current and sometimes are also blown upstream by wind action. Submerged macrophytes can show aerial parts on the water surface, such as flowers or portions of stems, but the largest proportion is found under the water surface and may or may not root to the bottom, depending on the species. Other type of plants frequently used in TWs are woody plants, such as willows (*Salix* spp.), *Melaleuca* spp., and mangroves.

The different life forms of aquatic plants are of great importance concerning their role in TWs. In a natural water body, a sequence of growth forms can be found along a depth gradient that extends from the shoreline to the deepest parts. In TWs, an attempt is made to represent this gradient by planting the different types of plants in the areas corresponding to their growth form. For example, at the edges of a free-water surface (FWS) wetland, it would be desirable to plant emergent species, while in the deeper central areas, free-floating species would develop. To achieve a greater dispersal of emergent or woody plants, a strategy to apply in FWS wetlands could be the addition of soil strips to decrease the depth of the water (Maine et al. 2009). On the other hand, in horizontal subsurface flow (HSSF) wetlands, emergent species are always used. They should be capable of growing in the substrate used, be it rock, light expanded clay aggregate (LECA), sand, etc. For example, the emergent species *Typha domingensis* and *Arundo donax* are used in different HSSF wetlands that were constructed in Argentina and Mexico (Fig. 3.3). In the case of TWs from Mexico, they commonly use *tezontle* as a substrate, which is a rock of volcanic origin that is abundant in the region. In TWs from Argentina, mainly river gravel is used.

The different plant species that inhabit a geographical region can be used in TWs since they have the advantage of being adapted to the climate and the prevailing water and soil conditions. The use of TWs in tropical and subtropical regions of the world is favored by the high plant availability due to the climate. Rodriguez-Dominguez et al. (2020) carried out an extensive and detailed review accounting for the TWs operating in Latin America and the Caribbean region. These authors reported on 520 systems in 20 countries with 114 different species, one of the most extensive reviews on plants used in TWs. One example from another tropical region is the work of Sharma et al. (2021), who studied a hybrid wetland system for the treatment of dairy farm wastewater in India. The whole system has an area of 40 m^2 and consists of a vertical flow wetland (VFW), an HSSF, and another VFW. In the VFWs, *Arundo donax* was planted, while in the HSSF, *Hibiscus esculentus* and *Solanum melongena* were used. This system reached removals of 84% for N, 86% for P, 92% for TSS, and 95% for BOD. The tissues of *A. donax* in the VFWs showed significantly higher nutrient concentrations in comparison with the control. Therefore, Sharma et al. (2021) concluded that the studied hybrid TW was efficient for the treatment of the wastewater from a dairy farm. Maine et al. (2019) evaluated the efficiency of hybrid systems for the treatment of wastewater from a fertilizer manufacturing plant with high ammonium concentration. The system was composed of FWS and HSSF wetlands and was planted with *T. domingensis* or *C. indica*.

Fig. 3.3 Emergent species used in TWs. (**a**) *Typha domingensis* in a constructed wetland for the treatment of effluent from a pet-care center depending on a pet-food factory in Argentina and (**b**) *Arundo donax* in a wetland constructed for the treatment of sewage in Mexico (Photos by H.R. Hadad)

In comparison with the unplanted systems, the TWs planted showed significantly higher pollutant removals and *T. domingensis* was the macrophyte with the highest growth, tolerating the wastewater conditions in both HSSFWs and FWSWs. Turcios et al. (2021) explained aspects related to the ecosystem services and sustainability of saline TWs. Hydro-halophyte herbaceous plants, such as *Schoenoplectus americanus*, *Salicornia fruticose*, and *Typha angustifolia*, show a great potential to be used in TWs in South America.

Many of the macrophyte species that are commonly used in TWs have a cosmopolitan distribution, e.g., *Typha* spp., *Phragmites australis, A. donax, Schoenoplectus californicus,* and *Phalaris arundinacea* (Kadlec and Wallace 2009). These species have been used for the treatment of household, sewage, and industrial effluents (Maine et al. 2007, 2013; Mbuligwe 2005; Nivala et al. 2013; Nocetti et al. 2020; Vymazal 2011; Vymazal and Kropfelová 2008). However, these cosmopolitan species are not always native species and they can be invasive (Bansal et al. 2019; Zedler and Kercher 2004; Rodriguez and Brisson 2015).

Typha spp. have proved to be among the most tolerant and productive macrophytes in TWs around the world (Calheiros et al. 2009; Carranza-Álvarez et al. 2008; Hadad et al. 2006, 2010, 2018, 2021a; Juwarker et al. 1995; Kadlec and Wallace 2009; Maddison et al. 2005; Maine et al. 2007, 2009, 2013; Manios et al. 2003; Rodriguez-Dominguez et al. 2020; Sesin et al. 2021; Vymazal 2013; Vymazal and Kröpfelová 2008). Due to their characteristics, *Typha* spp. form monospecific communities in the TWs where they are used. Hadad et al. (2006) evaluated the growth of macrophytes from an FWS wetland constructed at a pilot scale for the treatment of an effluent from a metallurgical industry. At the beginning of the study, 11 free-floating and emergent macrophytes commonly found in natural wetlands of the Middle Paraná River floodplain (Argentina) were planted. At the end of the study, only *T. domingensis* remained in the wetland, becoming the dominant macrophyte. Maine et al. (2022) assessed the efficiencies of *T. domingensis* and *C. indica* for Cr, Ni, and Zn removal from landfill leachate in VFWs at mesocosms-scale. The efficiencies of these systems were compared in two experiments which used two metal initial concentrations (0.2 and 1 mg L^{-1}). These authors concluded that metals were efficiently accumulated in the macrophyte tissues and the VFWs planted with *T. domingensis* showed higher metal removal in comparison with the systems planted with *C. indica* (Fig. 3.4).

Regarding woody species, one of the most used is the willow (*Salix* spp.). A TW based on willows was used to treat sewage from rural areas of Denmark (Gregersen and Brix 2001). The main attribute of this type of treatment is that the system does not have any liquid discharge due to evapotranspiration, and the nutrients are recycled through the biomass of the willows. The harvestable biomass is used as an energy source (Brix and Arias 2005; Kwasniewski et al. 2022). Vincent et al. (2014) studied a pilot-scale TW planted with *Salix miyabeana* to treat leachate from a deposit of wooden posts that were treated with preservatives. The willows were not significantly affected in their growth by the pollutants. Therefore, a TW based on

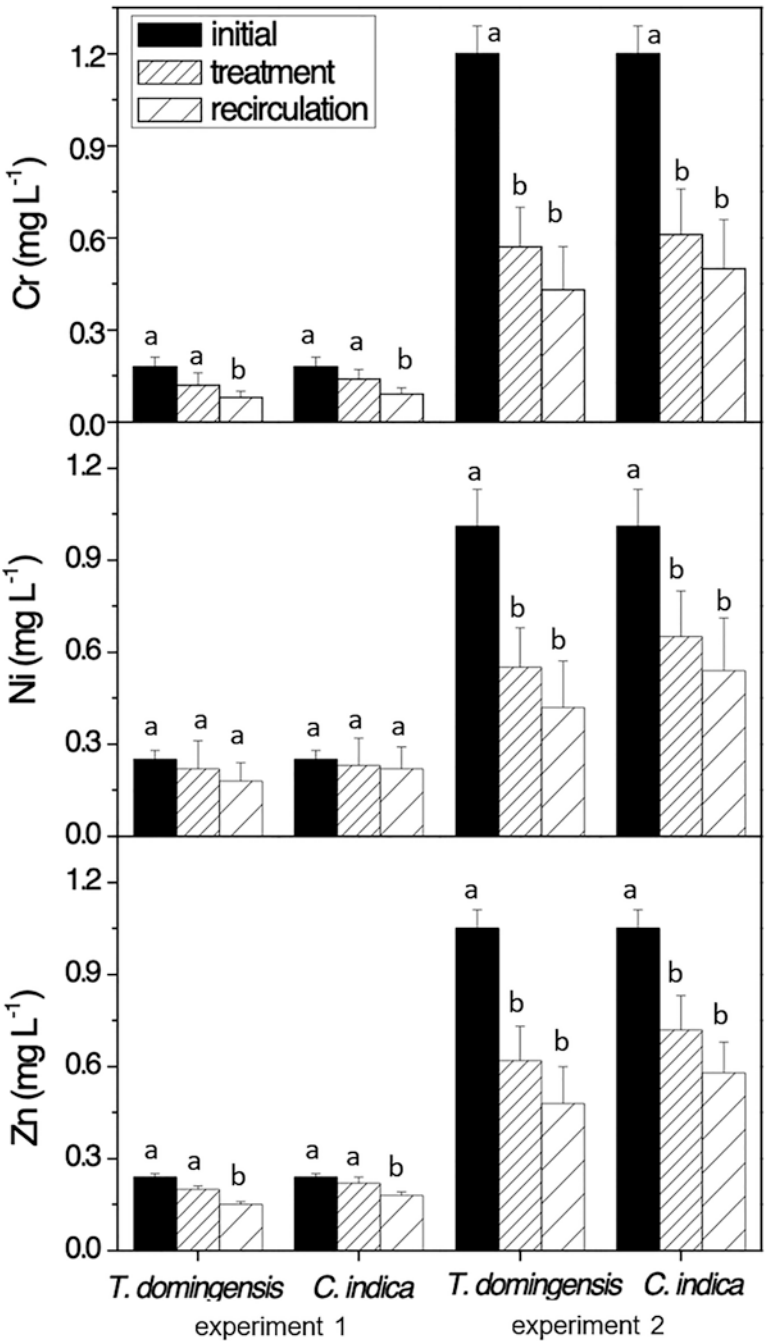

Fig. 3.4 Cr, Ni, and Zn concentrations (mean ± standard deviation) measured in leachate at the beginning, after the VFWs treatments, and after recirculation. The efficiencies of these systems were studied in two experiments, each with different metal initial concentration (0.2 and 1 mg L^{-1}). Different letters represent statistically significant differences among the initial values, after the treatment and after the recirculation, analyzed separately for *C. indica* and *T. domingensis*, and for the two metal concentrations used (Extracted from Maine et al. 2022)

willows could be used as a secondary treatment for this type of effluent. Another tree species is *Melaleuca* spp., which is used in TWs in Australia, where it is native. Bolton and Greenway (1997, 1999) compared the efficiency of *Melaleuca quinquenervia* and *Melalecua alternifolia* in a wetland constructed for the treatment of sewage. Taking into account the growth and accumulation of P in their tissues, both species were suitable for this treatment. In phytoremediation techniques, it is appropriate to use native species and not exotic species. For example, *Melaleuca* spp. are native to Australia but are highly invasive and disruptive in the USA where they were introduced (Turner et al. 1997).

Regarding mangroves, this type of vegetation forms intertidal forests that develop in saline inland water bodies and estuarine sites in warm parts of the world (Ivorra et al. 2021; Maurya et al. 2021). Mangroves located in estuaries may receive contaminant inputs from upstream sources, such as toxic metals (Maurya and Kumari 2021). Mangroves have demonstrated a high capability in metal phytoremediation, showing tolerance to these pollutants (Chowdhury et al. 2017; Zhang et al. 2017). Rahman et al. (2019) proposed that mangroves accumulate metals in their root system by cation exchange, absorption, permeation, and chemical changes in the rhizosphere.

In TWs operating under cold regions, the freezing of the wastewater to be treated, hydraulic short circuits, and stress on biotic components (plants and microorganisms) can occur. Water chemical and physical characteristics, such as TP, TN, and COD, were more efficiently reduced in TWs studied in regions with warm temperatures in comparison with TWs from cold regions (Varma et al. 2021). Regarding plant species, it was observed in cold regions that *Typha* spp. were more efficient in TWs than *Phragmites* spp. (Varma et al. 2021). However, there is experience in the world regarding the efficient operation of TWs in temperate and cold climate regions (Ham et al. 2004; Heyvaert et al. 2006; Jenssen et al. 1993; Kato et al. 2013; Liang et al. 2020; Mæhlum and Stålnacke 1999). To obtain an acceptable efficiency, these systems must be correctly designed according to the climate.

Due to the high plant diversity that exists in the tropical and subtropical regions, it would be valuable to study the efficiency and tolerance of plants of unconventional use in TWs. Over the years, the experience in the use of TWs worldwide indicates the use of species that were not commonly used before, such as the ornamental macrophytes *Zantedeschia aethiopica* and *Canna* spp. (Belmont and Metcalfe 2003; Konnerup et al. 2009; Macci et al. 2014; Zhang et al. 2011). Zurita et al. (2009) studied the efficiency of four ornamental species of commercial value (*Strelitzia reginae*, *Zantedeschia aethiopica*, *Anturium andreanum*, and *Agapanthus africanus*) in two types of HSSF wetlands constructed for the treatment of domestic effluent. These authors concluded that it is possible to produce commercial flowers in a TW without reducing the efficiency of the system.

Table 3.1 shows plant species that are generally used in TWs of tropical and subtropical regions and species that have a potential utility to be used in TWs in these regions, mainly species from South America. Free-floating species, such as *Salvinia* spp. and duckweeds (*Lemna* spp., etc.), were not proposed to be used in TWs because these plants present lower biomass compared to other species, showing a lower pollutant bioaccumulation efficiency.

Table 3.1 Tropical and subtropical species commonly used and some species that can be potentially used in TWs

Family	Species	Growth form	Distribution	Invasive
Acanthaceae	*Avicennia marina*	Tree	Africa, Arabia, India, Australia, Indian Ocean islands	No
Alismataceae	*Sagittaria montevidensis*	Emergent	North and South America	Yes
Amaranthaceae	*Alternanthera philoxeroides*	Emergent	North and South America	Yes
Amaryllidaceae	*Agapanthus africanus*	Emergent	Wideworld, native to Africa	Yes
Apiaceae	*Eryngium eburneum*	Emergent	South America	No
Araceae	*Pistia stratiotes*	Free-floating	Pantropical	Yes
	Zantedeschia aethiopica	Emergent	Central and South America, native to Africa	Yes
Araliaceae	*Hydrocotyle bonariensis*	Emergent	North and South America, Africa	Yes
	Hydrocotyle ranunculoides	Emergent	North and South America, worldwide	Yes
Cannaceae	*Canna glauca*	Emergent	North, Central, and South America	No
	Canna indica	Emergent	North, Central, and South America	No
Cyperaceae	*Cyperus alternifolius*	Emergent	Naturalized in many tropical regions, native to Madagascar	Yes
	Cyperus giganteus	Emergent	North, Central, and South America	No
	Cyperus papyrus	Emergent	North and South America, Africa	Yes
	Schoenoplectus californicus	Emergent	North, Central, and South America	No
Chenopodiaceae	*Sarcocornia perennis*	Emergent	North and South America	No
Compositae	*Senecio bonariensis*	Emergent	South America	No
Convolvulaceae	*Ipomoea aquatica*	Emergent	Central and South America, occurs in Africa, native to Asia	Yes
Iridaceae	*Iris pseudacorus*	Emergent	North and South America, native to Europe	Yes
Marantaceae	*Thalia geniculata*	Emergent	Southeast of North America and Central, and South America	No
Onagraceae	*Ludwigia peploides*	Emergent	North, Central, and South America	No
Poaceae	*Arundo donax*	Emergent	Widespread in tropical and subtropical regions of the world	Yes
	Cortaderia selloana	Emergent	South America	No

(continued)

Table 3.1 (continued)

Family	Species	Growth form	Distribution	Invasive
	Hymenachne amplexicaule	Emergent	Central and South America	No
	Panicum elephantipes	Emergent	Central and South America	Yes
	Paspalum repens	Emergent	North, Central, and South America	Yes
	Phragmites australis	Emergent	Worldwide	Yes
	Vetiveria zizanioides	Emergent	Widespread cultivated, naïve to Asia	Yes
Polygonaceae	*Polygonum punctatum*	Emergent	North, Central, and South America	No
Pontederiaceae	*Eichhornia crassipes*	Free-floating	Naturalized in the tropics of the world, native to South America	Yes
	Pontederia cordata	Emergent	North, Central, and South America, Africa	Yes
	Pontederia rotundifolia	Emergent	Central and South America	No
Salicaceae	*Salix humboldtiana*	Tree	Central and South America	No
Strelitziaceae	*Strelitzia reginae*	Emergent	Widespread cultivated, native to South Africa	Yes
Typhaceae	*Typha angustifolia*	Emergent	Nearly worldwide	Yes
	Typha domingensis	Emergent	North, Central, and South America	Yes
	Typha latifolia	Emergent	North, Central, and South America, Northern Europe	Yes
Zingiberaceae	*Hedychium coronarium*	Emergent	Naturalized in the tropics, native to Indomalasia, Himalayas	Yes

References: Plantas Acuáticas del Río Paraná Medio (Schneider et al. 2021), the world flora online, Flora del Cono Sur-Instituto de Botánica Darwinion website

3.3 Role of Plants in TWs

Plants growing in TWs have various properties that make them an essential component of these systems (Brix 1994). However, their role has been questioned because the most important pollutant removal processes in TWs are based on physical and microbial processes (Brix 1997). In both natural and TWs the macrophytic community produces the highest biomass in comparison with other plant groups, and through its metabolic activity, it is capable of influencing the dynamics of the system in different ways (Esteves 1988). Since macrophytes influence the biogeochemistry of sediments, increasing the environmental diversity of the rhizosphere and favoring chemical and biochemical reactions that improve purification (Jenssen et al. 1993; Opitz et al. 2021), they constitute the main biological component of TWs.

Jamwal et al. (2021) compared the efficiency of two HSSF wetlands, one of them was planted with *Canna indica*, and the other was operated without vegetation. These TWs were constructed at a school in southern India to treat the effluent from septic tanks. Both TWs were operated with the same hydraulic retention time (HRT) of 3.7 days and hydraulic loading rate (HLR) of 84 mm day^{-1}. The study was carried out during one year and the removal efficiencies obtained for BOD, COD, TP, TN, and TSS were significantly higher in the planted TW in comparison with the unplanted system. Probably, a period study of one year was not enough to observe that systems without plants run out of organic matter, which is necessary for the biogeochemical processes that drive the removal of pollutants. Another work compared HSSF wetlands with different substrates, with and without vegetation, and the efficiencies of these systems planted with *Typha domingensis* and *Canna glauca* for the treatment of diluted cheese production wastewater at a microcosms-scale (Nocetti et al. 2020). For some parameters, the vegetated systems showed the highest removals of the measured water parameters (Table 3.2).

Regarding metals, it has been proposed that the mechanisms used by plants to remove them are not necessarily the same for the different species and the different metals. Among these mechanisms are sorption by roots, biological processes, which include translocation to the aerial part, and precipitation induced by root exudates or by microorganisms. It has even been demonstrated that macrophytes not only absorb contaminants when they are alive, but also their dry biomass is capable of adsorbing metals (Dushenkov et al. 1995; Miretzky et al. 2006). As part of their growth cycles, dry biomass is an important compartment in the accumulation of contaminants in a TW. Maine et al. (2017) monitored a wetland constructed for the treatment of an industrial effluent containing Cr. These authors observed that detritus from *T. domingensis* leaves in the inlet zone accumulated high metal concentrations. This would be an important advantage for the management of TWs because when plants die, as their degradation is slow, they continue retaining metals within the wetland, and this debris can be easily removed for final disposal.

In the case of nutrients, Greenway (2007) proposed that their concentrations change with the age of the plants and leaves. Young plants and leaves generally

Table 3.2 COD, TN, NH_4^+-N, TP, and SRP removal efficiencies (%) calculated in the different HSSF wetlands (mean ± standard deviation)

Treatments	COD	TN	NH_4^+-N	TP	SRP
LECA					
Unplanted	84.1 ± 2.2 a	63.4 ± 10.7 a	67 ± 5.4 a	74.9 ± 5.0 a	58.3 ± 3.0 a
C. glauca	84.2 ± 2.8 a	74.6 ± 8.9 a	90.4 ± 7.8 b	81.9 ± 9.3 a	66.5 ± 5.7 b
T. domingensis	83.5 ± 5.0 a	71.5 ± 10.0 a	85.0 ± 2.5 b	81.7 ± 12.5 a	63.9 ± 3.0 b
River stone					
Unplanted	85.1 ± 2.3 a	74.5 ± 4.8 a	85.1 ± 9.0 b	72.2 ± 3.7 a	52.1 ± 2.8 c
C. glauca	83.3 ± 4.3 a	81.2 ± 6.6 b	92.5 ± 5.2 b	80.1 ± 8.9 a	60.1 ± 6.6 b
T. domingensis	83.4 ± 5.7 a	65.7 ± 1.5 a	76.3 ± 4.0 c	72.1 ± 6.8 a	51.9 ± 3.1 c

Different letters represent statistically significant differences among the treatments with the different HSSF wetlands (Extracted from Nocetti et al. 2020) show the highest concentrations of nutrients, especially nitrogen. When the plant reaches maturity, the concentration of nutrients decreases. However, because plant biomass increases with maturity, the total accumulation of nutrients also increases. During senescence, nutrients are translocated from mature leaves to growing shoots or storage organs. Therefore, dead plants have lower concentrations of nitrogen and phosphorus, and harvesting the dead tissues would remove a smaller proportion of nutrients compared to the harvesting of living plants.

Aeration in TWs is an important factor to reach acceptable pollutant removal efficiencies. The entry of oxygen to the system through the vegetation is a key process (Shubiao et al. 2014). Wetland plants possess abundant aerenchymatous tissues that transport oxygen from the aerial parts to the below-ground parts, thereby providing an oxygenated microenvironment within the rhizosphere that stimulates the decomposition of organic matter and the growth of microorganisms (Brix and Schierup 1990; Gersberg et al. 1986; Hadad et al. 2021b; Tanner et al. 2002; Zhang et al. 2014). At the same time, with the production of new biomass and the senescence of old tissues during growth cycles, an anaerobic sludge develops at the bottom of the system that provides favorable conditions for the denitrification process to take place (Vymazal 2013).

In TWs, plants can regulate the phytoplankton density by blocking solar radiation, secretion of allelopathic substances, provision of shelters and habitats for herbivores, and modification of the nutrient regime (Hadad et al. 2021b; Simões et al. 2012). Free-floating macrophytes, such as *E. crassipes*, *P. stratiotes*, *Lemna* spp., *Salvinia* spp., can cover the entire surface of a water body, limiting algal growth due to the scarcity of light. The functions of root exudates are diverse. One case is allelopathy, which is defined as the inhibition of a plant species due to a chemical compound synthesized by another plant species (Putnam 1985; Rice 1984; Szczepanski 1977). However, it is not clear how allelopathy might affect plants in a TW (Vymazal and Kröpfelová 2008).

Lastly, macrophytes can provide suitable habitat for wildlife (Mitsch and Gosselink 2000; Vymazal and Kropfelová 2008) and can improve the aesthetic appearance of the TWs.

3.4 Plant Ecological Dynamics in TWs

When there is enough space for colonization and abundant nutrient availability, aquatic plants show a high growth rate and propagation. This can be explained by their efficient agamic or vegetative reproduction mechanisms through rhizomes and stolons, depending on the species (Hadad et al. 2021a). Free-floating species frequently represent a problem in water bodies that receive a high nutrient discharge because they are able to cover all or a large part of the water surface (Fig. 3.5). Probably the simplest method used to control vegetation is to harvest the plants. Another method is biological control. However, this method takes more time and

Fig. 3.5 Duckweed invasion in a wetland constructed for the treatment of sewage in Mexico (Photo by H.R. Hadad)

could generate unwanted problems such as an infestation of the biological agent, causing changes in the efficiency of the system.

Their high growth rate is the reason why floating species are a concern in natural wetlands. However, this characteristic becomes an advantage when these species are used in TWs (Vymazal and Kröpfelová 2008). In a wetland constructed for the treatment of an effluent from a metallurgical industry, accidental dumping occurred of the raw effluent that contained a high concentration of Cr. This produced a decrease in the cover of *T. domingensis* plants with the consequent dominance of duckweeds (Maine et al. 2017).

Competition among macrophyte species is a factor that could determine the differences in the dominance of the vegetation. The literature available on the competition of macrophytes in natural environments (e.g., Dickinson and Miller 1998; Gaudet and Keddy 1995; Louback-Franco et al. 2020; Mal et al. 1997; Milne et al. 2007; Pulzatto et al. 2019) contrasts with the practically non-existent bibliography on changes in plant dominance in TWs. The possibility of monitoring a TW for many years allows us to know its ecological dynamics (Hadad et al. 2021a).

Another concern is the presence of herbivorous animals, which can have a significant impact on the plant biomass of a TW. Experience has shown that a wide variety of animals can interfere with the desired operating conditions in TWs. Animals that have caused some degree of impact include deer, elk, cattle, pigs, squirrels, manatees, and turtles. However, the most problematic species are birds, rodents, fishes, and mosquitoes (Kadlec and Wallace 2009; Wood et al. 2017).

For example, in North American wetlands, the muskrat (*Ondatra zibethicus*) feeds on a large number of emergent herbaceous plants (Latchum 1996). In TWs, the herbivory impacts can change the dominance of the vegetation, generating from densely vegetated areas to patches of free-vegetation water (Kadlec et al. 2007). In South America, coypu (*Myocastor coypus*) causes problems similar to those caused by the Muskrat in North America. Besides, waterfowl can affect plant productivity through selective herbivory (Bakker et al. 2016).

In Argentina, a free-water surface wetland for the treatment of effluents from a metallurgical industry showed different stages of vegetation dominance (Maine et al. 2013). Because the emergent species *T. domingensis* was more tolerant of the effluent conditions, it became dominant. However, a massive predation of the aerial parts occurred by capybaras (*Hydrochoerus hydrochaeris*) that lived within the industrial property in the vicinity of the TW. This animal is a large semi-aquatic South American rodent with a range weight of 35–70 kg, presenting the adult males at an average weight of 56 kg (Bolkovic et al. 2019). This species is well known to include macrophytes in its diet (Borges and Gonçalves Colares 2007). After the installation of a perimeter fence that prevented the animals from entering the wetland, the plants recovered their biomass. During the herbivory predation period, the plants continued to retain the effluent pollutants in their roots at the same time that the sediment increased its retention capacity. This occurred because the TW reached maturity with the full development of the root-rhizome system. Based on their results, Maine et al. (2013) concluded that a mature TW is capable of maintaining its efficiency and recovering its vegetation in the face of an herbivory predation event, demonstrating its robustness.

3.5 Conclusions

Due to the great plant diversity in tropical and subtropical regions, the use of TWs is a highly applicable nature-based solution. It is important to explore different native species that grow with high productivity in natural wetlands and that could potentially have a high capacity to accumulate different pollutants, but that are not commonly used in TWs. Non-native species should be avoided because they can become invasive.

In addition, it is necessary to deepen the studies focused on the treatment of new pollutants or emerging pollutants through phytoremediation. These studies are highly feasible to be carried out in tropical and subtropical regions due to the high availability of plants and the high density of the human population, which generates a great diversity of pollutants in natural water bodies.

The efficiency of TWs in tropical and subtropical regions has been proven for years. However, it is necessary to continue with studies focused on the role of aquatic plants in TWs, their response to different pollutants, their tolerance, and their ability to accumulate pollutants in tissues, among other variables.

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Chapter 4
Wetlands for Bioremediation in Pakistan

Muhammad Afzal and Ben LePage

Abstract In Pakistan, most of the sewage and industrial wastewater is released into the environment without treatment. This is mainly due to the high capital and operational costs of conventional wastewater treatment systems. Floating treatment wetlands and constructed wetlands use plants and their associated microbes, collectively called a microbial consortium, to treat sewage and industrial wastewater. The use of floating treatment wetlands and constructed wetlands is one of the most applicable approaches for wastewater treatment because of their low capital, maintenance, and operational costs. In floating treatment wetlands and constructed wetlands, the plant-associated microbes play an important role in the remediation of polluted wastewater. In this chapter, we review water contamination, wastewater treatment, and the application of floating treatment wetlands and constructed wetlands at the pilot- and full-scale levels at sites in Pakistan for the remediation of sewage and industrial wastewater. Wastewater treated using these approaches met Pakistan's National Wastewater Discharge Standards, which then deems the treated wastewater safe to be discharged into the environment. Furthermore, the application of floating treatment wetlands and constructed wetlands created habitat for the biota.

Keywords Nature-based solutions · Phytoremediation · Wastewater treatment · Plant-bacteria interactions · Floating wetlands · Constructed wetlands

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4.1 Introduction

Soil and water contamination due to pollution are prevalent problems in Pakistan. According to an International Monetary Fund report, Pakistan may have little clean water left by 2025, and a lack of clean water is a threat to the country's economic stability (IMF 2015; Shukla 2018; Nabi et al. 2019). The discharge of untreated wastewater into the environment, as well as global climate change, are reasons why clean water is lacking in the country. Discharged wastewater often contains high concentrations of toxic chemicals and elements, mostly metals and metalloids such as lead, chromium, mercury, selenium, zinc, arsenic, cadmium, silver, and nickel, and pathogens that deteriorate the quality of ground and surface water resources (Zia and Mehmood 2018; Rehman et al. 2018). Contaminated water is often used for personal consumption and agriculture because there are no other economically reasonable options (Azizullah et al. 2011; Maleksaeidi et al. 2018; Rashid et al. 2018; Cossio et al. 2021). Poor water quality contributes to 80% of the morbidity and 40% of the mortality of the people in Pakistan (Butt et al. 2020). Unfortunately, water purification treatment systems and engineered remediation approaches can be prohibitively expensive to build, operate, and maintain (Nawaz and Ali 2018; Afzal et al. 2019a, b). However, biological approaches can be used to remove pollutants from the environment.

Floating treatment wetlands and constructed wetlands are the engineered application of bioremediation for the treatment and recycling of wastewater. The application of floating treatment wetlands and constructed wetlands are the best options to treat sewage and industrial wastewater because they are rational and ecologically friendly (De Stefani et al. 2011; Afzal et al. 2019a, b; Henny and Kurniawan 2019; Prashant and Billore 2020; Wang et al. 2020). Moreover, the capital, maintenance, and operational costs of floating treatment wetlands and constructed wetlands are low compared to engineered approaches (MacDonald et al. 2016). In the last decade, great strides have been made using floating treatment wetlands and constructed wetlands for the remediation of contaminated water, and their application is becoming more attractive in light of their social, economic, environmental, and sustainability benefits, especially when aligning this technology with the United Nations, Sustainable Development Goals (Chang et al. 2014; Stefanakis 2019). Despite the high potential of floating treatment wetlands and constructed wetlands to remediate contaminated water, the application of such technologies in Pakistan is not well popularized (Elekwachi et al. 2014; Pal et al. 2010; Haydar et al. 2015; Chaudhary and Kim 2019). This is due to the lack of knowledge and interest about wastewater treatment and recycling by policymakers of Pakistan (Zhang et al. 2021). In this chapter, we discuss water contamination, bioremediation approaches, plant-bacteria partnerships, and the use of floating treatment wetlands and constructed wetlands in Pakistan for the treatment and recycling of water contaminated with sewage and industrial waste.

4.2 Water Contamination in Pakistan

In Pakistan, pollutants are being released into the environment via wastewater streams daily. For example, organic pollutants, including heavy metals, hydrocarbons, phenols, and dyes, have been reported in sewage and industrial wastewater streams (Ijaz et al. 2015; Hussain et al. 2018a, b, 2019; Rehman et al. 2018, 2019; Afzal et al. 2019a; Tara et al. 2019). Earlier, Afzal et al. (2014) found that lead, chromium, mercury, selenium, zinc, arsenic, cadmium, silver, and nickel were the most common metals and metalloids in the wastewater streams generated by the leather industry. Most of these metals and metalloids are naturally occurring and present at low levels in the soil and water, but at elevated levels, they are toxic and negatively impact human health and the environment (Pollard et al. 2014). Metals do not degrade and may bioconcentrate in food webs and remain persistent threats to impacted ecosystems (Malik et al. 2004; David et al. 2012; Tchounwou et al. 2012). The discharge of untreated domestic and industrial wastewater has already polluted the sources of ground and surface water in the cities and their surroundings in Pakistan (Azizullah et al. 2011; Daud et al. 2017; Nabi et al. 2019). For example, Afzal et al. (2014) found high concentrations of metals in groundwater from Kasur City, and Daud et al. (2017) and Sahoutara (2017) reported pathogens, fertilizers, pesticides, and industrial pollutants in drinking water samples collected from different areas of Pakistan.

4.3 Wastewater Treatment in Pakistan

Conventional pump and treat systems are effective in removing organic contaminants and metals from polluted water, especially where large amounts of water are being treated (United States Environmental Protection Agency, 1998). However, the capital, operation, and maintenance costs of these systems can be prohibitively expensive (Pandey et al. 2009; Afzal et al. 2019a, b; Kanwar et al. 2019). Therefore, most industries and the public discharge their wastewater to the ground surface, rivers, and streams without primary and secondary treatment. Although some companies have installed conventional wastewater treatment plants, they are few because these systems are expensive to install, operate, and maintain. In addition, most companies that have wastewater treatment systems only treat part of their wastewater stream or run the system during audits to remain in compliance and manage costs.

Constructed wetlands are one of the most applicable approaches to removing pollutants from wastewater streams in Pakistan (Ali et al. 2018). The costs of implementing constructed wetlands technologies are considerably lower than engineered approaches (Dixit et al. 2015; Macaulay and Rees 2014; Afzal et al. 2019a, b). Pollutants are removed from the wastewater through extraction, filtration, stabilization, degradation, and volatilization (United States Environmental Protection Agency 1999; Pilon-Smits 2005; Weyens et al. 2009; Afzal et al. 2019a). The microbial communities are commonly the living component of the soil or water and may already have resistance or tolerance to the contaminants. Their tolerance or resistance to contaminants that they are exposed to and their ability to detect and adapt to changes in contaminant concentrations and environmental conditions make them ideal candidates for identifying and implementing suitable remediation technologies to clean contaminated media (Ahmad et al. 2021; Tchounwou et al. 2012; Issazadeh et al. 2013; French et al. 2020).

4.4 Plant–Bacteria Synergisms in Treatment Wetlands

In floating treatment wetlands and constructed wetlands, plants remove pollutants from the wastewater generally by filtration, adsorption, and absorption (Rehman et al. 2019). There are millions of microbes associated with plant roots that mineralize the organic pollutants that are adsorbed and/or absorbed by the roots (Berg et al. 2005; Manter et al. 2010; Palacios and Winfrey 2021). However, the population of specific pollutant-degrading bacteria decreases in polluted soil and wastewater environments (Glick 2010). The augmentation of specific bacteria that have specific pollutant-degrading activities can be used to increase the population and activity of bacteria in floating treatment wetlands and constructed wetlands, which enhance the removal of organic and inorganic pollutants from the wastewater. For example, Shehzadi et al. (2014) observed that the augmentation of endophytic bacteria in constructed wetlands enhanced the bacterial population in plant roots and shoots, which then increased the removal of organic and inorganic pollutants from textile wastewater streams. Later, Ijaz et al. (2015, 2016) observed that the augmentation of bacteria in floating treatment wetlands enhanced pollutant removal from a sewage wastewater stream. In another study, Rehman et al. (2018, 2019) reported that the augmentation of bacteria in floating treatment wetlands enhanced the metabolic activities of the bacterial population and removal of hydrocarbons from the wastewater stream of an oil exploration company. This might be due to the fact that there were sufficient resources for the bacterial populations to remove the hydrocarbons (Keeling and Palmer 2008; Soucy et al. 2015).

In the plant-bacteria synergism, the rhizosphere and endophytic bacteria colonize plant roots, degrade the organic pollutants, decrease the toxicity of the wastewater stream, and improve plant growth (Shahid et al. 2019; Rehman et al. 2021). Moreover, some bacteria produce 1-aminocyclopropane-1-1carboxylate (ACC)-deaminase, siderophore, and indole acetic acid, which decreases the stress placed on plants due to pollution, improves plant growth, and improves the pollutant-degrading microbial population which subsequenly remove pollutants from the wastewater (Weyens et al. 2009; Glick 2010, Khan et al. 2013).

4.5 Wetland Bioremediation in Pakistan

The end goal of bioremediation is to remove or reduce harmful compounds to improve soil and water quality. In Pakistan, the use of indigenous plants and their associated microbes is an appropriate approach for the cleanup of polluted water (Ijaz et al. 2015; Rehman et al. 2018; Afzal et al. 2019a). According to the proposed classification by Fonder and Headley (2013), the floating treatment wetlands applied in Pakistan are floating emergent macrophyte treatment wetlands. Moreover, horizontal sub-surface flow and vertical down flow constructed wetlands are used in Pakistan to treat wastewater (Ali et al. 2018).

There have been many macrocosm-level pilot studies using microbes and indigenous plants to remediate polluted water initially having no primary or secondary treatment (Hussain et al. 2018a, b, 2019; Tara et al. 2019; Rehman et al. 2021). For example, Tara et al. (2019) used floating treatment wetlands in a pilot-scale study to remediate dye-rich textile wastewater (Fig. 4.1). *Phragmites australis* was planted in 1000-liter plastic tanks and some of the macrocosms were inoculated with the endophytic bacteria, *Acinetobacter junii* strain NT-15, *Rhodococcus* sp. strain NT-3, and *Pseudomonas indoloxydans* strain NT-38. These strains of bacteria possess the ability to remediate textile byproducts in wastewater streams. Over a 22-year period, the performance of the floating treatment wetlands was monitored for the removal of organic and inorganic pollutants from the wastewater. The floating treatment wetlands that were inoculated with these endophytic bacteria performed the best by reducing the chemical oxygen demand by 92%, the biochemical oxygen demand by 91%, and removing 87% of the trace metals from the wastewater. The endophytic bacteria showed good colonization in the roots and shoots of the plants and the high number of bacteria seen in the plant roots and wastewater suggests these bacteria are good candidates for remediating textile wastewater. The results of this study confirmed that the use of floating treatment wetlands for the remediation of textile wastewater is warranted. In another study, Rehman et al. (2021) evaluated the performance of floating treatment wetlands at a pilot-scale level for the remediation of crude oil-polluted wastewater. The floating treatment wetlands were vegetated with *Phragmites australis* and *Typha domingensis* and inoculated with hydrocarbon-degrading bacteria. The floating treatment wetlands possessing *Phragmites australis* performed better and removed 95% of the hydrocarbons from the wastewater. The bacteria used for inoculation colonized the rhizosphere and endosphere of *Phragmites australis* and *Typha domingensis* and were metabolically involved in hydrocarbon degradation.

The potential of constructed wetlands for the remediation of textile-industry wastewater having no initial primary or secondary treatment was also evaluated (Fig. 4.2). For example, the performance of vertical flow constructed wetlands was evaluated at the pilot scale for the remediation of dye-rich wastewater (Hussain et al. 2018a). The vertical flow constructed wetlands were constructed with *Brachiaria mutica* and augmented with endophytic bacteria having textile effluent-degradation capabilities. These wetlands reduced the chemical oxygen demand of

Fig. 4.1 Application of floating treatment wetlands at a pilot scale in the textile industry for the remediation of dye-rich textile wastewater. Vegetation of the plant in the floating mat (**a**), growth of the shoots (**b**–**d**), and roots of *Phragmites australis* (**e**, **f**)

the wastewater by 81%, the biochemical oxygen demand by 92%, and the color by 74%. In a second study, horizontal flow constructed wetlands were vegetated with *Leptochloa fusca* and inoculated with endophytic bacteria that had the ability to remediate textile-based wastewater (Hussain et al. 2018b). The horizontal flow constructed wetlands removed more organic and inorganic pollutants and reduced the chemical oxygen demand by 86% and the biochemical oxygen demand by 95%, and there was more reduction compared to the vertical flow constructed wetlands. In VFCWs, there was 81% chemical oxygen demand and 72% biochemical oxygen demand reduction. The bacteria had colonized the roots and shoots of the plants and

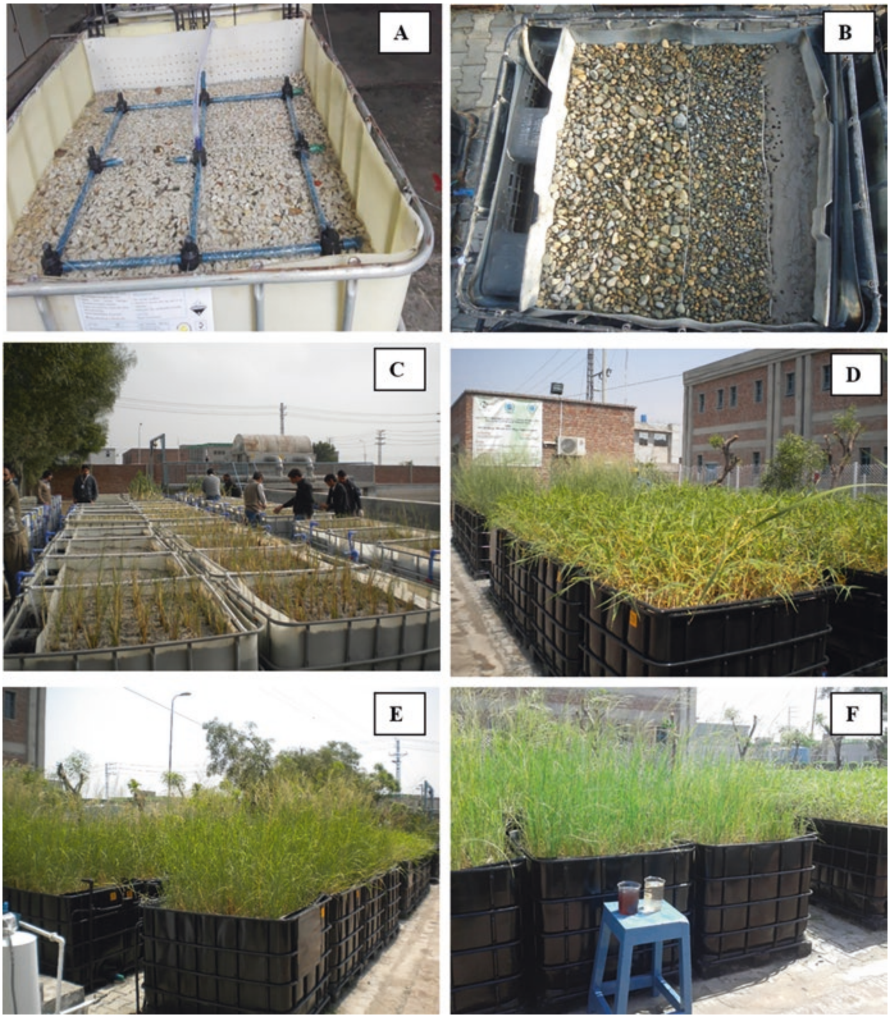

Fig. 4.2 Development and application of constructed wetlands at a pilot scale in the textile industry for the remediation of dye-rich textile wastewater. Vertical flow (**a**), horizontal flow (**b**), constructed wetlands, the vegetation of the cutting of the *Brachiaria mutica* and *Leptochloa fusca* (**c**), growth of the *Brachiaria mutica* (**d**), and growth of the *Leptochloa fusca* (**e**, **f**)

were present in the wastewater. In a third study, the performance of horizontal flow constructed wetlands and vertical flow constructed wetlands was compared at a pilot scale for remediating textile fabric bleach wastewater (Hussain et al. 2019). The constructed wetlands were planted with *Phragmites australis* and inoculated with bacteria having the ability to degrade textile effluent. More pollutant reduction was observed in the horizontal flow constructed wetlands than in the vertical flow constructed wetlands. There was an 89%, 91%, and 95% reduction in the chemical oxygen demand, biochemical oxygen demand, and concentration of total organic carbon in the wastewater treated by the horizontal flow constructed wetlands. These studies revealed that the application of floating treatment wetlands and constructed wetlands is a sound approach to treat contaminated wastewater streams.

Afzal et al. (2019a) used floating treatment wetlands at a full-scale level in sewage and industrial wastewater stabilization ponds in Faisalabad City (Fig. 4.3). This wastewater does not receive any primary or secondary treatment before the

Fig. 4.3 Application of floating treatment wetlands in the sewage wastewater stabilization ponds of Faisalabad City. Vegetation of the plants in the floating mats (**a**), and growth of the plants on the mats (**b**–**e**)

introduction of floating treatment wetlands to the wastewater. Initially, in 2014, 1858 square meter of floating treatment wetlands were deployed and vegetated with *Typha domingensis, Leptochloa fusca, Brachiaria mutica, Cyperus laevigatus, Phragmites australis, Rosa indica*, and *Canna indica* and the area since 2014 has been increased to 3251 square meter. The efficiency of the floating treatment wetlands for reducing the chemical oxygen demand, biochemical oxygen demand, and total dissolved solid concentration was monitored over 3 years, and the results showed that the floating treatment wetlands reduced the chemical oxygen demand, biochemical oxygen demand, and total dissolved solids by 79%, 88%, and 65% respectively (Table 4.1). This system is now treating about 180 million cubic meters of wastewater annually. In other studies, floating treatment wetlands were constructed at Akhuwat University, Kasur City for the treatment of its hostel and offices wastewater (unpublished data; Fig. 4.4a), the town of Manak and Lahore City for a sewage wastewater stream (unpublished data; Fig. 4.4b), and the towns of Gulshan and Jauharabad (unpublished data) for domestic wastewater streams. Recently, floating treatment wetlands were used in three villages located in Lahore City and Multan City for the treatment of sewage wastewater streams (unpublished data) and at Mari Petroleum Limited Ghotki City, Sindh, Province, Pakistan, for the remediation of crude oil-polluted wastewater (unpublished data).

Oil exploration companies in Pakistan store their highly polluted wastewater in pits or discharge the waste directly into the environment without any primary or secondary treatment. Floating treatment wetlands have also been used in the waste pits of oil exploration companies for the remediation of wastewater contaminated with crude oil. For example, a 3058 square meters of floating treatment wetlands were constructed to remediate wastewater contaminated with the crude oil at Rajian Oil, an exploration company in Chakwal (Afzal et al. 2019b). The floating wetland vegetation used to remediate the contaminants consisted of *Phragmites*

Table 4.1 Reduction in pollution level in the sewage wastewater by the application of floating treatment wetlands at Faisalabad, Pakistan

Parameter	Inlet wastewater	Outlet wastewater	NEQS
pH	7.3–8.9	7.2–8.1	6–9
Total dissolved solids (mg/l)	1640–2253	1005–1470	3500
Chemical oxygen demand (mg/l)	460–673	105–140	150
Biochemical oxygen demand (mg/l)	170–235	48–65	80
Chlorides (mg/l)	850–1046	480–630	1000
Nitrogen (mg/l)	41.5–60.4	15.7–20.5	NG
Phosphorus (mg/l)	14.8–21.4	5.2–8.3	NG
Iron (mg/l)	12–15	3.6–5.7	1.0
Chromium (mg/l)	2.5–3.4	0.6–0.8	0.1

mg/l milligrams per liter, *NG* not given in Pakistan's (NEQS) National Environmental Quality Standards list. These results were obtained by analyzing the samples collected between 1 April 2015 and 30 March 2016

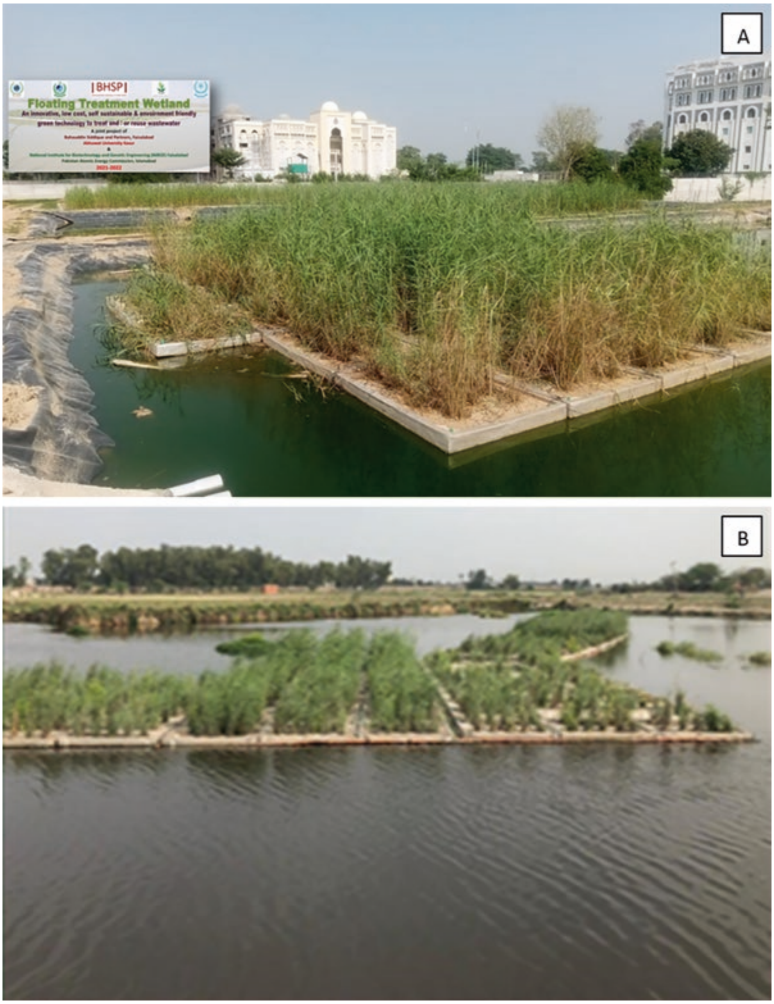

Fig. 4.4 Use of floating treatment wetlands at Akhuwat University, Kasur City, Pakistan (**a**) and Town of Manak, Lahore City, Pakistan (**b**), for the remediation of sewage wastewater

australis, Typha domingensis, Leptochloa fusca, and *Brachiaria mutica* (Fig. 4.5). The plants were also inoculated with hydrocarbon-degrading bacteria that were known to support plant growth and hydrocarbon degradation. The performance of the system was monitored for 18 months, and there was a 97.4%, 98.9%, 82.4%, 99.1%, and 80% reduction in the chemical oxygen demand, biochemical oxygen demand, and total dissolved solids, hydrocarbon, and heavy metal concentrations, respectively (Table 4.2). Among the plants that were used, *Phragmites australis* showed better growth than did *Leptochloa fusca* and *Brachiaria mutica*. In another

Fig. 4.5 The use of floating treatment wetlands in a wastewater pit for the remediation of crude oil-polluted wastewater of Rajian Oil Field, Chakwal City, Pakistan. Vegetation of the plant in the mat (**a**), transfer of the vegetated mats in the pit (**b**), the growth of the plant at the start (**c**), and growth of the plants after 6 months of vegetation (**d**)

Table 4.2 Reduction in pollution level in crude oil polluted wastewater by the application of floating treatment wetlands at Rajian Oil Field, Chakwal, Pakistan

Parameter	Initial value	Final value	NEQS
pH	7.49	7.9	6–9
Oil (mg/l)	319	<10	10
Chemical oxygen demand (mg/l)	1316	142	150
Biochemical oxygen demand (mg/l)	365	36	80
Total dissolved solids (mg/l)	8050	3390	3500
Chlorides (mg/l)	1330	770	1000
Sulphate (mg/l)	432	195	600
Copper (mg/l)	0.2	0.06	1.0
Iron (mg/l)	0.52	0.13	2.0
Chromium (mg/l)	0.20	0.04	0.1

mg/l milligrams per liter

study, 3058 meter squares of floating treatment wetlands were deployed in a wastewater pit in Attock City, Pakistan, to remediate wastewater at the Dakhni Gas Plant (unpublished data; Fig. 4.6). In this study, only *Phragmites australis* was used and the plants were inoculated with bacteria known to degrade hydrocarbons. The floating treatment wetlands removed both organic and inorganic pollutants from the wastewater, and the treated water met Pakistan's National Wastewater Discharge Standards (NEQS, 1997) within 3 months of deploying the floating treatment wetlands.

A 6307 square meter constructed wetland was used to remediate textile wastewater (Nawab et al. 2018). The system has been self-sustaining and maintenance-free for the last 17 years, removing 28–87% of the heavy metals and reducing the chemical oxygen demand of the wastewater by 53%. Among the different wetland plants, *Typha latifolia* and *Phragmites australis* were the dominant species. In another study, constructed wetlands were built at a pilot scale for the remediation of leachate wastewater obtained from domestic and industrial waste dumping site in Islamabad, Pakistan (Batool 2019). The vegetation used in this floating wetland consisted of *Phragmites australis* and *Typha latifolia*, and these plants removed 95%, 91%, and 89% of the copper, zinc, and lead from the wastewater.

Constructed wetlands have been applied at a full-scale level in Pakistan for the remediation of domestic wastewater, such as that seen in Islamabad City, Pakistan (Ali et al. 2018). A reduction of 80% in the chemical oxygen demand, 78% in the biochemical oxygen demand, and a reduction of 81% in the nitrogen and 82% of the total suspended solid concentrations were observed after 1 year. In another study, constructed wetlands vegetated with *Phragmites karka* were developed for the remediation of a domestic wastewater stream with a flow of 1 cubic meter per day (Mustafa 2013). There was a 44% reduction in the chemical oxygen demand,

Fig. 4.6 Application of floating treatment wetlands in a pit at Dakhni Gas Plant, Attock City, Pakistan, for the remediation of crude oil-polluted wastewater

50% reduction in the biochemical oxygen demand, 78% reduction of the total suspended solid concentration, 49% reduction in the ammonium-nitrogen, and 52% reduction in the phosphate-phosphoreous concentrations in the wastewater stream, and the treated wastewater met Pakistan's National Wastewater Discharge Standards. At the Pakistan Agriculture Research Council in Islamabad City, vertical flow constructed wetlands were developed for the remediation of domestic wastewater (Bibi et al. 2011). There was a 92% reduction in the chemical oxygen demand, 98% reduction in the biochemical oxygen demand, and a 41% reduction in the total dissolved solid concentration. In another study, vertical flow constructed wetlands were established at a pilot scale for the remediation of oil refinery wastewater (Mustafa and Ali 2014). Vertical flow constructed wetlands were developed and vegetated with *Phragmites karka* and *Typha domingensis*. There was an 83% reduction in the oil and grease concentration, 72% in the chemical oxygen demand, and 73% in the total dissolved solid concentration seen in the wastewater. Between the two plants, the performance of constructed wetlands vegetated with *Typha domingensis* was better than with *Phragmites karka*.

Floating treatment wetlands and constructed wetlands have also been integrated to treat and reuse wastewater at Toyota Lyallpur Motors, Faisalabad City (Fig. 4.7), Toyota Chenab Motors, Faisalabad City, and Momentum Logistics, Khanewal City, for the treatment and reuse of the car-wash wastewater. At all three sites, the treated water is being recycled for the car washing process.

4.6 Conclusions

In Pakistan, more than 99% of the wastewater that is produced is discharged into the environment without primary or secondary treatment. This is due mainly to the high capital, operation, and maintenance costs of conventional wastewater treatment systems. The use of floating treatment wetlands and constructed wetlands is one of the most applicable and cost-effective approaches for wastewater treatment and water reuse in Pakistan. Recently, the floating treatment wetlands and constructed wetlands have been applied in Pakistan at the pilot and full scales for the remediation of sewage and industrial wastewater streams and the treated water met Pakistan's National Wastewater Discharge Standards. In addition, the use of pollutant-degrading bacteria in floating treatment wetlands and constructed wetlands has enhanced the efficacy of wastewater treatment. More input/examples may be required to convince Pakistan's policymakers that the construction of floating treatment wetlands and constructed wetlands for the remediation of wastewater sources in Pakistan is appropriate and the direction needed as global issues begin to compound, impacting human health and the environment.

Fig. 4.7 Integration of floating treatment wetlands and constructed wetlands at a car-wash station for the treatment and reuse of the wastewater

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Chapter 5
Urban Wetlands in the Tropics – Taiwan as an Example

Wei-Ta Fang, Chia-Hsuan Hsu, Ben LePage, and Chin-Ching Liu

Abstract We discuss the importance and role of urban wetlands using Taiwan as a model to illustrate how water could be managed to support a growing world population while facing the social, economic, and environmental uncertainties that arise as the effects of global change become more prominent. Taiwan was selected because the island is densely populated, rainfall is high, and landslides, earthquakes, and typhoons are common. These parameters, coupled with the impacts of population growth and global change, make Taiwan a good model where innovative strategies can be developed to manage water in an environmentally responsible and sustainable manner. By 2050, the global population is expected to increase by about two billion people and urban areas are expected to expand by about 30%. More people and a lack of space combined with sea-level rise and other aspects of global change are predicted to have substantial impacts on the world's major coastal cities, such as Taipei. As such, innovative approaches to urban planning, including the development of strategies to recycle and use water sustainably, will be needed to maintain food and water security. Examples of constructed wetlands and the benefits provided are presented.

Keywords Constructed wetlands · Food and water security · Sponge city · Taiwan · Wise use

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5.1 Introduction

As the effects of global change are becoming more substantial with respect to the environment, society, and economics, the recycling and sustainable use of our natural resources are becoming more prevalent. Sponge City is an urban water management approach that is being implemented to strengthen the ecological infrastructure and drainage systems in Chinese megacities. Low-Impact Development, Green Infrastructure, and nature-based solutions are being used to reduce erosion and flooding, improve riparian base flow, increase urban cooling, drought resistance, and carbon capture, and minimize the effects of flooding in cities while increasing sub-surface water supplies and improving water quality (EC 2013; Li et al. 2016; Chan et al. 2018; Wang et al. 2018a). Over the last 10 years Sponge City pilot studies have been initiated in at least 30 cities in China (Lashford et al. 2019). In Wuhan, for example, the increase in impervious cover that occurred throughout the city over 40 years of development has decreased the ability of the Yangtze and Han Rivers, their tributaries, and floodplains to store and desynchronize stormwater flow (Liu et al. 2014). As a result, flash flooding is more prevalent and flood water is now reaching the city's underground infrastructure, creating substantial economic losses and impacts on human lives (Wu et al. 2020). To determine whether implementing the Low-Impact Development, Green Infrastructure, and nature-based solutions would be the best management practices to achieve the Sponge City goals and gain stakeholder support, Hou et al. (2019) developed a computer model illustrating how the use of these practices weakened the effects of rain, heat, and pollution in cities. Healthy ecosystems and quality of life are conjoined, and efforts to remedy urban water issues to achieve environmentally sustainable development in cities are essential. The Sponge City concept is complex and requires a good understanding of the local ecology and stakeholder needs. Although hard engineering strategies alone could be used to achieve the Sponge City goals, the interactions between the physical, biological, and social variables need to be considered. Re-establishing natural water cycles where they've been impacted provides important benefits to the environment and people (Barron et al. 1989; McQuade 2022).

In this chapter, we discuss the importance of constructing wetlands in urban ecosystems in light of global change, that is, bringing nature into the cities. The growing human population and decreasing area available for living space, coupled with the increasing need for clean water and food and the infrastructure needed to manage water and other natural resources sustainably, will require innovative approaches and changes in our behavior toward the environment. In 2022, Taiwan was ranked as the 16th most densely populated country in the world (https://worldpopulationreview.com/countries/taiwan-population). The experiences of the Taiwanese people may provide good examples for water management and sustainability for many large coastal population centers around the world. We discuss some of the global issues that the world and Taiwan will experience in the coming years in the context of water availability in the future.

5.1.1 Stakeholders

People are interacting less with nature, and the ability to recognize and understand biological and earth system processes is being lost. People have become disconnected and are less likely to interact with nature, a phenomenon that Pyle (1993) called extinction of experience (Soga and Gaston 2016). Shanahan et al. (2015) consider this phenomenon to be a major public health issue. Continued urbanization is largely at fault because the opportunity to have human-nature interactions and develop an emotional connectedness with nature in cities is becoming more difficult (Soga and Gaston 2016). Consequences of this phenomenon include changes in health and well-being, emotions, attitudes, and behaviors toward nature (Soga and Gaston 2016). Therefore, interactions with and between stakeholders require robust engagement and communication programs.

Scientists and environmental educators have the responsibility to frame and explain complex environmental issues in a manner that all stakeholders understand. Therein lies the problem. Many scientists follow the Deficit Model, where scientific data are presented by the scientists to non-specialists and the scientists naturally assume that the non-specialists will fully understand the information that they've been presented and its importance. As such, stakeholder discussions involving the need to preserve complex ecosystems such as wetlands are often heated because the stakeholders see the wetlands as nothing more than mosquito and disease breeding areas. Convincing non-specialists that such ecosystems are valuable and preserving them in an environmentally responsible and sustainable manner can be challenging to scientists that have little to no experience talking to stakeholders. Furthermore, ignoring or minimizing stakeholders that may have knowledge or a better understanding of the local conditions creates obstacles in the design and implementation of programs that are aimed to improve the environment and the human condition (Wang et al. 2018b, 2019; Zevenbergen et al. 2018).

In cities where flooding has become more pervasive, models that enabled stakeholders to visualize flooding events in three dimensions together with sound scientific explanations needed to mitigate such flooding events have been successfully used to explain concepts like Sponge City to non-specialists (Wang et al. 2019). Wang et al. (2019) indicated that this form of engagement allowed the people to visualize and ask questions so they could better understand why the infrastructure modifications were needed to enhance water interception, infiltration, and purification pathways in a Sponge City model. Intergroup and social interactions can substantially influence people's attitudes, beliefs, and actions on global change and environmental issues (Fielding and Hornsey 2016). We presume that as more people move into cities, phenomena such as extinction of experience and the potential to increase negative attitudes toward nature (biophobia) will grow. Biophobia is the fear of living things and aversion and an alienation from nature (Simaika and Samways 2010). Ironically, the inclusion of wetlands and other types of natural ecosystems in our cities may play an important role in providing growing populations with improved human health and wellness and clean water and food in the future. As Soga and Gaston (2016) point out, many people, including city planners and policymakers, consider urban greenspace and natural environmental elements in cities to be luxuries rather than necessities. While this may have been true in the past, and contrary to some opinions (Larondelle et al. 2014; Haase et al. 2017), integrating nature into the city is probably now a necessity for survival.

5.1.2 Global Change – Sea Level Rise

Between 1902 and 2015, the global mean sea level rise was 0.12–0.21 m (IPCC 2019), and by 2100 it is projected to be in the range of 0.29–1.10 m (McMichael et al. 2020). Estimates of sea-level rise vary substantially as does the number of people that will be impacted. The International Panel on Climate Change projected a high-end estimate for a sea-level rise of 0.6 m by 2099 (IPCC 2007), but this value was raised to 0.9 m in their 2014 report (IPCC 2014; Mooney 2017). Others suggest that a sea-level rise of 2.0–2.7 m this century is possible (NOAA 2017; WCRP 2018; Bamber et al. 2019). These new projected values are based on revised thermal expansion estimates of the water, but they don't account for the amount of ice from Greenland and Antarctica that actually melts, so global sea-level rise could be much higher. The impacts of sea-level rise on humanity will be substantial. Andrews (2021) reported that a projected global sea-level rise of 40 cm by 2050 would put at least 800 million people at risk. Kulp and Strauss (2019) estimate about one billion people live in areas that are less than 10 m above the current high tide line, and their projected sea level elevation data triple the estimates of the vulnerability to sea-level rise and coastal flooding (Kulp and Strauss 2019). Despite all of the noise around global change, we need to remind ourselves that a sea level rise of 0.3–1.0 m is inevitable and that the geographic area and number of people that will be impacted are underestimated (Bilbao et al. 2015). Depending on Antarctica's ice contribution and the amount of atmospheric warming, others indicate a sea-level rise of 0.61–1.10 m relative to sea levels from the 1950s is likely if global warming exceeds four degrees centigrade by 2100 (Siegert et al. 2020).

Although a discussion of Taiwan's geologic history seems out of place, a brief discussion is warranted. Taiwan formed through the collision of the Philippine (Luzon Volcanic Arc) and Eurasian Plates approximately four to eight million years ago (Sibuet and Hsu 2004). Most of the Philippine Plate is still being subducted (pushed) under the Eurasian Plate and those parts that are not being subducted are being "scraped off" and pushed up and out of the ocean, forming mountain ranges. Taiwan is a good example of this process, and because the mountain building-process is still active today, the mountains are steep, erode easily, and landslides are prevalent. Five mountain ranges greater than 3000 m in elevation extend from the south to the north in a North-North-East direction along the spine of the island (Sibuet and Hsu 2004). They dominate the topography and play an important role in shaping Taiwan's climate and weather patterns (Kanehama et al. 2019; Babaei et al. 2021). Because of the mountains, three to four typhoons strike Taiwan annually, and typhoons are a trigger for landslides (Chiang and Chang 2011). Global change will increase the number of typhoons that impact Taiwan and their intensity and subsequent number of landslides is expected to increase. Chiang and Chang (2011) show that rainfall and areas of instability will increase by about 12–15% by 2099. This does not portend well for water recycling and sustainability programs that do not consider managing potential impacts to the environment and human health.

5.1.3 Global Population, Land, and Global Water Use

Population growth and global change play heavily into how water will be used and managed in the future. In July 2022, there were 7.8 billion people in the world (Vollset et al. 2020), and the global population is expected to increase substantially in the next 78 years (Table 5.1; UN 2015). Today, there are about 4.2 billion people that live in urban areas and 3.4 billion people in rural areas (https://ourworldindata.org/urbanization#how-many-people-will-live-in-urban-areas-in-the-future). By 2050, the United Nations (2018) predicts that about 6.7 billion people will live in cities and 3.1 billion people in rural areas. This is a 62% increase in the number of people living in cities compared to today. Therefore, we should be concerned about population growth in the cities. Li et al. (2019) found that the urban land area that people would occupy would increase by roughly 40–67% until 2050 and that this trend would continue to more than 200% by 2100 relative to the areas classified as urban in 2013. The increased need for space comes at a time when food, clean water, and more infrastructure are needed as global climate change and rising sea levels become more prevalent.

As the global population grows, the amount of water needed daily for personal, agricultural, and industrial uses will grow too. The global average for water consumption was 3794 liters per person per day in 2012. Using this value and the projected number of people in 2030, 2050, and 2100 the average amount of water needed per capita is presented in Table 5.1. While the increased amount of water is substantial, it will need to be found. Famiglietti (2014) indicates most aquifers are being mined at unsustainable rates and at least two billion people rely on groundwater as their primary water source, and more than one-half of the world's food is supplied from underground water sources (Rodell et al. 2019). In addition, the ancillary infrastructure needed to manage the volume of water coming into as well as out of our transmission, distribution, and treatment systems will need to be

Table 5.1 Increase in the amount of water relative to population growth

Year	Number of people	Liters of water per person per day	Percent increase from 2022
2022	7,800,000,000	29,593,200,000,000	–
2030	8,500,000,000	32,249,000,000,000	9.0
2050	9,700,000,000	36,801,800,000,000	24.4
2100	11,200,000,000	42,492,800,000,000	43.6

upgraded or built. The United Nations Population Fund (2001) warned that the world will begin to run out of fresh water by the year 2050, which is probably coincident with the trend in population growth (Shaikh 2017).

In 2017, the amount of freshwater extracted annually from streams, lakes, groundwater, and reservoirs was 3881 cubic kilometers per year, which is about 15 and a half times as much as the 600 cubic kilometers per year that are thought to have been extracted in 1900 (United Nations 2021, 2022). The data show Asia extracts 2497 cubic kilometers or 64.3% of the water extracted globally, and of this, 71.3% (2766 cubic kilometers) is used for agricultural purposes (Table 5.2). Comparison of the amount of water used by each continent and globally compared to the renewal amounts show Asia extracts 41.1 and 6.7% of the amount of water received. Globally, 10.5% (3881 cubic kilometers) of water received is used for agricultural, domestic, and industrial purposes, and of this amount, 7.5% (2766 cubic kilometers) is used for agriculture (Tables 5.2 and 5.3; United Nations 2022). Said another way, 60 times more water accumulates in South America compared to the amount used. In Asia that value drops to 2.4, indicating much of the water received is being used. However, we caution readers on how these values should be used or what they really mean because the amounts may be a reflection of continental climate and precipitation rather than anthropogenic use. That is, it just rains less in Asia compared to South America. Nonetheless, the United Nations (2022) indicates that the rate of water use increases by about 1% annually, which is consistent with the population growth rate.

An underlying issue that has not been fully considered is the management of gray water and sewage that an additional two billion people will produce. On average, a person produces about one-half of a kilogram of feces per day (https://www.livescience.com/61966-how-much-you-poop-in-lifetime.html) plus the water used to flush the waste. Infrastructure in many cities is meeting the end of its lifecycle and replacing the thousands of kilometers of piping and treatment facilities needed to process the additional waste and water will be expensive. For example, the American Society of Civil Engineers indicated a 10.45 billion-dollar investment will be needed to improve the existing drinking water, wastewater, and stormwater infrastructure in the United States between 2020 and 2029 (ASCE 2021). These cost estimates are based on the existing infrastructure and do not consider population

Table 5.2 Cubic kilometers of water extracted in 2017

Continent	Agriculture	Domestic	Industrial	Amount (2017)	Percent
North America	268	85	249	602	15.5
South America	150	38	24	212	5.4
Europe	84	64	129	277	7.1
Africa	210	33	16	259	6.7
Asia	2038	238	229	2497	64.3
Oceania	16	5	5	26	0.6
Total	2766	463	652	3881	–
Percent	71.3	11.9	16.8	–	–

Table 5.3 Cubic kilometers of water extracted in 2017 compared to the 2015 renewal amounts

Continent	Renewal (2015)	Extraction (2017)	Percent extracted relative to the amount renewed by continent	Percent relative to the amount renewed globally	Amount added compared to that extracted by continent (x)
N. America	6812	602	8.8	1.6	11.3
S. America	12,724	212	1.6	0.6	60
Europe	6577	277	4.2	0.7	24
Africa	3931	259	6.6	0.7	15.2
Asia	6071	2497	41.1	6.7	2.4
Oceania	902	26	2.9	0.07	34.7
Total	37,017	3881	–	10.5	9.5
Agriculture	Unknown	2766	Unknown	7.5	13.4

growth, new technologies, or changes to state and federal discharge regulations and only consider the existing infrastructure and regulatory framework.

Alternatively, constructed wetlands provide great opportunities to meet the anticipated demands related to the increased volume of water that would need to be treated the waste and more stringent requirements in meeting future screening values. Screening values are defined as the maximum concentration of a chemical in treated soil, water, or air that can be released into the environment (IRTC 2005). The Elk River Wastewater Treatment Plant in Eureka, California, United States, is a good example of integrating biological and engineered solutions. This facility combined 56 hectares of freshwater and tidal marshes, ponds, and riparian habitat with a traditional sewage treatment facility that was designed to treat the sewage generated by about 45,000 people (Eureka 2022). Perhaps a downside to building more treatment wetlands such as the Eureka facility is that the space needed to build the wetlands will be competing with land needed for people to live and produce food. As noted previously, the area covered by the world's cities is expected to triple in the next 30 years. There are cities in Taiwan that serve as good examples of where treatment wetlands are being used to manage sewage (Hsieh et al. 2013, 2015; Shiau et al. 2022; Tan et al. 2017). Nonetheless, novel ideas and pilot studies to implement sustainable water storage, recycling, and waste treatment systems need to be designed and tested.

5.2 Water in Taiwan

Taiwan's climate is subtropical to tropical and monsoonal. Between 1949 and 2009, the average annual rainfall ranged from 2154 to 2932 millimeters, with a country average of 2510 mm (https://eng.wra.gov.tw/cp.aspx?n=5121). This is much greater than the world average of 870 mm (Lubbe 2019). The mountains, steep topography, high rainfall, and earthquake frequency contribute to high rates of erosion, frequent landslides, and flashy river systems (Lee et al. 2016). Taiwan has a population of about 23,886,253 people (as of 2/12/2022) with a density of 673 people per square kilometer (Worldometers 2022). It is one of the most densely populated places on Earth (Ugo 2022) and the effects of global change and population growth will add many more people to an already crowded space. Consequently, Taiwan and many of its cities can serve as ideal examples where sustainable water management approaches can be developed.

Between 2000 and 2009, Taiwan received 95.0 billion tons of rain annually, which was divided into runoff, evaporation, and groundwater (Table 5.4; https://en.wikipedia.org/wiki/Water_supply_and_sanitation_in_Taiwan#:~:text=From%20the%20surface%20runoff%20water,tons%20became%20river%20water%20diversion). The runoff amount was further subdivided into discharge to the sea, reservoirs, and diversions (Table 5.5; https://eng.wra.gov.tw/7618/7664/7718/7724/12929/). Of the total amount received, roughly, 74.7 (78.6%) of the 95 billion tons of water were discharged to the sea or evaporated. We presumed that the water transpired by the plants is included in the evaporation value. The remaining 21 tons (22%) is the water used for personal and industrial uses that was obtained from the reservoir, groundwater, and diversion supplies. The Taiwan Water Corporation defines water diversion as the process of sending water from its head point to a water purification system. Although the water used for domestic and industrial purposes is replaced annually, we assume that most of the remaining 15.4 tons are discharged to the sea as power plant cooling water, sewage, and gray water, lost in agriculture by transpiration and evaporation, as well as commercial and industrial processing (e.g., semiconductor chips). Innovative agricultural, water storage, recycling, and sustainability strategies will need to be incorporated into the future urban planning designs. Not only does every living organism need water to survive, but the environmental, social, and economic benefits are also reliant on a clean and constant supply of fresh, clean water.

5.2.1 Combined (Domestic and Industrial)

Water use values in the literature vary considerably. For example, the Dutch Water Sector indicates the real water use of the average world citizen is 4000 liters per person per day (https://www.dutchwatersector.com/news/real-water-use-of-the-average-world-citizens-is-an-astonishing-4000-liter-per-day). We use Hoekstra and Mekonnen's global water use average of 3794 liters per person per day. Using the equation below, the amount of water for domestic and industrial use in Taiwan is

Table 5.4 Annual rainfall in Taiwan and the amount that evaporated, became runoff, and recharged the aquifers

	Annual rainfall	Evaporation	Runoff	Groundwater
Tons of water	95.0	20.0	70.10	5.60
Percent	–	21	74	5

Table 5.5 Runoff amount that was discharged to the Sea, retained in reservoirs, and entered diversion systems

	Runoff	Discharge to sea	Reservoirs	Diversion
Tons of water	70.10	54.70	4.35	11.05
Percent of runoff amount	–	78.0	6.2	15.7
Percent of total annual rainfall	74	57.6	4.6	11.6

about 32.5 billion tons per year. This is about double the 16.2 tons that are available in reservoirs and diversions (Table 5.5). Despite being in a drought, the amount of water used in Taiwan on a per capita basis is substantially less than the global average of 3794 liters per person per day. In fact, if we assumed that every drop of water in the reservoirs and diversions were used and replaced annually, the average water use per person would need to be 1900 liters per person per day or less. This suggests that water from other sources that are not being measured are being used. Unfortunately, finding and using a value that has the potential to change daily from study to study is well outside of the scope of this paper, but it certainly warrants further study.

- 1 ton water = 1018.32 l
- 23,886,253 people × 3724 liters per person per day = 90,624,443,882 liters per day/1018.32 l = 89,022,047.03 tons per day * 365 days = 32,493,047,167.91 tons per year.

By 2050, and presuming Taiwan's population will increase by 25% from 23,886,253 to 29,917,352 people, the number of liters per person per day will need to drop from 3724 to about 1500 liters per person per day to stay within the 16.2 billion tons of water that are classified as reservoir and diversion water.

5.2.2 *Domestic*

Reliable data related to water use per person and the activities included under domestic, commercial, agricultural, and industrial uses vary substantially in the literature. Nonetheless, we've used values that are often cited in the literature and seem reasonable. In this case, we use 289 liters per person per day for the Taiwanese. [https://www.statista.com/statistics/319859/taiwan-monthly-per-capita-monthly-water-consumption/].

- 23,886,253 people × 289 liters per person per day = 6,903,127,117 liters per day/1018.32 l = 6,778,937 tons per day × * 365 days = 2,474,312,002 tons per year.

Therefore, about 2.5 billion tons of water per year are needed for domestic use if we use Taiwan's current population and a water use value of 289 liters per person per day. If we use the current water use and increase the population by 25% due to population growth, then about 3.1 billion tons of water per year will be needed for domestic use in Taiwan in 2050. Although this amount is less than the 4.6 billion tons of water stored in the reservoirs (Table 5.5), we may be at a tipping point. The United Nations warned that if the current trends continue, the world could face a water availability shortfall by 2030 (United Nations 2016). During Taiwan's rapid economic and urban development, much of the green space in Taipei and other cities throughout the world was replaced with impervious cover, resulting in the loss of the land's original water storage and infiltration functions. In addition, Taipei is built on a predominantly marine basin that has a complex geologic history. The Taipei Basin was originally a shallow coastal to marine basin and about two million years ago, the area that was pulled apart and was filled with sediment from the surrounding mountains and volcanoes as the land surface dropped (Teng et al. 2001). Shallow marine, coastal, estuarine, and terrestrial deposits are evident of its complex geologic history. Jang and Liu indicate that the southern part of the Taipei Basin is suitable for potable water. Aquifers that are being pumped for water should be monitored closely for subsidence and recharged to prevent subsidence. The ability to store fresh water safely has been made difficult because of the mountainous terrain, young geology, and tectonic setting, which makes the construction of reservoirs for water storage expensive, environmentally unfriendly, and puts the public at risk should a reservoir fail. Storing, distributing, and managing water in the future will require innovative interdisciplinary approaches to plan, develop, operate, and make water management decisions.

5.3 Urban Wetlands

Urban water management requires an interdisciplinary approach for planning, developing, operating, and making decisions that influence how the resources are managed. More importantly, a good water management plan is needed to ensure safe, clean water is made available to the public and not wasted. Ideally, a closed-loop strategy would help provide fresh, clean water for domestic, commercial, industrial, agricultural, and environmental uses in a responsible and sustainable manner before returning it to the ground. Urban wetlands should be considered as a water management alternative to compensate for or replace some of the ecosystem services that the natural wetlands that were destroyed during the country's economic development phase once provided.

As discussed previously, the Sponge City concept is based on incorporating and/or retrofitting cities with Low-Impact Development, Green Infrastructure, and nature-based solutions Best Management Practices, as well as constructed wetlands into urban space (USEPA 2000, 2006; Liu et al. 2017). The concept emphasizes the conversion of impervious surfaces to permeable surfaces and incorporating Low-Impact Development, Green Infrastructure, and nature-based solutions Best Management Practices to better manage stormwater and pollutants, provide habitat, and education sites (EC 2013; Li et al. 2016; Chan et al. 2018; Wang et al. 2018a).

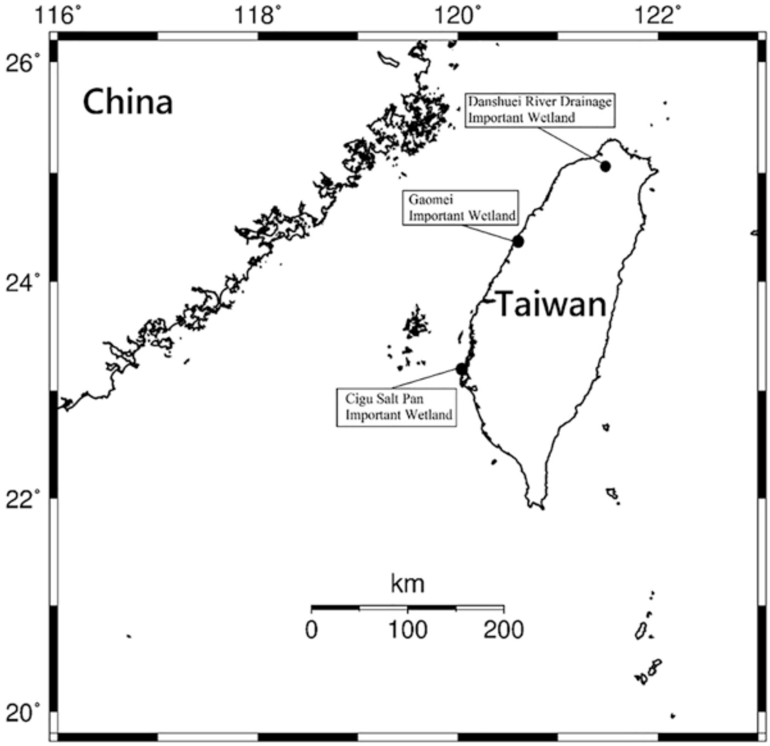

Fig. 5.1 The locations of the three urban wetlands of importance that are discussed in this chapter. (Illustrated by Chia-Hsuan Hsu)

Urban wetlands can collect and treat nutrients that enter municipal wastewater streams, support high levels of biodiversity, and maintain the ecological functions of natural wetlands and they should be incorporated into urban designs and landscapes of the future. The goal of Sponge City is to increase the area of urban land area that is able to absorb surface water by 20% and retain or reuse about 70% of the stormwater generated in the city by 2020 and up to 80% by the 2030s.

Traditionally, urban planners have emphasized the importance of increasing green space (Zou and Wang 2021). However, it's not been easy to incorporate green areas into urban settings instead of using the space for residential, commercial, industrial, and agricultural use. Taiwan has been constructing wetlands in urban areas for at least 30 years to manage stormwater and create habitat for the benefit of humans and the environment (Hu and Sun 2015). Below are three examples in which we discuss the biotic and abiotic benefits of urban wetlands in Taiwan (Fig. 5.1 and Table 5.6).

Table 5.6 Three wetlands of importance in Taiwan

Wetland	Area (hectares)	Category	Authorities
Danshui River Wetland	1788	Coastal, manmade, and inland wetlands	Taiwan International Institute for Water Education
Gaomei Wetland	734	Natural coastal wetlands	Taichung City Government
Cigu Salt Pan Wetland	3697	Natural coastal wetlands and small manmade wetlands	Taijiang National Park

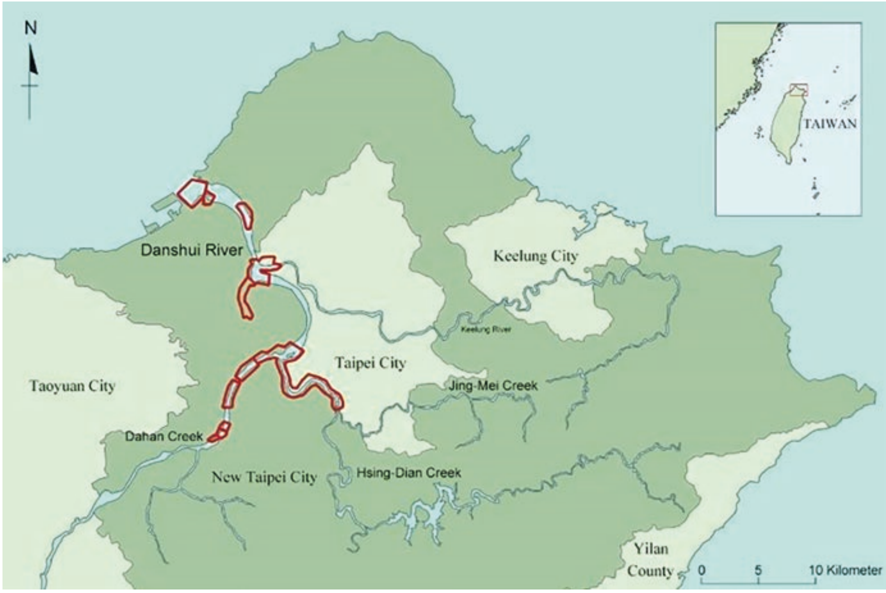

Fig. 5.2 The Danshui River wetlands are located in the red polygons. (Illustrated by Yi-Te Chiang)

5.3.1 Danshui River Wetlands

The Danshui River wetland is located in the third-largest watershed in Taiwan and adjacent to Taipei and New Taipei City (Fig. 5.2). The wetlands encompass an area of about 1788 hectares and are situated along tributaries of the Danshui River. Its location near Taipei, a region that has the highest population density and heavy land use practices, has negatively impacted water quality and the associated wetland functions substantially (Chen et al. 2011).

Water quality was severely impacted due to human activities, especially before a sewage treatment system was built in 1997 (Putri et al. 2018). However, the Danshui River and wetlands are of a substantial benefit to the people by providing cooling in Taipei, reducing flooding, and providing an area for recreation, human health and

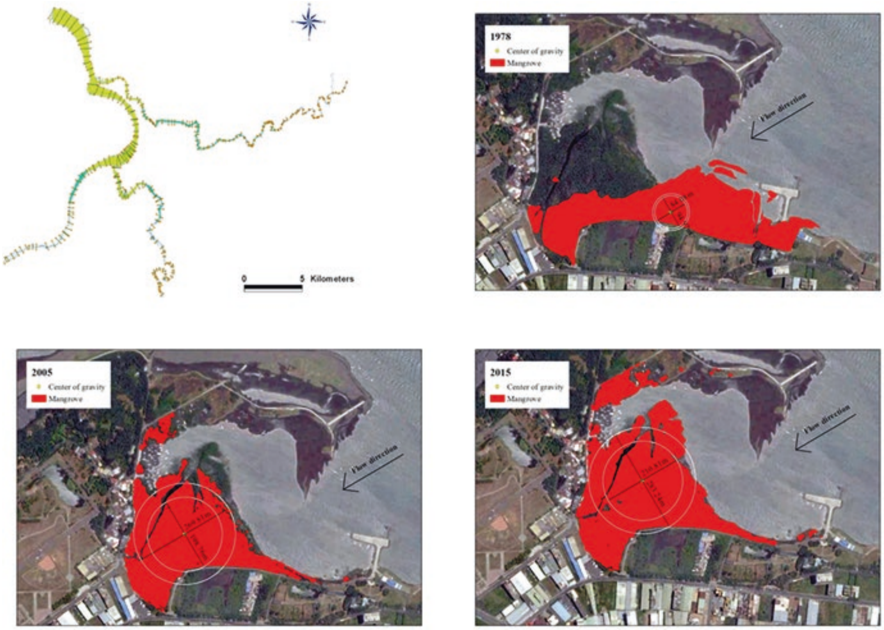

Fig. 5.3 The Danshui River estuary showing encroachment of the mangroves from 1978, 2005, and 2015. (Illustrated by Shang-Su Shih, see Shih et al. (2015a, b))

well-being (Hsieh et al. 2015), and wildlife habitat. Moreover, the naturally occurring microbial consortia bioremediate pollutants that enter these systems in the wastewater and sewage streams.

The Danshui River estuary is roughly 40 hectares in size and home to one of the largest *Kandelia candel* (mangrove) population in the world (Fig. 5.2). Mangroves have evolved special adaptations for life in wetlands, such as vivipary, snorkeling, porous roots for gas exchange, and structurally supporting roots. Therefore, they provide coastal protection from wind and anti-tidal erosion and maintain the rich ecology of Danshui River estuarine wetlands. They turn into “Aquatic forests” during high tide. Due to siltation of the Danshui River mouth, mangrove forests have been expanding (Fig. 5.3). The estuary is a typical estuarine ecosystem that intercepts a large volume of sediment and organic matter from the upstream freshwater rivers. However, due to the pollution of the river water and sand mining, the survival of mangroves is threatened. In order to preserve the ecosystem formed by mangroves and their associated animals and plants and avoid human disturbance, the Agricultural Committee designated the area as the protected “Wazihwei Nature Reserve” in January 1983.

Because the abundance of mangroves could increase flood risk due to upstream water retention, Shih et al. (2015a, b) suggest removing 20% of the mangrove trees to allow water to flow through the mangroves as a tradeoff between reducing flood risk and carbon storage. Furthermore, Huang et al. suggested that managing the

Fig. 5.4 The Gaomei wetlands are located in the red polygons in Taichung City, Taiwan. (Illustrated by Yi-Te Chiang)

density of mangrove trees could improve habitat use by shorebirds. The meiofaunal community consists of nematodes, copepods, ostracods, polychaetes, oligochaetes, and gatrotriches, which are important food sources for migratory and endemic bird species. The taxonomic composition of the meiofaunal community makes it unique from other estuaries. The Danshui River wetland also provides a fishery resource consisting of fish, crabs, shrimp, and bivalves.

In addition to the natural wetlands in this watershed, several wetlands have been constructed, such as the Guandu wetlands (Fig. 5.4). A creek-pond combination wetland was added to the Shezi wetland to provide new habitat for shorebirds and fish (Shih et al. 2015a, b) while the Daniaopi wetland was built to enhance the efficacy of nitrogen transformation and removal pathways in the surface flow of domestic wastewater (Tan et al. 2017).

5.3.2 Gaomei Wetlands

The Gaomei wetlands are natural coastal wetlands that encompass roughly 734 hectares with a 3-kilometer-wide intertidal zone along the Taichung City coastline (Fig. 5.5). These wetlands provide habitat for the endangered *Bolboschoenus planiculmis* (Yulin sedge) and *Hygrophila pogonocalyx* (Pogonocalyx) and about 33 species of crustaceans, 155 species of birds, 13 species of fish, and 105 species of plants (https://www.saygaomei.com.tw/en/; Fig. 5.6).

Fig. 5.5 The tuberous bulrush *Bolboschoenus planiculmis* in the Gaomei wetland. (Photo by Wei-Ta Fang)

Fig. 5.6 The boardwalk in Gaomei Wetland. (Photo by Chia-Hsuan Hsu)

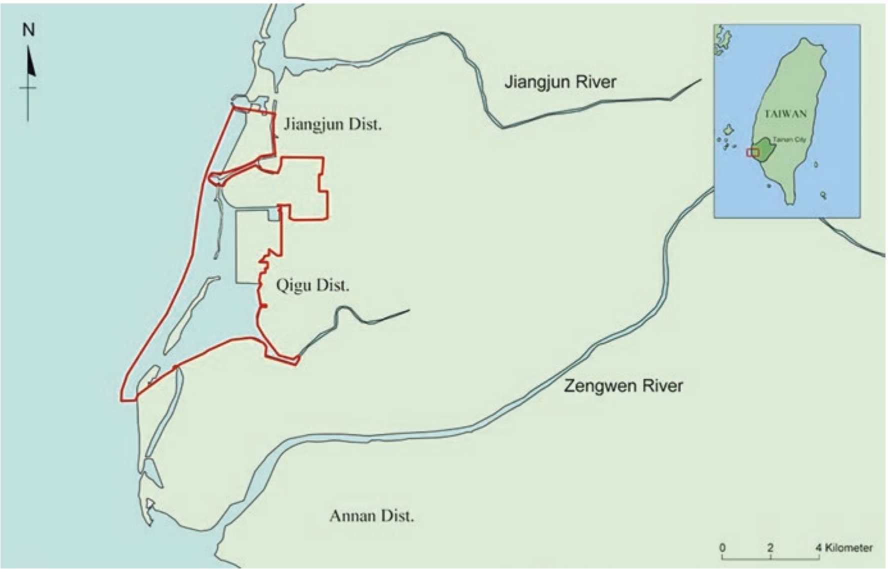

Fig. 5.7 The Cigu Salt Pan Wetland located near Tainan. (Illustrated by Yi-Te Chiang)

Gaomei Wetland is a popular attraction in Taiwan because of its stunning seascapes and sunsets and rich ecological resources. These wetlands provide good recreational and educational opportunities for the public (Hsu and Shen 2014). Thousands of people visit these wetlands annually; however, the increased tourist traffic is a worry because of the environmental problems, such as land crab roadkill (Tsai et al. 2017) and trampling of macroinfaunal habitat (Hsu et al. 2009). In an effort to minimize the environmental damage in these wetlands, the Taichung City Government built a 691-meter-long walkway in 2014 through the wetland (Fig. 5.7). Nonetheless, the number of people visiting this wetland remains high and strategies to mitigate the impacts of tourism still need to be better understood.

5.3.3 Cigu Salt Pan Wetland

Cigu Salt Pan wetland is located along Taiwan's southwestern coast near Tainan City (Fig. 5.8). The Cigu Salt Pan Wetland is under the authority of the Taijiang National Park. This wetland is 3697 hectares in size and supports a diverse array of terrestrial and aquatic habitats. About 50% of the world's Black-faced Spoonbill (*Platalea minor*) population, a critically endangered species on the International Union for Conservation of Nature Red list, uses these wetlands. By manipulating the salt pan and fishpond water depths, different habitats for the biota have been created (Wang et al. 2018b).

Fig. 5.8 Cigu Salt Pan Wetlands located in Taijiang National Park. The lower right photo, from left to right, Rob McInnes, Ben LePage, and Marinus Otte discussing park issues with a park ranger about salt-pan management in 2012. (Photo by Wei-Ta Fang)

Although the traditional way of making sun-dried salt stopped for economic reasons in 2002, the local people developed an ecotourism industry in and around Cigu Salt Pan Wetland (Fig. 5.8). Most local tourism industries were composed of small and medium-scale enterprises and used wetland resources from the local environment, especially seafood. Lee showed that community attachment and community involvement were critical factors for sustainable tourism development in these wetlands. Together with the stakeholders and a deep respect for the local and indigenous cultures, respect for their lifestyle, and compliance with destination guidelines, a sustainable wetland-based economy was achieved.

However, the Taiwanese Government has recently been promoting solar power generation in the Cigu Wetland to achieve Taiwan's non-nuclear policy, and the habitat of birds using these wetlands for food, breeding, and overwintering could be impacted. According to recent research, the local people would agree to small changes in the status quo and implementation of a fishery and solar power symbiosis (FSPS) policy. As a carbon sink, Liu et al. (2014) showed that the Chigu Wetlands can sequester 68,348.52 tons of carbon annually under current conditions. However, if the salt fields are developed with photovoltaic panels, the migratory birds will no longer use these wetlands, putting the ecotourism-based economy of the region at risk of being lost. The shore being covered with wave-absorbing blocks is another issue, and that, together with the shallow-water wetland fishing grounds being developed with photovoltaic panels, constitutes a contentious issue between green energy and bird conservation in southern Taiwan.

5.4 Conclusion

Taiwan is surrounded by the sea on all sides. This island nation is topographically varied and has a tropical to subtropical climate, which contributes to the diversity of wetlands. It can be said that it is an island surrounded by coastal wetlands. However, there are lakes, streams, ponds, paddy fields, and other types of freshwater wetlands in the inland areas. The effects of global change and population growth add new dimensions to environmental responsibility and sustainability. Groundwater depletion the world over poses a far greater threat to global water security than is currently acknowledged (Famiglietti 2014). In this chapter, we built on the Sponge City concept and focused on examples of water storage and treatment with possible solutions in a tropical and highly urbanized region. Our future is changing, and clean water and food will be driving strategies to adapt to these changing conditions.

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Chapter 6
Treatment of Urban Stormwater Through Constructed Wetlands – Experiences and Practical Guidance for Tropical and Non-tropical Settings

Swapan Paul and Max Finlayson

Abstract With rapid urbanisation across the globe, constructed wetlands are becoming integral components of urbanised landscapes. Their key purpose is to treat urban stormwater for the reduction of nutrient, sediment, and other pollutants before they escape further downstream. Aesthetics and biodiversity have become secondary objectives, but other environmental benefits are often overlooked. With climate change being an unavoidable phenomenon, there needs to be a holistic view to constructing wetlands. The chapter will highlight the above based on experience in the Sydney Olympic Park (Australia) wetlands and offer a comprehensive approach to achieving multiple outcomes at the same time sustaining the benefits from small wetlands. The experiences are applicable to other locations, especially where wetland loss has occurred and population or development pressures preclude the creation of large wetlands.

Keywords Constructed wetland · Sydney Olympic Park · Holistic approach

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6.1 Introduction

Constructed wetlands (CW) have been built primarily in urban settings to serve specific purposes; most notably to treat stormwater for pollution reduction and volume optimization to handle potential or actual flooding. In many cases, these wetlands did not exist in their present locations. Hence, they are neither remnant nor historical in nature; rather they are newly constructed in the landscape.

Wetland construction has likely been taking place over millennia and perhaps in many parts of the globe (Brix 1994) and as such, wetlands that have been built to serve the abovementioned purposes often serve aesthetic and biodiversity functions as well. As an example, the use of constructed wetlands for urban stormwater quality improvement is widely adopted in Australian cities, many of which have been successfully incorporated into the urban landscape (Wong and Breen 2004).

A constructed wetland is, therefore, a phenomenon that is largely specific to urban and peri-urban settings because, with the increasing rise in human densification in urban areas, there have been drastic modifications in the landforms and intensification of land use for housing, industry, business, infrastructure, and other services. Such developments have reduced the extent of water infiltration through the soil to almost zero across increasingly impervious areas. Such transformed landscapes generally discharge significantly higher quantities of pollutants than natural landscapes, and, at the same time, they have the potential to upset the natural water balance by primarily causing flooding. Therefore, to avoid and mitigate flooding, handle pollutants, and provide aesthetics and other services, constructed wetlands are increasingly being included as integral parts of such transformed urban and peri-urban landscapes.

There are many advantages of a functioning constructed wetland. These include high levels of nutrient and pollutant reduction through macrophytes and microorganisms as well as other assemblages. The same authors also highlight the limitations that CW may have, including the need for actual land in high-demand urban space, high construction and maintenance costs, the risk of poor aesthetics, and other inconveniences. This warrants the need for a comprehensive set of analyses, systematic steps, and suitable management protocols in both tropical and non-tropical locations.

CW as an effective and low-cost technology has been proven applicable in water pollution control in many countries, including China (Zhang et al. 2021). Given the relatively rapid growth in the CW sector, there have been correspondingly rapid developments in the approach, including in science, tools, technology, perception, and acceptance of CW. To meet the increasing demands, it is plausible that often things are fast-tracked, and wetlands are constructed without completing a full cycle of study of their functionality, efficiency, suitability, etc. This may leave unwanted room for a mismatch between need and operations, hence underperformances. In the worst cases, these wetland 'assets' can quickly turn into 'liabilities'.

Sydney Olympic Park is one of the early innovators and adopters in this field. This took place in connection with the organising of the 'Green Games' – the

Sydney 2000 Summer Olympics. This article elaborates on some aspects of the CW approaches and operations as well as development in the Park. To illustrate how CW perform their specific design objectives, examples, and lessons from over more than two decades are shared. These examples will, on one hand, highlight their efficiency and on the other, indicate the evolution of various CW design and management approaches over that length of time. While these wetlands are located outside the tropics in a humid maritime climate on the eastern coast of Australia, there are many lessons and approaches that can serve as examples for the use of CW elsewhere, especially in large cities in coastal environments where many of the natural wetlands have been lost and remnants only remain, or there is limited land available for the construction and operation of large wetlands. The experiences in the Sydney Olympic Park cover both governance and operational issues and serve as a practical model for elsewhere, including in tropical coastal environments. The practical basis of this experience is well shown in a workbook for managing urban wetlands that draws upon the knowledge and managerial expertise from the site but is presented as a resource for wetland managers more generally, with many of the individual contributors having experience in tropical locations (Paul 2013).

6.2 The Sydney Olympic Park CW Story

The Sydney Olympic Park Authority manages a large and diverse urban park in the geographical heart of Sydney, Australia (Fig. 6.1; S 33°50′56.09″, E 151°4′3.79″). The Park is a combination of 210 ha of business and sporting venues and 430 hectares of parklands, including fresh and estuarine wetlands, remnant terrestrial forest, and remediated industrial land and landfills. Water has been an important element in the design of Sydney Olympic Park, where potable water, reclaimed water, sewerage, and stormwater have been integrated into a water infrastructure network. Provisioning for biodiversity conservation function and aesthetics have also been embedded in the water story.

Freshwater wetlands were constructed mainly between 1998 and 2000 to receive, treat, and store stormwater as well as to provide habitat for flora and fauna, mainly for providing habitat for the endangered Green and Golden Bell Frog (*Litoria aurea*) (hereafter called, bell frog) (O'Meara and Darcovich 2008). A further goal was to attain environmental sustainability through clever management of resources and to provide an information base for other wetland managers.

From the 1920s to the 1970s much of Sydney's wetlands were removed through drainage and land reclamation projects to allow industrial action and dumping activities (Taylor and Hutchings 1996). As a result, approximately 70% of the extensive flats and saltmarsh around Homebush Bay (Saintilan and Williams 2000) and its associated freshwater wetlands were lost. Between the early 1980s and late 1990s, several initiatives were taken to remediate and reclaim the lands as part of an urban renewal project. The immediate result was remediation of 160 hectares of contaminated land and the recovery, consolidation, and on-site containment of excavated

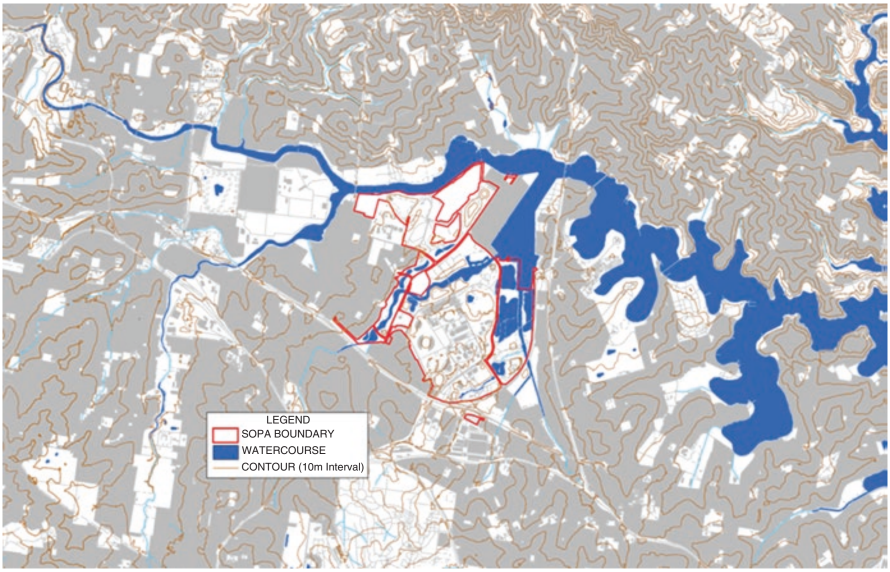

Fig. 6.1 Sydney Olympic Park in the greater catchment of the upper Parramatta River

waste (SOPA 2012). Ultimately, a new landscape was transformed with some protected remnant wetlands, new freshwater and estuarine wetlands, grasslands, woodlands, and saltmarsh landscapes (Paul and Farran 2009).

The latter is a key lesson from this case example; a new landscape was developed and transformed in response to local conditions, including those from the landscape, as well as those from the governance and planning mechanisms. They were further supported by an adaptive learning approach that enabled the general knowledge about CW to be applied and developed in response to these local conditions. In tropical settings versus the sub-tropical setting of the Sydney Olympic Park wetlands, differences in climate and landscapes will occur. Hence, as occurred in Sydney, these will guide what is achievable, and while doing so, local benefits will accrue and in turn help guide the development of the concepts and the outcomes.

6.2.1 Extent of the Wetlands at SOP

There are approximately 200 hectares of wetland areas in the 430 hectares of parklands within the 640 hectares total area of Sydney Olympic Park. Freshwater-constructed wetlands cover approximately 60 hectares. The remaining areas are supporting habitats for bell frog and estuarine wetlands containing mangrove, coastal saltmarsh, estuarine lagoon, estuarine stormwater creeks, and mudflat areas. Nearly 200 individual waterbodies are spread across three clusters of freshwater wetlands. These clusters are known as Narawang Wetlands cluster, Kronos Hill

Wetlands cluster, and the Brickpit Wetlands cluster. Each of these clusters consists of large stormwater storage ponds and smaller satellite habitat ponds built as habitat for the bell frog (Fig. 6.2). The catchment area of each wetland cluster is primarily local and restricted to the adjacent urban area, although Narawang Wetland also functions as a floodplain for the adjacent estuarine Haslam's Creek. The larger stormwater retention ponds provide for the multiple, often overlapping functions of stormwater collection, treatment, storage, and source of water for local irrigation (Fig. 6.2), habitat values, aesthetics, and biodiversity.

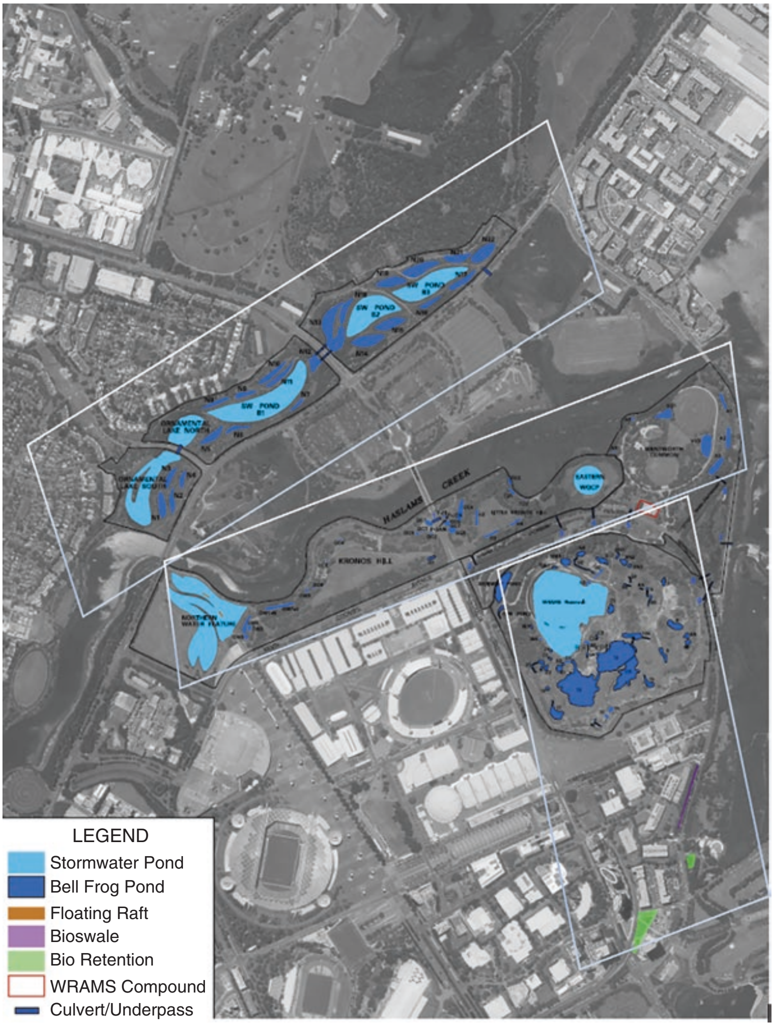

Fig. 6.2 Various types of constructed wetlands and related assembly in Sydney Olympic Park. (top rectangle: Narawang Wetlands Cluster, middle rectangle: Kronos Hill Wetlands Cluster; bottom rectangle: Brickpit Wetlands Cluster)

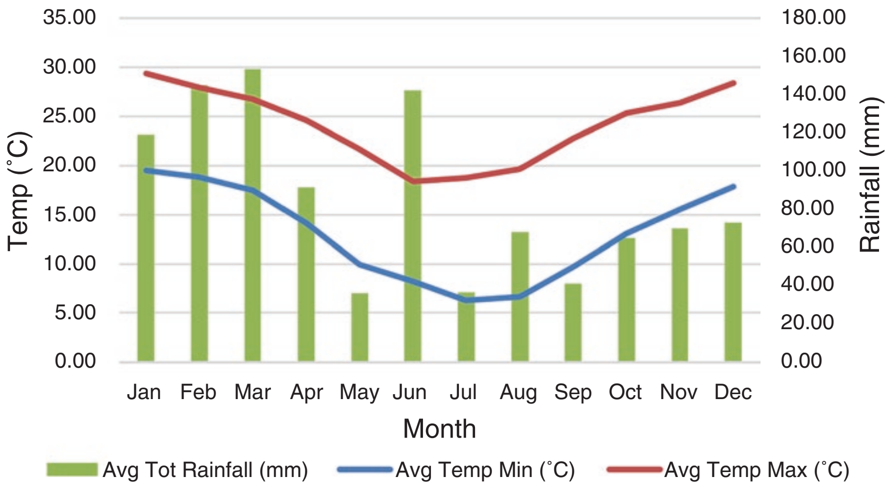

Fig. 6.3 Monthly average temperature and total rainfall in Sydney Olympic Park during 2012–2020

In terms of the overall geographic and environmental features of the locality, the monthly average temperature and rainfall situations categorise the area as a sub-tropical climatic zone during 2012–2020 (Fig. 6.3).

6.2.2 Evolution and Manifestation of Constructed Wetlands at Sydney Olympic Park

At a global scale, the manifestations of CW have been fast evolving. This evolution was most probably guided by the specific needs that such systems are desired to perform along with the change in landscapes. Hence, whilst a constructed wetland can be simply featured as a relatively shallow water body that is built to manage stormwater quantity and quality in urban situations, some are built to meet other specific functions for biodiversity, aesthetics, and social outcomes. Consequently, many distinct yet closely related manifestations have been developed and adopted.

Ghanem and Simpson (2008) have categorised CW as two basic types: Stormwater Constructed wetlands and Wastewater Treatment wetlands. To illustrate the CW types in Sydney Olympic Park, the former type is slightly adjusted; hence, Sydney Olympic Park has both categories of wetlands, however, there have been further advancements since the 2008 categorization. Fonder and Headley (2013) have classified CW and proposed six distinct standard types. In addition, there could be yet further purpose-built wetlands for aquaculture and agriculture, which are rare in urban landscapes but often seen in peri-urban situations; not that these are expected to be built in the Sydney Olympic Park precinct and hence they are not further addressed in this article.

Based on local experiences at Sydney Olympic Park, the CW are briefly outlined below, with a checklist of the distinguishing key characteristics (Table 6.1).

Table 6.1 Key distinguishing features of various CW types present in Sydney Olympic Park

Wetland type	Water Storage	Surface Water Flow	Sub-surface Water Flow	Sedimentation Basin	Macrophytes	Faunal Significance	Ephemerality	Flood Mitigation	Nutrient Removal	Sediment removal	Pollutant Removal	Educational Values	Aesthetics Aspects	Social
	Stormwater treatment wetlands													
Bioretention basin		√	√		√	√	√		√		√	√	√	√
Bioswale		√	√				√		√		√	√	√	√
Raingarden		√	√		√	√	√		√		√	√	√	√
Floodplain		√				√	√	√					√	
Sedimentation pond	√			√	√		√	√	√	√	√		√	
Stormwater pond	√			√	√	√		√	√	√	√	√	√	√
Habitat pond (GGBF)			√		√	√	√		√			√	√	
Floating raft					√	√			√	√	√	√	√	
	Wastewater treatment wetlands													
Leachate treatment pond		√			√		√		√		√	√		
Sewerage treatment					√		√		√		√	√		

A. **Stormwater Treatment Wetlands**

Bioretention basin – a bioretention device, which is generally a very shallow basin that provides a temporary stormwater retention function and also a stormwater corridor as well as a filtration feature so it can remove minimal sediment but maximal nutrients from the incoming stormwater runoff. These systems usually contain dense sedges and rushes to filter stormwater and the stormwater may travel both horizontally and vertically.

Bioswale – another bioretention device; a gently flat, linear landscape feature that acts as a stormwater corridor and a playing field for local people to play as it is maintained as a lawn. To avoid clogging of such systems, high-sediment water is not usually added to such wetlands and stormwater travels horizontally.

Raingarden – yet another type of bioretention type of feature but contains trees and shrubs as well as water-tolerant sedges where water travels mainly vertically but may also flow horizontally. It is worth mentioning here that these features are relatively new and have evolved over the past 2–3 decades and have been added to the Park progressively and mainly in the last 5 years. Figure 6.4 shows some examples of these wetland types.

Floodplain – when relatively flat land is transformed to convey the rainwater arriving from heavy storm events or water surge from rivers. It mostly serves as a corridor, and water recedes quickly (within a few days to weeks) by mainly surficial drainage, some infiltration and evaporation. Such areas may have seasonal water plants and occasional animals.

Sedimentation Basin (Pond) – a standalone waterbody or a component of an integrated constructed wetland to capture most of the sediment incoming from the catchment. These systems need the speed of the inflowing water to be reduced so that sediment can quickly settle at the bottom, which then gets removed every so often. The next compartment is usually a vegetated, shallow zone prior to the outlet.

Stormwater Pond – a stormwater detention pond that primarily serves a detention function and may or may not retain water throughout the year (perennial) – unlike a lake does. It has the functions mostly of a lake, but it is purposefully constructed for stormwater detention; hence has additional infrastructure requirements. It may have some overlapping functions, with sedimentation occurring at the bottom and macrophytes establishing along the edges.

Habitat Pond – a very shallow pond, mostly with dense macrophytes but with 20–40% surface area usually open for habitat functions; primarily for bell frogs. These receive water from their own local catchment but no runoff directly from road and other impervious surfaces to avoid pollution. When necessary, the ponds are topped-up with water from a Stormwater Pond and may be ephemeral, however, these may be flooded when heavy rainfalls overtop the low banks.

Floating Raft – these are yet another constructed wetlands that float on the water surface with the help of some floating devices but a mattress is planted with macrophytes. Within a year, the macrophytes establish and perform water quality enhancement and habitat functions. Such structures are particularly helpful where water level fluctuation is rapid and large.

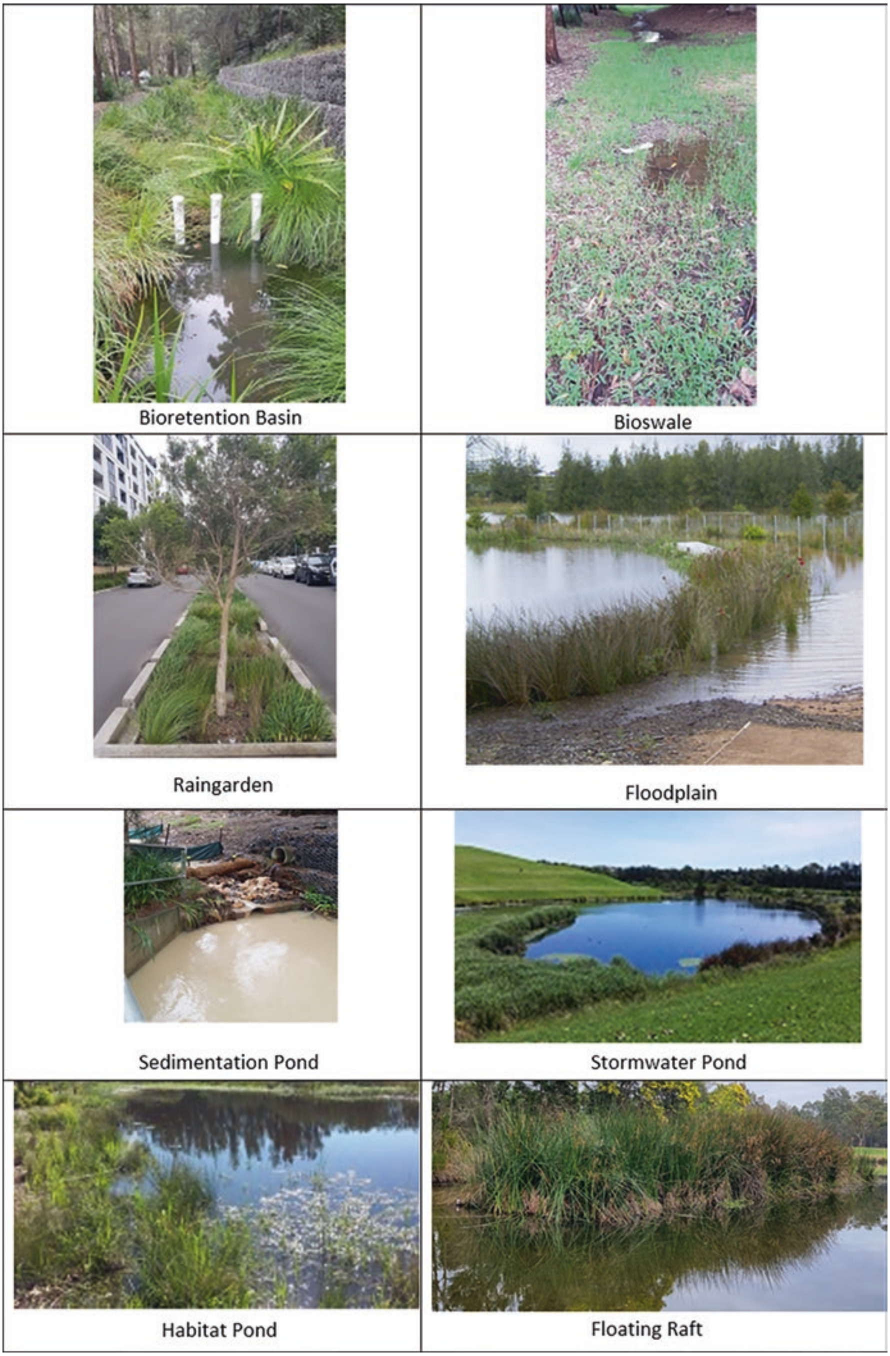

Fig. 6.4 Various constructed wetland types in Sydney Olympic Park (Photo: S Paul)

B. **Wastewater Treatment Wetlands**

Leachate Treatment Pond – a constructed feature, specifically constructed to treat leachate appearing from a structured landfill. It may or may not seep (leachate) year-round, but in most cases, specific macrophytes are planted to serve targeted pollutant removal functions. These could have surficial, sub-surface, and horizontal flows.

Sewerage Treatment – usually these receive sewerage that is diverted into a series of such ponds, and often with chemical doses. These ponds require frequent removal of sludge from the bottom, which is accumulated from the incoming sewer. However, in Sydney Olympic Park, there are no dedicated ponds for this; instead, the sewerage is injected directly into the treatment plant (see below for WRAMS).

Table 6.1 contains the key similarities or dissimilarities among various CW types that exist in Sydney Olympic Park.

6.3 Sydney Olympic Park CW Description

6.3.1 Narawang Wetlands Cluster

Narawang Wetland consists of a narrow, 1.6 km corridor of three stormwater ponds and an ornamental lake, and 22 habitat ponds (for Green and Golden Bell Frog; known as 'bell frog') – all completed in the year 2000. The stormwater ponds receive stormwater from the adjacent Newington residential suburb and provide irrigation water to parts of the suburb. The habitat ponds of Narawang Wetlands (Fig. 6.2; top rectangle) are clay-based and vary in shape, depth, and size, with the largest pond being 1332 m^2 and the smallest 338 m^2 (average pond size 852 m^2). (O'Meara and Jack 2012).

The three large stormwater ponds of Narawang Wetlands were not designed as habitat ponds for the bell frog but are used as sources of water for park irrigation and maintaining suitable water levels in habitat ponds. These stormwater ponds perform the treatment function that is needed for local irrigation and not particularly for discharging into the downstream waterways.

6.3.2 Kronos Hill Wetlands Cluster

During the construction phase, the Kronos Hill areas underwent large-scale remediation with waste containment under two large, clay-capped hills. Between 1993 and 2000, two large wetlands were constructed at either end of the cluster to perform primarily stormwater management and irrigation functions. These ponds are known as the Eastern Water Pond (EWP), built in 1997, and the Northern Water Feature (NWF) was built in 1999.

These two ponds, in conjunction with the Brickpit Reservoir (Fig. 6.2; middle rectangle), form the network of ponds that support the WRAMS (Water Reclamation and Management Scheme) program. These stormwater ponds are then linked to the water treatment and filtration plant, where Reverse Osmosis and other filtration functions are performed prior to distribution to the local network for use for non-drinking purposes. The WRAMS program treats and pumps more than 800,000 ML of stormwater every year. In addition, it has expanded its capacity by treating sewerage from the local network. A total of 29 habitat ponds were constructed to form a corridor (Fig. 6.2) so that the bell frog can travel between clusters.

In 2017 and 2019, Floating Macrophyte Rafts were added in the Northern Water Feature (NWF) as supplementary treatment wetlands to complement the NWF treatment function. The greatest advantage of these floating raft wetlands is that they float with the water level fluctuations and because of this, these are not impacted by the changing hydrological regime. That way, these wetlands can perform their water treatment and biodiversity harbouring services without having to depend on the macrophytes that could be impacted by the water level fluctuations.

6.3.3 The Brickpit Wetlands Cluster

The Brickpit was formerly a quarry for the NSW State Brickworks that ceased operation in 1992 and developed into a natural system of freshwater wetlands dispersed over three levels (terraces). The bottommost terrace was converted to a water reservoir and linked to the WRAMS network. A semi-permanent cluster of large ponds now forms the centre of breeding activity, with ephemeral and smaller ponds used intermittently by bell frogs (Fig. 6.2; bottom rectangle). After the successful colonisation of constructed habitat in Narawang Wetlands and Kronos Hill by the bell frog, replacement frog habitat consisting of 19 ponds, 10 soaks, 5000 tonnes of rock piles, and 4000 m^2 of grassland was placed in the Brickpit to form this constructed habitat.

As part of the residential development program, the former SWP was replaced with a bioretention basin in 2019/2020 (Fig. 6.2) and it has been performing its pollution reduction function since. This basin has three connected chambers, and each performs its pollution reduction function in combination. A pair of longitudinal bioretention basins have also been constructed alongside Bennelong Parkway to treat road runoff. These also perform quite well.

In addition to the CW described above, there are some other freshwater wetlands in other areas of the Park which do play some stormwater treatment roles but because those wetlands are not performing any key pollution reduction functions, they were not included here.

6.4 Nutrient, Sediment and Pollution Reduction Functions of the Sydney Olympic Park CW

Obviously, the above-described CW (Table 6.1) in the Park have been demonstrating various levels and degrees of efficiencies in terms of their abilities to remove sediment, reduce nutrients, and filter other pollutants. Table 6.2 below provides a summary of the treatment functions that three prominent Stormwater Ponds have performed from 2006 through 2009. Two of these ponds are still functioning, but the third pond, SWP (Southern Water Pond), was replaced with a Bioretention System in 2019/2020, which came as part of site development for a residential tower construction.

The percentage reduction shown in Table 6.2 compares well with the sort of reduction that CW are generally expected to achieve, where TSS is reduced up to 80%, TN 20–55% and TP 40–60% (MSH 2016). Nevertheless, it is worth mentioning that these CW were designed in the mid-1990s and constructed in the late 1990s. These have simplistic layouts and structures, with a primary focus on stormwater detention and water quality enhancement as a secondary feature. Therefore, they contained a stormwater detention compartment and a rather smaller proportion of macrophyte zone for stormwater filtration. Despite their relatively simplistic design features, these CW managed to remove considerable proportions of pollutants from the incoming stormwater. As opposed to these ponds cited in Table 6.2, the most recently designed and constructed Bioretention Systems have been performing significantly better (Table 6.3). The primary reason for this has been the improved filter medium, better growing medium for macrophytes, and suitable soil porosity for both vertical and horizontal flow (and treatment) ability.

Table 6.2 Extent of nutrient and pollution reduction during 2006 through 2009 by various constructed wetlands in Sydney Olympic Park

	NWF	EWP	SWP[a]
Suspended solids			
Max reduction %	97	88	80
Avg % reduction	53	56	55
Total N			
Max reduction %	85	90	94
Avg % reduction	64	59	69
Total P			
Max reduction %	89	76	78
Avg % reduction	63	53	51

[a]In 2019/2020, this Stormwater Pond was demolished by constructing a residential tower and a Bioretention System, which has been in operation since

Table 6.3 Extent of sediment, nutrient, and pollution reduction during 2020 by the Bioretention System operating in the Opal Tower precinct in Sydney Olympic Park

Date of observation	Statistic	Total P inlet (mg/l)	Total P outlet (mg/l)	% Reduction Total P	Total N inlet (mg/l)	Total N outlet (mg/l)	% Reduction total N	TSS inlet (mg/l)	TSS outlet (mg/l)	% Reduction TSS
07.2.2020	Mean	0.15	0.01	90.91	1.44	1.07	25.75	68.00	3.97	94.17
	SD[a]	0.09	0.01		0.50	0.47		23.52	3.55	
21.5.2020	Mean	0.22	0.07	67.16	1.49	0.58	61.16	32.33	1.90	94.12
	SD	0.01	0.01		0.08	0.14		2.52	0.17	
10.8.2020	Mean	0.32	0.05	85.57	2.40	1.10	54.17	99.00	7.57	92.36
	SD	0.02	0.01		0.26	0.35		17.04	0.49	

[a]SD – Standard Deviation (n = 5)

6.5 Management Approach

Wetland and stormwater management in Sydney Olympic Park are guided by many plans and policies, which are operated under an overarching set of legislation and other instruments. Among these, the most pertinent guiding document is the SOPA Stormwater Management and Water Sensitive Urban Design Policy (2012) and then the Guidelines published in 2016. These help in achieving the integrated water management outcomes in the precinct. This is further elaborated in the later part of this article. However, the water management approach is always achieved from an environmental sustainability standpoint.

6.5.1 Water Management

Effective coordination is required between the different stakeholders involved in the functions of the water and wetland management. Operating protocols identify appropriate water levels for the stormwater ponds depending on seasonality and changing water demands. The summer months require a higher operating level to balance increased water usage in frog ponds and less reliable rainfall, whereas during winter and early spring, water levels are maintained at lower levels to allow small-scale drying periods, ensuring macrophyte health and seed germination. Macrophytes health is critical for the water treatment and biodiversity objectives. Given the multiplicity of the overlapping (and often competing) objectives in water conservation, stormwater treatment, habitat functions and biodiversity, aesthetics, and other social imperatives; coordination is critical. This takes place quite effectively in Sydney Olympic Park.

6.5.2 Bell Frog Habitat Management

Bell frog habitat management activities aim to ensure quality habitat is available for the bell frog to breed and forage. Work is undertaken as per protocols contained in the Biodiversity Management Plan (BMP), which forms the basis of licencing under NSW State legislation and conditionally authorises specified activities carried out in accordance with the provisions of the Plan. Management activities include control of water levels, vegetation management of both wetland and terrestrial habitats, control of threatening processes, visitor management, and enhancement work in the Park.

6.5.3 Landscape Management

The primary and supplementary frog habitats require regular maintenance to retain their habitat values, control invading weeds (see Sect. 6.5.6), and maintain presentation standards. Many landscape activities can potentially involve 'harm' to the bell frog or its habitat and hence can only be conducted in accordance with a Licence and are regulated by the protocols. Core primary frog habitat is conserved with no net loss of habitat, and habitat connectivity between ponds is maintained and enhanced to facilitate bell frog movement.

6.5.4 Gross Pollutants Control

Gross pollutants are filtered out at the very outset. There is a network of underground GPTs (Gross Pollutant Traps) which separate and capture such pollutants prior to the stormwater entering CW. Where a GPT network is not present, traps are present in stormwater pits to capture gross pollutants. Moreover, the Sydney Olympic Park precinct as such does not generate much gross pollutants except leaf litter that arises from the landscape and street trees.

6.5.5 Sediment Control

Sedimentation in wetlands is a natural process, but due to heavy urbanisation, CW can receive excessive loads of sediment, silt, and leaf litter through stormwater. The functionality of stormwater ponds and some of the larger habitat ponds are now, 20 years after completion, threatened by increasing sedimentation. Dredging of some ponds has been necessary due to impacts on the infrastructure of the ponds. Connecting pipes and other services are becoming partially filled and the integrity of the irrigation system is heavily compromised.

In most cases, machinery access to remove sediment was not considered in the original design phase. Most CW are surrounded by bell frog habitat and sediment removal requires a staged clearing of vegetation. The original design criteria did not consider a full schedule of maintenance specifications, and with progressive learning, new protocols and procedures are adopted. It is, however, given that when a lot of sediment removal is needed, it warrants re-setting the macrophyte basin. This takes time to regenerate and be ready for the nutrient reduction role. Should pre-grown macrophyte beds be available, such replacement and renewed functionality could be expedited.

6.5.6 Weed Management

Weed management, especially aquatic weeds, has been a major challenge, which is often difficult to manage in sympathy with bell frog management, particularly when herbicide is not generally permitted. Weeds are unwanted, as they may dominate the desirable macrophytes. It is against the standard practices to overlook weed infestations. Weeds must be suppressed early in a proactive manner to prevent infestations becoming established. Unfortunately, at Sydney Olympic Park, adjacent lands are constant sources of weeds, and the wetlands are a recipient of water that transports weeds from upstream. Weeds of significance have been identified in Narawang Wetlands cluster, which include Alligator Weed (*Alternanthera philoxeroides).* Constant vigilance is required to monitor for new outbreaks of weeds. Strict hygiene practices assist in reducing the likelihood of introduction of weeds from outside sources.

6.5.7 Algae Management

Filamentous algae control is often required to maintain a minimum of 20% openwater area in habitat ponds, improve water quality, or make the wetlands more aesthetically appealing. The 20% openwater target helps waterbirds forage, bell frogs' free movement, and aesthetics. With the stormwater ponds ageing, blooms of filamentous algae have occurred on occasion. If required, algae removal must be sympathetic to the potential presence of bell frog, as tadpoles and juveniles may be removed with the harvested algae. Algae removal may also affect nesting activity of waterbirds, such as black swan (*Cygnus atratus*) and Eurasian Coot (*Fulica atra*). Whilst the design considerations of stormwater ponds such as Eastern Water Pond and the Northern Water Feature included water quality improvement, it appears that there could be further considerations given to filtering the water even before entering such ponds.

6.5.8 Managing Water Quality

Water quality management is a priority in the mix of biodiversity and sustainability objectives. This was achieved by adopting several water quality management strategies. These include intercepting gross pollutants, reducing sediment, and road runoff from the catchments, improving management of the land for reducing nutrient loads, etc. Consequently, there was not any noticeable bloom of filamentous or microscopic algae in any of the CW, though at times filamentous algae would have become excessive in habitat ponds. This has never been a serious issue. Also, the combination of shallow ponds and storage ponds helped achieving water quantity and quality goals. However, as seen in Table 6.2, the CW in the Park have been largely very effective in water quality enhancement.

6.6 Research Activities

Research in the constructed wetlands has primarily focussed on the bell frog, with other investigations targeting water quality, fish genetics, floating aquatic weed management, toxic blue-green algae management, and management of threatened aquatic plants. Research that benefits management of the wetlands is encouraged and, where possible, incorporated into the adaptive management strategy.

In 2008, a five-year research programme, 'Building sound ecological restoration strategies for endangered species' funded by the Australian Government ARC funds and the Sydney Olympic Park Authority, was entered into with the University of Newcastle and other industry partners. The aims of the programme are to understand the spatial and temporal dynamics of the population and to determine the impact of deterministic factors such as predation, succession, and disease.

6.7 Monitoring Activities

Sydney Olympic Park uses an adaptive management model where monitoring informs management of ecological responses to management activities. Water Quality monitoring has been ongoing since the beginning of the Park, but over time the intensity and extent of the monitoring have changed; some due to the project-bound demand and others due to the ongoing monitoring cost. However, management of the CW and particularly bell frog habitats requires knowledge of long-term trends to accommodate large-scale temporal and spatial impacts, as past and current management practices can have a long legacy effect. Pond health is measured by water quality, the extent and diversity of emergent macrophytes, submerged macrophytes, and general, faunal activity.

Monitoring of the stormwater ponds and the bell frog habitats aims to understand the stormwater dynamics. Water quality is monitored through water sampling and in-situ testing, whereas bell frog population status and waterbird population are assessed for each of the three clusters. Adjustments and modifications to actions aim to closely align landscape maintenance with CW health and bell frog conservation, responding to a feedback programme of ecological information, derived from research, monitoring, expert advice, and operational experience.

6.8 Awareness, Education, and Training

As a community-orientated initiative since 2002, the Authority has been sharing the knowledge and experience that it gathers in managing its urban wetlands for water treatment and biodiversity conservation purposes. This has been performed in various ways, including organising hands-on workshops for practitioners. This is achieved by sharing the theoretical knowledge as well as providing practical experience – both in the field and indoor. In addition, webinar sessions are organised where specific topics are discussed and newsletter and other forms of communications are arranged to share the knowledge.

6.9 Integrated Water Management and Development at Sydney Olympic Park

6.9.1 The Guidelines

Sydney Olympic Park enjoys an integrated water management approach, and it has been implemented since the Sydney 2000 Summer Olympics. The scheme is known as the Water Reclamation and Management Scheme (WRAMS). The Scheme has resulted in a reduction in potable water use by more than 50%. It treats approximately 800 ML/year of sewer, 700 ML/year of stormwater, and then 800 ML/year of recycled water is supplied to the customers (SOPA 2021). However, from time to time, there needed to be reviews and amendments to the approach, nonetheless, the intent remained very similar. At the core of this approach has been water conservation; achieved through water harvesting, treatment and reuse. The final treatment is undertaken through a reverse osmosis pathway, but the initial treatment is undertaken through CW of various sizes, shapes, and functional capacities (as stated in Tables 6.1 and 6.2). The integrated water management guidelines include:

- Apply best-practice design principles, innovative technology, water-sensitive urban design, and water demand management techniques to all new developments and building refurbishments, to facility upgrades, and to public domain works.
- Connect all new developments to Sydney Olympic Park's recycled water system (WRAMS), where available, for all approved uses of recycled water.
- Use recycled water or rainwater where potable water quality is not required.
- Maximise water use efficiency and long-term water savings in day-to-day activities.
- Comply with NSW Government Resource Efficiency Policy water efficiency standards for new and refurbished office buildings and new water-using appliances.

- Reduce the volume and manage the quality of stormwater discharged to creeks and wetlands from buildings, roads, carparks, and paving to protect the habitats of receiving waters and with consideration of environmental flow requirements.
- Manage water harvesting from constructed wetlands with consideration of aquatic habitat requirements.

The above guideline is implemented under SOPA's Water Quality and Water Quantity targets for Development Sites. As per SOPA's Stormwater Management and Water Sensitive Urban Design Guidelines (SOPA 2016), there are sets of water quality and water quantity guidelines and targets for development applications. These water quality targets are aimed for nutrient and pollution reduction prior to the water entering the receiving waters in the waterways. However, the constructed wetlands in the Park are not considered as receiving waters. Hence, a constructed wetland may end up receiving stormwater that does not necessarily meet the water quality targets, but these wetlands themselves are expected to further act as treatment systems. The water quality targets require that all development must, as a minimum, achieve 45% reduction in the mean annual load of Total Nitrogen, 65% reduction in the mean annual load of Total Phosphorus, 85% reduction in the mean annual load of Total Suspended Solids, 90% reduction in the mean annual load of hydrocarbons, and 95% reduction in the mean annual load of gross pollutants.

These targets are generally met and often exceeded.

6.9.2 The MUSIC Pathway

The Authority, to ensure water conservation is achieved to the maximum level possible, MUSIC (Model for Urban Stormwater Improvement Conceptualisation) software is always used (MUSIC 2022). This is compulsory for any land development proposals so that each sub-catchment is looked at closely to ensure water quantity and quality objectives are achieved. MUSIC clearly guides both of these objectives, which helps in formulating recommendations for on-ground measures,, such as construction of bioretention systems, sediment control measures, etc. This is an essential tool in the integrated water management practice at the Park. In fact, it is a legislative requirement in the State of New South Wales in Australia, where the Sydney Olympic Park belongs, that MUSIC model is run for any major development proposal that involves land excavation and alteration of the existing landscape.

6.9.3 The Management Approaches

The management approach puts people at the centre of its activities. People include Park patrons in the forms of daily park users as picnickers, walkers and joggers, spiritual, and aesthetics; education sector includes in excess of 25,000 school students learning about wetlands each year and professional development courses in wetland management; volunteers and researchers; business community; sporting spectators, and many more. Works are in progress to put people at the centre of activities, but the integrated water management programme has already made the Park a popular place where CW are a magnet.

One of the best ways that the Authority has achieved its environmental sustainability, including water conservation and saving while at the same time providing environmental water for its biodiversity mandate, was through clever use of water on the site. This included capture and storage of stormwater; treatment of water through wetlands; filtration of water through reverse osmosis and other means; and the reuse of the treated water for irrigation and outdoor household purpose. In doing so, the Authority's WRAMS has been acclaimed as Australia's first and most successful (large scale) water recycling system. This has helped to ensure the Authority's environmental sustainability objective in a big way and along with other sustainability goals, the Authority is a Green Star (SOPBA 2021) awardee.

Management of CW in the Park comes with many challenges. Often, public presence creates some unknown issues in terms of pollution, disturbance to biota, and/or interaction with CW in various ways. Generally, no public access across the wetland boundary is permissible, yet often the public are unknowingly interfering with wetlands. More works need to be undertaken in this area so that the members of the public are rather a positive force in CW management in the Park. Bigger challenges are yet to surface, which are likely to emanate from climate change. Unpredictable rainfalls, temperature variations, and other natural phenomenon will place unfamiliar pressures, as outlined in a vulnerability assessment (Finlayson and Spiers 2010; Paul 2008).

6.10 The Need for a Comprehensive and Holistic Approach in the CW Sector

In light of the above-cited examples in Sydney Olympic Park, it is realised that an integrated water management approach is possible. Some other places in Australia do have integrated water management schemes of various degrees of complexities and dimensions. The Gold Coast City (Australia) has a Water Strategy 2019–2024 (GCCC 2020), the Urban Water Strategy of Melbourne City (Australia) has its vision (CoM 2017) and some other large cities have also developed their own. Whilst these cities have remarkable water strategies and plans, yet it is apparent that a more holistic approach is needed to achieve the best environmental, economic, and social outcomes. The Sydney Olympic Park examples are integrated but perhaps come short of being a holistic approach.

A closer look at the history reveals that historical use of wetlands for water pollution control can be traced to the ancient Chinese and Egyptian civilizations (Brix 1994). With humans continuing to discharge raw sewage into the environment, wetlands had invariably been cleaning the wastewater. In the previous century, artificial wetlands started to be constructed for the purpose of treating different kinds of wastewater, and in the last few decades, constructed wetlands have been developed into fully engineered systems. The first scientific proof of constructed wetland for water treatment or purification was perhaps observed in 1953 when Dr. Käthe Seidel found that reed could remove both organic and inorganic pollutants in Germany, but the first constructed wetland was claimed to have been created and applied in an engineering sense in 1974 in Liebenburg Therese, Germany (Vymazal 2011). Ever since, constructed wetlands have become almost an integral feature of urban landscapes, with an increasing trend in expanding this also to peri-urban landscapes. To ensure that it is a sustainable practice, a holistic approach is needed so that financial efficiency, resource optimisation, material longevity and durability, environmental sustainability, futures-proof, and most importantly, people-centric wetland management can be reached. These are briefly outlined below:

- Much attention is needed in working out how macrophyte beds may be better designed and constructed. It could be economically and operationally sensible to develop pre-cast and pre-grown macrophytes established in strong trays and such trays replace the non-functional older trays as necessary. So, this will require a technological advancement and a commercially viable supply chain where modular series of trays will be ready for a bioretention system.
- The stormwater inlet and outlet structures, devices, and features have much room for improvement for gaining operational efficiencies and financial advantages.
- Sediment removal at the pre-wetland stage (pollutant traps or membrane filtration system) needs to be more practicable and efficient. Often these are installed willy-nilly without taking into consideration the maintenance and servicing convenience and costs.
- It is also necessary to undertake more research in biotechnology in the field of microbial efficiency and possible inoculation of a better-performing microbial colony in CW treatment train. Macrophyte functions can be enhanced, and stormwater treatment outcome can be improved in this way.
- Much research also needs to be undertaken in harnessing traditional wisdom and approaches for solving perennial and emerging challenges; many of which are yet to emerge from the climate change pressures. There is a need for a successful marriage between tradition and technology because these, in isolation, may not be helpful in resolving all our CW problems.
- Finally, the holistic approach will need to place people at the centre. This, instead of people being ‘users’ of CW to their advantage; they can perceive CW as part of the nature and part of themselves. To achieve this, firstly, instead of perceiving the presence of people near the wetlands or in the catchment as ‘problem’, there is a need for shifting the mindset and considering ‘people’ as the solution. This is particularly true in the case of CW, as these are urban focussed, and CW are primarily an urban feature. The faster people will be receptive of CW, the better it is and the quicker these systems will be able to realise their greatest potentials. The other aspect of people-centric design and features of CW is the ever-changing

demography in the existing and emerging cities and towns. These places are becoming more multicultural than ever before. What this requires is that CW design features should incorporate elements that satisfy various multicultural nostalgia without compromising the primary design intent of a CW. Place-making, as an emerging sector, itself has been paying more attention to this and CW are a great place to furthering this.

6.11 Conclusions

The clusters of freshwater-constructed wetlands in Sydney Olympic Park were designed to provide stormwater treatment and ecosystem functions (habitat, stormwater retention, nutrient recycling, irrigation, and water recycling) while supporting passive recreational opportunities and education programs. Since construction, their management has presented challenges that include meeting statutory obligations for the protection of biodiversity; increasing local pressures from residential, business, and visitational demands; managing a remediated site; increasing public events; managing pest and weeds; conflicting management objectives such as water treatment and recycling versus wetlands health; mosquito control versus public visitation; species conservation; and most importantly, public use versus carrying capacity of the Parklands. During the initial design and construction phases, not all of the uses and functions of the wetlands were clearly perceived or documented. A few services were not expected or articulated at the design and construction phases; nonetheless, over time, the wetlands have performed such additional services.

When the wetland clusters were designed at Sydney Olympic Park much of the knowledge was not available for the stormwater treatment and bell frog, and design was required to provide elasticity to bridge the gap. This has resulted in a small but inevitable weakening of potential and some failure in individual stormwater systems. However, with the ability to adapt to changing demands and increasing knowledge, particularly within a maturing CW sector, the wetlands have allowed for improved performance in the Park.

The potential effects of climate change on these constructed wetlands have not been contemplated at the design and construction phases. With the advancement of our understanding of these two phenomena, many structural and functional adjustments are inevitable.

The CW sector requires more research and a way to develop a holistic approach to managing stormwater by constructing wetlands. A people-centric approach is what is likely to last long and meet the test of time. The latter is seen as a critical step, and if adopted, it should enable the CW sectors in other locations to develop facilities that suit their interests and local environments. The Sydney Olympic Park example can be used as a guide for other cities, with lessons coming from the experience and accumulated knowledge, especially in situations where local knowledge may be lacking, at least initially. A key element of any attempts to integrate CW into urban landscapes is to plan within the context of local conditions, including the social-cultural settings, and to understand the local nuances of the ecology and landscape functioning and use these to advantage. These are important lessons for cities in tropical locations – the example provided here is a guide, not a recipe for other locations. The local issues can be addressed in an adaptive manner, with monitoring and planning approaches used to ensure the setting supports the best use of CW for local purposes.

CW can be applied in tropical and non-tropical settings alike, with local conditions determining what is likely to be successful. Examples such as those in Sydney Olympic Park, a sub-tropical and coastal setting, are based on particular experiences but, in a general sense, provide practical guidance for other settings, particularly when the benefits of local planning and monitoring are incorporated in an adaptive manner. A further benefit from this example is the realisation that people can be at the centre of a complex urban setting with CW – it is not a matter of keeping people out, it is a matter of working with people and understanding their values and aspirations within the context of a complex CW project.

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Chapter 7
Phytoremediation of Agricultural Pollutants in the Tropics

Megan L. Murray and Brad R. Murray

Abstract Agricultural pollutants known to have harmful impacts on aquatic species and ecosystems include excess levels of plant nutrients (e.g., ammonium nitrate and phosphate from fertilizers) as well as inorganic (e.g., heavy metals) and organic compounds (e.g., pesticides including insecticides and herbicides) commonly associated with global farming practices. This chapter examines the role of phytoremediation in decontaminating these key pollutants of agricultural origins, with a particular focus on the plant species and environmental dynamics which occur in tropical regions. This chapter also includes strategic applications (e.g., terrestrial barrier plantings around sensitive wetlands), which could provide safe, affordable, and environmentally sustainable solutions for reducing the impacts of agricultural practices on tropical wetlands.

Keywords Farms · Agricultural wastewaters · Wetlands · Decontamination · Barrier plantings · Soils

7.1 Introduction

Agriculture is the practice of farming plants and animals to produce the essential resources required by global communities. Agriculture spans edible crops, timber and fiber, and a range of animals for a variety of products. In modern agricultural practice, crop species are generally grown in uniform monoculture fields or climate-controlled greenhouses supported by highly cultivated soils and water delivered by irrigation systems and occasionally rainfall (Hoffman et al. 1995). Stock animals are conventionally raised in rangelands, typically with barn structures for shelter

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and feedstock and water continually supplied for animals to access. Agricultural productivity varies depending on region and demand, but many crops and animals are widely farmed and have comparable resource requirements, including cleared land and water supply (FAO 2021). Collectively, these industries contributed $3.5 trillion to the global economy in 2018 alone, with farm productivity continually increasing in alignment with human population growth and increasing consumer demand (FAO 2021).

Agricultural practices radically transform natural landscapes through land clearing (i.e., deforestation and shrubland clearing), soil modifications, and altered water regimes (Fig. 7.1). Agriculture is also deemed responsible for ~22% of anthropogenic greenhouse gas emissions and it is acknowledged as a major contributor to global warming (Grosso and Cavigelli 2012). Wastes are continually generated by farmland operations. Waste originating from agricultural activities encompasses diverse pollutant mixtures with varying environmental risks, including animal-origin effluent from farms (e.g., ammonia in soils and as vapor release), plant harvest by-products, nutrient run-off from fertilizer applications, pesticides (including insecticides and herbicides), as well as a wide range of heavy metals and salts (Alengebawy et al. 2021). The composition and concentrations of these chemicals

Fig. 7.1 Aerial image of agricultural plots lining a waterway demonstrating the impacts of land-use change for agriculture (Fisk 2014)

relate to the type and scale of the agricultural practice in operation, but each pollutant creates its own challenges for land and water integrity. In some instances, like plant by-products and manures, wastes may be diverted and recycled for useful purposes (Yang et al. 2021); however, certain pollutants are hazardous in nature and contribute to land degradation. Farmers in developing nations often have little choice but to use polluted landscapes and risk food contamination, given the climatic, spatial, and socio-economic limitations to landscapes where key food crops can be produced (Xiao et al. 2017). Sustainable agriculture, which is focused on long-term crop and livestock production with minimal impacts on the environment, is therefore an immediate global priority in order to ensure a balance between resource production and the preservation of the environment (Hejna et al. 2021).

Developing new methods for remediating environments impacted by agricultural pollutants, including ecologically sensitive wetlands, is an important step in transforming agricultural industries to become safer and more sustainable. This aligns with a growing ambition to improve land management methods and prevent global pollution in general, seeking technologies with higher efficiency, lower costs, and safer implementation which can be tailored to industries and ecosystems of importance, including farmlands of tropical regions (Paz-Ferreiro et al. 2014). Phytoremediation presents an important opportunity for the passive decontamination and management of such pollutants, preventing their spread to vulnerable ecosystems and species. Phytoremediation is a phytotechnology used to clean up contaminated surface waters, soils, air, and groundwater. It is a cost-efficient (Mosa et al. 2016), non-invasive (Dietz and Schnoor 2001), longer-term biotechnology that can be applied in situ to decontaminate sites where contaminants are within reach of plant roots. To date, there has been substantial focus on addressing environmental problems associated with industrial activities and mining operations using applied phytoremediation (Peco et al. 2021), but considerably less attention has been given to the potential for phytoremediation to ameliorate the impacts of the vast range of ubiquitous pollutants associated with agricultural practice as well as protect pollutant-sensitive components of ecosystems.

This chapter examines research progress in the phytoremediation of globally common agricultural pollutants of water and soils, including fertilizers, ammonia discharge, and heavy metals, as well as select insecticides and herbicides. The implementation challenges for phytoremediation associated with decontaminating these pollutants and protecting wetland ecosystems are explored within the context of tropical regions, as well as future opportunities for applied research.

7.2 Fertilizer Pollution

Runoff from agricultural developments often carries excess nutrients from plant fertilizers that are not sufficiently removed by existing control measures (Fig. 7.2). Most elemental nutrients are essential for plant growth, which is the basis for applying supplemental fertilizer to crops, including nitrogen (N), phosphorus (P),

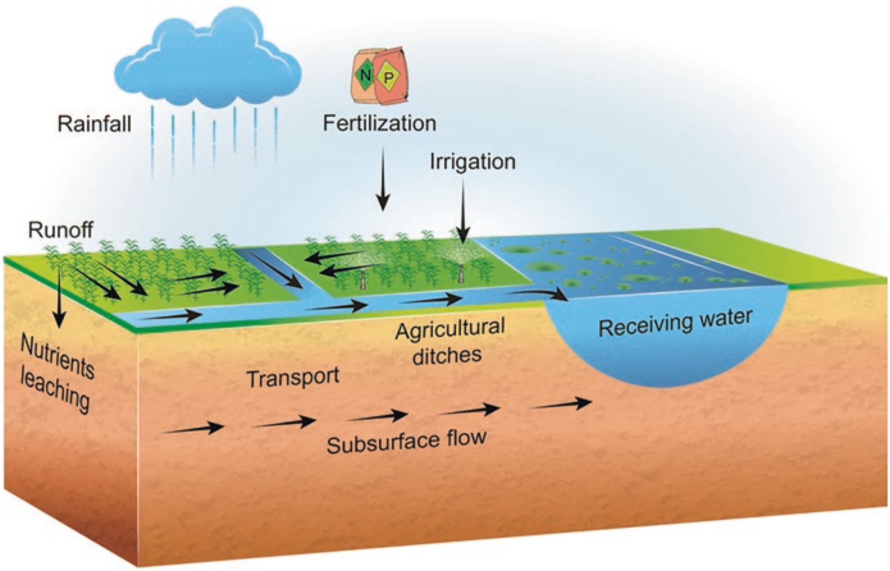

Fig. 7.2 Terrestrial agricultural pollutants and flows into aquatic systems (Xia et al. 2020)

andpotassium (K). However, in excess concentrations, these elements can cause significant stress, degradation, and impairment of ecosystem functions, including eutrophication and catastrophic species declines in freshwater and wetland ecosystems (GES 1997).

Nutrients originating from plant fertilizers have adverse effects on aquatic communities, often acting as a catalyst for eutrophication, triggering rapid growth in aquatic vascular plant and algal biomass and associated declines in water health (Sims et al. 1998). Therefore, although not toxic in trace amounts like certain heavy metals, phytoremediation of excess nutrients, including P, has enormous potential to protect ecologically sensitive areas, including natural water bodies and aquatic communities, from environmental harm.

7.3 Phosphorus

Over the last 50 years, P levels in soils disturbed or modified by human activities have been rising (Coale 2000). Highly elevated soil P levels (i.e., typically defined as 45 mg/kg^{-1}) are frequently recorded in areas of the world where intense animal farming activity occurs, as well as in areas where fertilizers have been applied as supplements for plant crops (Coale 2000). When soil P exceeds the capacity of the substrate to bind P, surface runoff or P transfer to freshwater systems, including wetlands, becomes an environmental concern. The presence of dissolved P, particulate P, and organic P in water may contribute to the eutrophication of rivers and lakes (Sims et al. 1998). Soils with excessive P levels have been identified as an important source of diffuse pollution (Sharpley et al. 1994). Therefore, finding and applying phytoremediator species capable of continually absorbing high levels of P into plant tissues and minimizing the impacts of P pollution is a key objective for protecting sensitive aquatic communities.

In a study by Delorme et al. (2000), phytoremediation of phosphorus-enriched farm soils was trialed using 12 common crop species and grass species, each previously shown to be successful in accumulating heavy metals in non-agricultural contexts. A dual study comprising greenhouse trials and field applications was performed, incorporating farm soils which were artificially enriched with P derived from inorganic fertilizers and manures. While all phytoremediation species were shown to actively remove P from the soil in the two contexts, the P content varied greatly across species. No species demonstrated foliar hyperaccumulation properties of P, but seed-sown corn (*Zea mays*) and Indian mustard (*Brassica juncea*) showed high P removal rates within their root tissues, up to 114 and 108 kg ha^{-1}, respectively (Delorme et al. 2000), which may make them strong candidates for terrestrial buffer plantings on downslope edges of farm sites, thereby preventing excess phosphorus from running off farm sites and entering nearby waterways. A further consideration for these two species is their potential to be further re-used as an animal food supplement or composted into a green fertilizer, given that P enrichment is not noted as harmful to farm animals, unlike heavy metals. One limitation to P being accumulated in below-ground tissues of these two species is that whole-of-plant harvests would therefore be necessary to remove the P-concentrated roots, and fibrous roots typical of these species could lead to the incomplete removal of plants and some P returning to the soil.

7.4 Ammonia

Ammonia nitrogen is a common toxicant derived from animal wastes as well as supplemental fertilizers. Ammonia nitrogen encompasses both the ionized form (i.e., ammonium NH_4^+) and the unionized form (i.e., ammonia NH_3). An increase in environmental pH favors formation of the more environmentally harmful unionized form (NH_3), while pH decreases favor the ionized (NH_4^+) form. NH_3 from poultry production is a major environmental concern for environmental pollution (Fig. 7.3). When birds consume protein, they produce uric acid, which is ultimately converted to NH_3 (Naseem and King 2018). Factors that increase ammonia outputs from poultry farms include soil pH, local climate, litter type, bird age, manure age, and barn ventilation (Naseem and King 2018).

Like P, NH_3 pollution is a common cause of the environmental degradation of wetlands and other water bodies and is attributed to fish kill events (Milne et al. 2000). However, the most common problems associated with NH_3 enrichment often relate to elevated concentrations negatively impacting fish growth and gill condition rather than rapid, mass mortality events (Milne et al. 2000).

Fig. 7.3 Poultry farm operations are a significant source of ammonia for adjacent water systems (Jordan 2011)

Table 7.1 Agricultural-origin NH_3 tolerance in aquatic macrophyte species

Species	Tolerance (mg/L)	Observations	Reference
Eichhornia crassipes	56–136	Survived 4 weeks, degradation of plant health noted in raw sewage (i.e., but not dairy manure)	Ayade (1998)
Hydrocotyle umbellata	136	Wilted on seventh day of test period in dairy manure	Sooknah and Wilkie (2004)
Lemna minor	7	50% growth inhibition reported	Wang (1991)
Pistia stratiotes	136	Wilted on seventh day of test period in dairy manure	Sooknah and Wilkie (2004)

There has been focused research on aquatic plant tolerance responses to NH_3 exposure (i.e., generated by agricultural wastes) which may reveal certain species as ideal candidates for within-wetland phytoremediation applications (Table 7.1). Vascular plants absorb three forms of nitrogen, namely, nitrate ions, urea, and ammonium ions (Kinidi and Salleh 2017). Once NH_3 is absorbed, it is broken down into chemical constituents and incorporated into proteins and other organic combinations through biochemical reactions. However, only the ammonium ions are assimilated into the organic molecules in the plant tissues by means of enzymatic processes (Masclaux-Daubresse et al. 2010). In the previous tolerance studies, plant health and survival were observed and recorded (Table 7.1), but not specifically NH_3 decontamination.

Given that ammonium and nitrate ions are principal sources of nitrogen, which support plant growth, phytoremediation and plant-based technologies are ideal solutions for such agricultural wastes. In a recent 2020 study, water quality improvements including NH_3, total suspended solids (TSS), and chemical oxygen demand (COD) were measured in a before-after experiment of aquatic macrophytes grown in wastewater (Abdul Aziz et al. 2020). It was found that *Lemna minor*, *Salvinia minima*, *Ipomoea aquatica,* and *Centella asiatica* were each able to reduce NH_3 by 80.4%, 89.9%, 97.3%, and 79.1%, respectively; TSS by 50.8%, 77.6%, 85.6%, and 67.6%, respectively; and COD by 75%, 82%, 44.8%, and 36.46%, respectively (Abdul Aziz et al. 2020). The *Ipomoea* species showed the strongest phytoremediation potential for NH_3 decontamination, while the *Salvinia* species was more effective at reducing TSS and COD. This demonstrates that mixed-species macrophyte plantings may provide a good "all around" solution for remediating wetland ecosystems which have water quality issues beyond increased NH_3.

Similarly, in a simulated microcosm study using polluted water sourced from Estero de San Miguel in the Republic of the Philippines, both NH_3 and P decontamination were investigated in a multi-factorial phytoremediation experiment featuring macrophytes transplanted into agricultural wastewaters (Acero 2019). Result revealed that a monoculture plantings of *Azolla pinnata* significantly lowered the NH_3 concentrations in the wastewaters over a 14-day period. Mixed species plantings of both *A. pinnata* and *Eichhornia crassipes* significantly lowered the P level of the wastewaters in the same time frame. Thus, both aquatic macrophyte species were fast acting in reducing both target pollutants and identified as potential phytoremediation options for aquatic environments in this tropical region (Acero 2019). While these results are promising, the tendency for both aquatic species to become invasive and form monocultures in wetland environments is worthy of careful consideration for environmental managers. Rezania et al. (2015) acknowledged this risk and proposed a range of controls that could be used in combination to manage *E. crassipes* in-situ as part of an integrated aquatic phytoremediation strategy, including combination of herbicides, integrated biological controls, and, ideally, watershed management to control nutrient supply (and therefore, restrict plant growth) although some of these environmental controls are more viable at large-scale than others, and the addition of chemicals including herbicides may significantly harm non-target species in sensitive wetland regions. Likewise, each of these control strategies could be evaluated for *A. pinnata,* as well as other aquatic species likely to become overabundant. Introducing any potentially invasive species into sensitive aquatic ecosystems is worthy of deep risk for a wetland system. A rigorous ongoing monitoring should be used.

7.5 Heavy Metals

A wide range of soils used for agricultural activities has been found to be contaminated with certain heavy metals, including Cadmium (Cd), Chromium (Cr), and Lead (Pb) (Nanda Kumar et al. 1995). In France, about 1% of 11,400 agricultural soil samples taken from across the country exceeded the national safe exposure limits for Pb (i.e., 100 mg kg^{-1}) (Mench and Baize 2004). Agricultural contamination of heavy metals, particularly those which bioaccumulate and impact food chains, is a critical risk to food security as well as ecosystem and community safety (Nanda Kumar et al. 1995). The investigation of heavy metal decontamination within the field of phytoremediation has received strong attention, particularly in other contexts, including mining activities and pollution from industrial processes. As of 2020, more than 450 different plant species from at least 45 angiosperm families had been identified as heavy metal hyperaccumulators (Suman et al. 2018). The aquatic macrophyte species *E. crassipes* has been examined in more than ten such phytoremediation studies of heavy-metal polluted water systems, demonstrating strong capacity to extract Cr (i.e., 65% removal) and Cu (i.e., 61–97% removal, depending on initial concentrations) from synthetic wastewaters and simulated wetland environments (Lissy and Madhu 2011; Mokhtar et al. 2011). In one study, which focused on decontaminating heavy metals from agricultural activities, *Zea mays* plantings were shown to be useful in accumulating both Cr and Pb from soils (Braud et al. 2009). Furthermore, bioaugmentation of siderophore-producing bacteria was shown to increase Cr and Pb accumulation in the plants by a factor of 5.4 and 3.8, respectively (Braud et al. 2009). A second field-based study conducted in Zhangshi, China, evaluated the phytoremediation efficiency of *Beta vulgaris* var. *cicla* in agricultural wastewater during a 2-month growing season (Song et al. 2012). These plants were directly exposed to agricultural wastewaters, which had elevated concentrations of Cd. The cultivar was found to accumulate 144.6 mg/ha of Cd over the course of the study. Amending the soil with supplemental organic manure was found to promote biomass increases of the plants, but inhibited Cd phytoremediation efficiency (Song et al. 2012).

These findings suggest that soil amendments aimed at increasing heavy metal uptake into plants should be reviewed on a species-specific basis. A deeper understanding of individual phytoremediator species and their potential interactions with the biotic and abiotic features of sites (e.g., including soil nutrients and rhizosphere activity) is important for achieving optimal outcomes for decontamination.

7.6 Insecticides

Insecticides are broadly defined as chemicals used to protect farmed plants and animals by killing insect species, preventing their reproduction, or deterring herbivory (Fig. 7.4). Insecticides are classified based on their chemical structures and

Fig. 7.4 A farmer manually applying pesticide to an open field (Balazs 2022)

modes of action and ecological research has examined their potential for unintended harm on non-target species, as well as residence time and degradation patterns in soils and water (Hedlund et al. 2019). Many insecticides are designed to act upon insect nervous systems (e.g., cholinesterase inhibition), while others act as growth regulators or endotoxins.

Systemic neonicotinoids are a sub-group of insecticides used to protect a wide variety of crop species. Based on their efficacy to control many insect pests and their systemic activity, they are used extensively in agriculture, so much so that by 2008, neonicotinoids accounted for one quarter of the global insecticide market (Jeschke et al. 2011) and this rate is increasing (Simon-Delso et al. 2014). However, increasing evidence indicates that this large-scale use results in high broad-spectrum insecticidal activity of the neonicotinoids even at very low dosages, and this has led to serious risk of environmental impact (Henry et al. 2012; Goulson 2013). Soil erosion from high-intensity agriculture facilitates the transport of insecticides into waterbodies (Kreuger et al. 1999). Some insecticides are accumulated by aquatic organisms and transferred to their predators, and insecticides by design are lethal to insects, so they pose a particular risk to aquatic insects, but they also affect other aquatic organisms (Goulson 2013). Accordingly, a recent study was designed to assess the neonicotinoid phytoremediation abilities of plant species commonly used in constructed wetland systems: *Acorus calamus*, *Typha orientalis*, *Arundo donax*, *Thalia dealbata*, *Canna indica*, *Iris pseudacorus*, *Cyperus alternifolius*, *Cyperus papyrus*, and *Juncus effusus* (Liu et al. 2021). Compared with the other neonicotinoids in the study, imidacloprid, thiacloprid, and acetamiprid were most readily removed by all plant species. Of note, *C. alternifolius* and *C. papyrus* exhibited the best phytoremediation performance for all six neonicotinoid types; the main phytoremediation mechanisms identified were plant accumulation and biodegradation (Liu et al. 2021).

Alternatives to neonicotinoids include organochlorides and pyrethroid-based insecticides. Pyrethroids can be very toxic to non-target aquatic organisms (i.e., arthropods are particularly sensitive) (Van Wijngaarden et al. 2005; Maund 2009), while several organochlorides are used extensively in agriculture, historically as well as presently (e.g., endosulfan and DDT), have been shown to accumulate in fish species (Darko et al. 2008) and are associated with biomagnification and harm to non-target species, including apex predators (Carson 1962). Using a water-based system, Riaz et al. (2017) evaluated the phytoremediation potential of macrophyte species *Eichhornia crassipes, Pistia stratiotes,* and a mixed algae species treatment (*Chaetomorpha sutoria*, *Sirogonium sticticum,* and *Zygnema sp*.) for removing organochlorine and pyrethroid residues from water. During the experiment, *P. stratiotes, E. crassipes* and all algae species showed insecticide removal efficiency, with 62%, 60%, and 58%, respectively for organochlorines, and 76%, 68%, and 70%, respectively for pyrethroids, with consistently higher concentrations of both pesticides detected in root tissues of the macrophyte species (Riaz et al. 2017). These results indicate aquatic systems for removing insecticides are worth consideration, particularly if farms are near natural water bodies. Insecticides are applied in various formulations and delivery systems (e.g., sprays (Fig. 7.4), slow-release diffusion) that influence their transport and chemical transformation after release. Mobilization of insecticides from farmlands into other ecosystems in the nearby vicinity can occur via runoff (i.e., dissolved or sorbed to soils), atmospheric deposition, or sub-surface flows (Goring and Hamaker 1972; Moore and Ramamoorthy 1984). Considering these scenarios, installations of aquatic macrophyte phytoremediators (e.g., as in-situ floating wetlands or tiered shoreline plantings between the pollutant sources and open waters) may provide low-cost protection from insecticide run-off for rivers, lakes, and wetlands alike.

7.7 Considerations for Tropical Regions

The tropics host one-third of the world's soils, which in turn support more than three-quarters of the world's population (Hartemink 2004; Kummu and Varis 2011). Tropical soils are influenced by highly variable weather patterns, with a predominance of high temperatures and abundant rainfall resulting in the effects of material weathering being more prominent than in other global regions. For example, Cuba tends to contain extensively weathered tropical soils, with 69.6% of soils exhibiting low organic matter and 43.3% with heavy erosion (Olivera Viciedo et al. 2018). As erosion is considerable in the tropics, the inherent deficiencies of weathered soil mean that for agricultural practices, supplementary fertilizers and nutrient enrichment will be necessary to support most food and textile crops in the future.

Considering this, more research on phytoremediation installations designed for P, NH_3, and other nutrient-enriching pollutants is merited, particularly vegetative buffer installations which can be designed to protect waterways and aquatic ecosystems that are sensitive to these chemicals. Further, insecticides are also noted to move off-target due to many factors, including improper application or unpredictable rainfall events, resulting in contamination of areas in the vicinity of agricultural practice and causing adverse effects on inhabiting species (Bish et al. 2020). Through better site management practices, including the implementation of protective vegetative buffer strips, off-target movement of pesticides and other agricultural pollutants can be decreased, while compound degradation and pollutant uptake can be increased via phytoremediation (McKnight et al. 2021). The main types of tropical soils, particularly oxisols and ultisols, differ from most temperate soils in terms of having low organic matter content, low pH values, and high levels of Fe oxides (Guerra Sierra et al. 2021). These soils are found in most of the tropical areas of Africa, the Asia-Pacific region, and Central and South America, coinciding with areas of high agricultural activity (Guerra Sierra et al. 2021; Caritat and Cooper 2011). Rather than a disadvantage, acidic soils can increase the mobility and bioavailability of certain inorganic contaminants (e.g., heavy metals), similar to chelating agents, which may promote the uptake of these pollutants more readily into phytoremediator species (Chen and Huang 2003).

Anthropogenic activities such as deforestation and habitat fragmentation affect the quality of various types of tropical soil, with the biodiversity of tropical forests severely impacted in the last century (Sala et al. 2000; Guerra Sierra et al. 2021). A major benefit of phytoremediation compared to other decontaminating technologies is that it increases site biodiversity both directly, via plant abundance and diversity increases (i.e., the latter for mixed-species plantings), and indirectly, by supporting pollinators and other insect species that use the plants for habitat (Garbisu et al. 2020). Bioprospecting for new, locally endemic tropical phytoremediator species, as opposed to relying on common crop species like *Zea mays* and *Beta vulgaris,* is an important goal for the field of phytoremediation (Prasad and De Oliveira Freitas 2003). Such species are pre-adapted to local soils and climate and are more likely to establish and self-sustain in these dynamic environments, as well as support and protect the biodiversity of tropical ecosystems.

Tropical wetlands also support critical ecosystem services for the planet, including carbon accumulation and storage (Donato et al. 2011), thereby providing resilience against accelerated global warming. These ecosystems are also some of the most biodiverse wetlands in the world for both floral and faunal species (Junk et al. 2006). Mangrove systems in particular are important rearing areas for fish and shellfish species and are responsible for 48% to near 100% of their population reproduction (Rönnbäck 1999). Ensuring these systems are protected from agricultural contaminants, as well as actively remediated when contamination is found to be present, provides more certainty that these critical services can be maintained for future generations. Beyond phytoremediation, there has been much progress in the field of constructed wetland systems for filtrating a wide range of pollutants. A review by Wang et al. (2017) demonstrated that properly designed constructed wetland systems can be a cost-effective strategy for removing a range of agricultural pollutants, including NH_4^+, total N, and total P, but a definitively "one-for-all design" does not exist. The performance of these systems varies with seasonal conditions as well as local operational parameters, including water flow rates. In addition to plant species selection, the optimization of other design criteria, such as pre-treatments, recirculation, forced aeration, and in-series landscape design, can substantially enhance contaminant removal, providing a sustainable and low-energy approach to agricultural wastewater management compared to traditional wastewater treatment facilities (Wang et al. 2017). Applying these engineering principles to planned phytoremediation installations would provide enhanced opportunities for effective aquatic decontamination, which could function in tandem with terrestrial barrier installations, to protect sensitive water systems from the ongoing agricultural processes which generate pollutants.

A further dimension for consideration of phytoremediation is that several secondary products have been proposed as ways to divert the plant material from general landfill and further increase the sustainability of this technology, including animal and fish feed, combustible briquettes for power generation, ethanol, compost, and construction fiber (Rezania et al. 2015). It is important to note that the process of phytoremediation may result in the plant waste becoming saturated with certain pollutants, particularly so in the case of bioaccumulating pollutants like heavy metals, although reducing landfill volume is an important goal for global communities. In the event the plant material has elevated pollutants, construction fiber and ethanol may be safer alternatives for tropical communities compared to using the plant material as stock feed or compost.

Lastly, plant-pollutant interactions should continue to be evaluated in the context of food safety and security in agricultural systems. Rice (*Oryza sativa*) is a wetland cereal crop (Fig. 7.5) belonging to the family *Poaceae* and is a food staple for over half the world's population (Muthayya et al. 2014). It is also noted to accumulate small amounts of As, Cd, Ni, and Pb from polluted agricultural waters (Sridhara Chary et al. 2008; Mao et al. 2019; Song et al. 2021). Of these heavy metals, Ni and Pb were found to exceed the safe limits for cereals and vegetables of the WHO (2010) and FAO (2007). While this species may be absorbing pollutants from agricultural waters, albeit in relatively small concentrations, the application of this species in polluted waters is not recommended if its end-use is intended for human or animal consumption.

7.8 Conclusions

A wide variety of anthropogenic pollutants are generated from animal and plant farming activities, and this is increasing in line with global productivity and population increases. Tropical regions are rich with agricultural activities, and certain pollutants, including animal wastes and other chemicals associated with farming, cause significant environmental degradation in surrounding ecosystems. Certain

Fig. 7.5 Rice produced in a monoculture field in tropical Indonesia (Fisk 2019)

pollutants have more phytoremediation data reported, for example, heavy metals including Cd and Pb. Other more ubiquitous, but less hazardous agricultural pollutants (e.g., P and NH_3) present an important opportunity for future investigation, given their widespread detection and potential impacts. Further in-situ research into phytoremediation systems tailored specifically for agricultural practices in tropical regions is merited, with strength in mixed-species terrestrial phytoremediation plantings as well as water-based systems. In addition, identifying new phytoremediator species that are locally endemic to tropical regions, as opposed to common crop species, can further support, and protect, the local biodiversity of valuable and vulnerable tropical regions.

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Chapter 8
Wetlands to Treat Mining Tailings in the Tropics of Central and South America

Aurora M. Pat-Espadas and Leonel E. Amabilis-Sosa

Abstract Wetlands are considered as natural filters because of their ability to retain contaminants from water passing through them. However, the performance of wetlands is also influenced by the activities of microorganisms and vegetation contained in them. Constructed wetlands are Nature-Based Solutions, which take inherent benefits and clean-up abilities from natural systems and adapt them via engineering principles to treat polluted water. Acid mine drainage and mine tailings can be treated by these wetland systems if their design appropriately addresses issues such as vegetation and lining material, which should be selected according to the resources available in the geographic area, to effectively promote increased pH in acid waters. The implementation of these systems in the Tropics is favored because of the prevalent weather conditions. Treatment by constructed wetlands of metal-containing water is less well understood compared to their implementation for wastewater treatment. Hence this field has considerable opportunities for further research.

Keywords Mine tailings · Metal immobilization · Potentially toxic elements · Constructed wetlands · Nature-based solutions

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8.1 Introduction

Water contamination is a worldwide topic of major concern; hence, different efforts are carried out to treat and remove the pollutants, especially the contamination by metals is of interest because it represents a threat to ecosystems and human health. Natural wetlands are recognized for their contribution to environmental restoration and ecological stability, and a wide range of other useful ecological function, including water quality amelioration by chemical and biochemical processes; hence, wetlands are sometimes called as natural filters.

Scenarios of metal contamination have encouraged the development of different technologies, such as chemical, physicochemical, and biological processes, to help and enhance natural attenuation, such as those provided by natural wetlands, to deal with pollution problems. Nowadays natural strategies, such as Nature-Based Solutions (NBS), defined as actions inspired by, supported by, or copied from nature (ECDG 2015) are becoming more important. In this context, Constructed Wetlands (CWs) are defined as man-made complexes of saturated substrate, emergent and submerged vegetation, animal life, and water that simulate natural wetlands for human use and benefits (Horst et al. 2020). This kind of solution offers social, environmental, and economic benefits. For example, CWs, being passive treatment systems, demand less energy as compared to active systems, while the water treated by these systems can be reused, and the plants, if harvested, can have economic value (Mburu et al. 2013).

This more holistic approach of remediation by CWs is well-aligned with sustainable development. In recent years, this option has been considered for industries and for remediation of abandoned mine sites (Figs. 8.1 and 8.2). In addition, CWs offer landscape and revegetation options, while at the same time removing toxic metals. In addition, the advantages of the CW for water treatment make it highly applicable in developing countries because they are relatively cheap to build, operate, and maintain (Mburu et al. 2013). It is considered that CWs are especially appropriate in the Tropics because the tropical warm climate favors two key elements of the systems that positively affect the water treatment and ensure the CW functionality, namely microbial activity and plant growth all the year-round (Tripathi et al. 1991; Kelvin and Tole 2011).

Among the common conditions of natural tropical wetlands, the high rainfall, high input of organic matter (often provided by overstory rainforest), and high mean annual temperature (Page et al. 1999; Sjögersten et al. 2011) all help promote CW efficiency. The effects of evapotranspiration on removal of metals are not well understood but need to be taken into account when assessing mass balances rather than just considering the difference between inflow and outflow concentrations (Bojcevska and Tonderski 2007). In addition, Tropical regimes tend to promote a faster ecological succession and abundant tropical biodiversity (Kelvin and Tole 2011).

Fig. 8.1 Abandoned mine tailings in the northwest region of Mexico

This chapter explains the main aspects of CWs as NBS for their application in the treatment of mine tailings and acid mine drainage with a focus on Tropical conditions (Fig. 8.2).

8.2 Mine Tailings/Acid Mine Drainage (AMD) Characteristics and Challenges for Treatment

Mine activity is necessary to satisfy the demand of the population for basic materials and is crucial for technological development and clean energy production (IEA 2021). Mining is an old activity all over the world (Coulson 2012) that has also left behind negative environmental legacies in many countries. Abandoned mine wastes generally contain high concentrations of highly persistent metal(oid)s that need to be remediated or mitigated to prevent pollutant transport (Valenzuela et al. 2020). Old tailings pose higher potential ecological risks than fresh tailings (Rubinos et al.

Fig. 8.2 Acid mine drainage examples in the Northwest Region in Mexico

2021). The abandoned tailings possess characteristics that are of great concern and also a challenge for remediation strategies, among others: large amounts of material, high toxicity, high costs, unstable landscapes, and risks of collapse. While some regions are aware of the locations of these sites, others are still trying to create databases to identify all their abandoned mine tailings.

A concerning and important problem related to exposed mining waste is acid mine drainage (AMD), which is mainly caused by the oxidation of sulfide mineral ores in contact with water (originating from precipitation, surface water, and pore fluids, run-off of rainfall, groundwater, or discharges). These waters, which drain from abandoned mining areas, are characterized by low pH (below 5.5), elevated concentration of potentially toxic dissolved metals (Fe, Al, Cu, Pb, Zn, Cd, Co, Cr, Ni, Hg, etc.), metalloids (As, Sb), and sulfate (generally, higher than 1000 mg/L) (Lottermoser and Ashley 2011).

The appearance of AMD is orange ochre (caused by ferric hydroxide precipitating when it reacts with dissolved oxygen in surface waters), and it can affect aquatic life and food chains. The generated AMD water elevates the level of dissolved metal in the receiving surface water stream hence harming ecosystems (Robb and Robinson 1995; Kefeni et al. 2017). It is important to emphasize that there is a wide variation of mineral content and dissolved ions in AMD according to the geological strata of individual mining areas (Kefeni et al. 2017). Hence, nowadays, there are methods to predict acid generation from mining waste (US EPA 1994). The problem of AMD not only occurs at abandoned sites but also at active mines, but the problem is often more visible at abandoned sites because modern mines try to prevent it following the current legislation.

Where the implementation is possible, with the required area available and adequate topographic studies, wetlands can be designed and constructed to favor treatment (Karna and Hettiarachchi 2018; Valenzuela et al. 2020). Challenges associated with AMD include high acidity and toxic metal concentrations which can adversely affect CW functioning (Pat-Espadas et al. 2018). Nevertheless, it has been documented that natural wetlands can remove metals from AMD such as iron and manganese (Akcil and Koldas 2006).

Other important aspects to consider in relation to AMD treatment and composition are climate, geology, and mine site practices. For instance, a study by Streten-Joyce et al. (2013) demonstrated that chemical composition and bacterial communities in acid and metalliferous drainage from the wet-dry tropics are seasonally-dependent. Their results show that the concentrations of elements in the AMD samples analyzed were higher in the dry season when water bodies were evaporating and lower in the wet season when rainfall diluted the discharge.

Preliminary calculations have shown that seasonal temperature variation can have a significant impact on the amount of AMD formed; hence, global warming can also influence important aspects (Street 2013). However, in the Tropics, the average temperatures are expected to rise less than in the poles; hence, this effect may not be of crucial impact, as calculations by Street (2013) based on chalcopyrite, with pyrite and bacterial impact of temperature on the formation of H_2SO_4 suggest about a 250% relative rate of increase in acid formation at 40 °C with 100%, 140%, and 180% increases at 25 °C, 30 °C, and 35 °C, respectively.

Other studies have explored how possible effects of climate change, i.e., increased rainfall and flooding events, can damage the mine tailings covers and dams, and how changes in rainfall patterns, evapotranspiration, and hotter temperature will affect vegetation (Anawar 2013). Hence, there are many reasons to encourage the appropriate design of remediation strategies considering all the factors required to adapt the technology to these impacts, including climate change and extreme conditions.

8.3 Constructed Wetlands Design, Importance, and Implications for Tropics Conditions

CWs as engineered systems for the retention of metals offer advantages such as no chemicals addition requirement, no labor inputs, little maintenance, as well as long expected lifespans (from 20 years (Skousen et al. 2017) to longevity in the order of several centuries (Beining and Otte 1997) which make them attractive for remediation of abandoned mine tailings and deposits. In addition, the treatment of mine wastes such as AMD by CWs offers the removal of sulfate found as contaminants in these types of waters (O'Sullivan et al. 2000). Since these wastes contain sulfide-minerals that are relatively immobile under waterlogged conditions (Gambrell 1994; Karna and Hettiarachchi 2018; Valenzuela et al. 2020) and wetlands, have a water table above or at the soil surface for a significant proportion of the year (O'Sullivan et al. 1999) they promote immobilization of metals.

As in wetlands, more than one process helps deal with metals, physical, chemical, and biological (Sheoran and Sheoran 2006); the characteristics of the CW can be selected and designed to achieve optimal overall performance. Basically, the system comprises four main components: substrate or supporting media, vegetation, microorganisms, and the water column (Sundaravadivel and Vigneswaran 2001). Regarding water flow, it is an important aspect of the design of CWs which can be classified according to this aspect into surface flow constructed wetlands (S-CW) and sub-surface flow wetlands (SS-CW). The first are systems in which the water level is above the ground surface. Hence water is exposed to the atmosphere and circulates between the vegetation. The second type has a water level below ground, usually 0.3–0.9 m, thus the water flow passes through sand or gravel beds (Davis 1995). The influent flow direction in the CW can be engineered to be horizontal or vertical, which also subclassifies them. There is also a third class of CW that incorporates the use of the mentioned types of surface and sub-surface flow in the arrangement (Davis 1995; Pat-Espadas et al. 2018). CWs can be designed using these various flow options to suit individual sets of site conditions.

All the configurations mentioned, S-CW, SS-CW, and hybrid systems have been implemented to treat AMD with acceptable performance and efficiencies but under different operational parameters at small and full scales (Pat-Espadas et al. 2018). Each design has its own set of advantages and conditions that need to be considered for integral solutions to issues such as the prevalence of wet versus dry-season conditions in the Tropics. For instance, S-CW designs usually implemented to treat AMD have metal removal efficiencies that are high but vary depending on the metal itself. Long-term field conditions have demonstrated that planted S-CWs achieve acidity removal in the range of 80–90%, and also the surface flow design is more resistant to external influences, especially heavy rain events (Kohler et al. 2004; Wiessner et al. 2006; Kuschk et al. 2006) mainly due to the water rising above the surface. Moreover, this design also permits a combination of aerobic (near-surface layer) and anaerobic (deeper waters and substrate) conditions, which could further promote transformations of different metals. The main disadvantage of these systems is the land area required, which is usually larger than other systems (Davis 1995).

On the other hand, SS-CW is recommended for AMD since the anaerobic conditions promote the sulfate-reduction process, driven by microorganisms under anaerobic conditions, which helps immobilization of metals of environmental concern that are difficult to remove in S-CW. As stated before, this configuration can be divided into a vertical and horizontal flow which also offers the distribution of the influent in the whole system. These systems are more appropriate for waters with low concentrations of solids to avoid clogging.

In addition, studies have found that small planted SS-CWs are sensitive to heavy rainfall which affects the stability of the AMD treatment processes (Kuschk et al. 2006). However, it also translates into the recommendation of SS-CW for large-scale systems (several thousand m^2) that are usually less sensitive to rain events (Stark et al. 1994). Some advantages of SS-CWs are cold tolerance (temperature is a crucial factor determining the performance of CW), minimization of odor problems, and, possibly, greater assimilation potential per unit of land area compared to

S-CW (Davis 1995). Regarding cold tolerance, it can't be considered a problem in the Tropics; however, there are some places with mining activities and AMD issues at high altitudes. For example, in 2005/2006, it was constructed a new water treatment facility to manage drainage from the expanding Tucush Valley waste rock dump at Antamina Cu-Zn-Mo mine, located in the rugged high Andes in Peru (4200 m above sea level). The treatment system comprised a combination of sediment ponds, serpentine channels, and wetlands. The wetlands cover six ha and are designed to treat up to 115 L/s for removal of nitrates, ammonia, and metals (Strachotta et al. 2009). Another example is Cajamarca, Peru, located at 2750 m above sea level, where ecological restoration has been carried out simultaneously with ongoing mineral exploration and AMD is planned to be treated with a series of CWs since area and scenario make possible the implementation (Macera et al. 2020).

For all designs, it is important to consider that in zones with prolonged droughts, wetlands may lose important amounts of water by evapotranspiration. In this situation, a supplemental source of water may be needed to maintain adequate levels (Davis 1995).

In general, the performance of CW is expected to change over time, for example, initial retention capacities may be high in the short term and change over the long term. The systems also need to be monitored for their service as habitats for plants, animals, birds, etc., over time. Another important aspect is related to the scale of the system, for instance, Maine et al. (2006) compared the removal efficiency of large versus small-scale free water CW and found that Cr, Ni, and Zn were retained better by macrophytes present in a large wetland but better in the sediments in a small wetland.

The pros and cons of the above designs can be overcome by the implementation of hybrid systems. The design, in this case, includes stages or cells, for example, the first one can be an S-CW followed by SS-CW to promote aerobic and then anaerobic reactions.

One of the limitations of CWs as treatment systems is that their performance or efficiencies may vary seasonally in response to changing environmental conditions, including rainfall and drought because wetlands interact strongly with the atmosphere through rainfall and evapotranspiration. This special consideration is important if effluent quality must meet standards at all times throughout the year (Davis 1995).

The substrate or supporting media can be selected from a wide range of options from natural to engineered materials. Traditionally it can be soil, sand, gravel, rock, and compost, but nowadays, other promising materials such as biochar are being considered (Davis 1995; Deng et al. 2021). However, selection criteria can be based on some important aspects such as those suggested by Wang et al. (2020): source and cost, hydraulic and engineering feasibility, ability to remove the contaminants, support for plant growth and microbial adhesion, safety, substrate plugging, and lifespan.

Special considerations of climatic perturbation to the systems can also be included in the design. Hence, the design must, at a minimum, consider AMD characteristics, water quality required, standards for effluent discharge, the area available, site aspect, long-term operation, and weather conditions.

8.4 The Crucial Role of Substrate in CW for Treating Mine Tailings/AMD

Only SS-CWs contain substrate or supporting media, and they have engineering aspects that are a major determinant of the removal efficiencies of any contaminant. On the one hand, flow within SS-CWs occurs through the supporting media interstices. The volume of these interstices is determined by the porosity of the SS-CW, and therefore, it defines the hydraulic residence time (HRT). This HRT is perhaps the design parameter most closely related to the removal efficiencies of contaminants contained in mine tailings/AMD. The supporting media geometry is not only crucial for hydraulic aspects but also for aspects related to microbial biomass. Indeed, the microorganisms present in SS-CWs are characterized by adhering to the substrate and growing in a biofilm. The physical data of the biofilm, such as volume and area, allow design and operational parameters related to the microorganisms such as organic load and cell residence time (CRT) of SS-CWs to be calculated (Hadad et al. 2018).

Biofilm aspects are a determinant for SS-CW removal efficiencies but are even more relevant when the contaminants are metals and metalloids contained in water with acidic pH values. Precisely the characteristics of mine tailings and AMD. These adverse conditions for the microorganisms can be tolerated depending on the biofilm structure, and, in turn, the biofilm depends on the interstices of the substrate (Li et al. 2019). Ji et al. (2021) demonstrated synergism between microorganisms and supporting media. The substrate increases its adsorption capacity for heavy metals, and microorganisms can release chelates that trap metals. These solids or precipitates are deposited in the smaller interstices of the supporting media or in the so-called occluded porosity, which is not considered during CW design.

As mentioned in Sect. 8.2, low pH levels characterize AMD, and it is expected that the most common supporting media used in artificial wetlands are those whose mineralogical composition produces neutralization reactions with the acidic aqueous medium. For the implementation of CW, it is recommended to use locally available material as supporting media to reduce costs and aspects such as adaptation of vegetation and microorganisms. The variety of rocks found between the Tropic of Cancer and Capricorn is wide. However, those of particular interest are limestone, gravel, dolomite and bentonite, and any other type containing an important fraction of calcite or carbonates (Zhang et al. 2020). For example, dolomite and limestone react with the aqueous medium of the AMD and release Mg^{2+} and Ca^{2+} ions, respectively. In turn, the hydronium ions are consumed to form bicarbonates (HCO_3^-), which ultimately results in an increase in pH, and stimulates the adsorption capacity of the substrate, especially for Fe, Pb, Ni, and Mn, the most abundant metals in

AMD (Nagy et al. 2020; RoyChowdhury et al. 2015). That situation suggests that the best-supporting media to be used in CW is limestone or dolomite, both abundant in karstic zones, which are generally found closer to the Equator (Álvarez-Rivera et al. 2021). Aspects related to the geometry of the supporting media directly influence essential design parameters such as porosity, HRT, CRT, and, therefore, removal efficiency. Nevertheless, if the water to be treated is mining tail/AMD, the supporting media must contain carbonates in its geological structure.

On the other hand, recently, organic substrates have been incorporated into the substrate. The rationale is the cation exchange and physisorption that exists between heavy metals and organic groups. In fact, in agriculture, it is essential to quantify the soil's organic matter content and cation exchange capacity to know the degree of metal fixation in the soil. Forests and rainforests can provide a wide variety of organic material options to enhance the substrate of CW in order to immobilize heavy metals from AMD. These have ranged from biosolids (Malgorzata Kacprzak et al. 2014), lysine fermentation products (Nagy et al. 2020), and biochar, an excellent promising material (Sui et al. 2021). Where the mining tailings are located in areas lacking limestone or dolomite limestone, the CW systems should consider incorporating organic material in the substrate to reduce the bioavailability of heavy metals. This inclusion can be performed as a pretreatment, resulting in hybrid systems (Zhang et al. 2020).

Also, a well-known function of the substrate is to support vegetation, which can play a crucial role in AMD treatment (Sect. 8.5). Unlike conventional wastewater, AMD does not contain nitrates, ammonium, orthophosphates, or other nutrients needed by plants. One option to solve this requirement for wetland vegetation is, again, the addition of organic materials, which can come from solid agricultural residues such as manure and stubble. Thus, the adsorption capacity of the supporting media would be improved, and the subsistence of the vegetation would be guaranteed.

8.5 Vegetation and Climate Considerations

Although vegetation is usually a component in the different types of artificial wetlands, some research has shown that it is possible to achieve high removal efficiencies of heavy metals in AMD without vegetation having a significant effect. For example, Singh and Chakraborty (2021) achieved average removals of chromium (97%), zinc (94%), nickel (97%), and iron (88%) both with and without the presence of *Typha latifolia*. The authors also demonstrated that the removal efficiency was mainly due to precipitation phenomena, whose kinetics is faster than phytoextraction (Amabilis-Sosa et al. 2015). In a study involving more plant species, Ma et al. (2015) found that 13 plant species are highly tolerant of acidic AMD conditions. Mainly, *A. auriculisform* and *Jatropha carcas* have demonstrated high performance in mining tailings remediation.

In some studies, perhaps the effect of vegetation was overshadowed by other processes. In artificial wetlands, many processes are stimulated for metal removal, and sometimes, chemical rather than biochemical processes such as precipitation could predominate (Singh and Chakraborty 2021). Thus, in SS-CW operated for periods longer than 60 days, the effect of vegetation is significant (Amábilis et al. 2015; Tauqeer et al. 2016). Indeed, the mechanisms that hydrophytes possess for metal removal/tolerance are based on immobilization or "sequestration" of these compounds to reduce toxicity. This phenomenon can occur by transport into vacuoles and cytoplasmic material. Processes can include chelation by root exudates and transport in ionic form by the different plant tissues, mainly by parenchyma (fundamental) and xylem (vascular) (Clemens 2017).

The variety among hydrophytic vegetation families is very wide. However, in terms of design it is possible to group vegetation into submerged, floating, and emergent. Submerged vegetation is found below the water surface, commonly rooted in the sediment. Floating vegetation is not rooted and floats on the surface of the water. Both submerged and floating plants are those used in S-CWs. Emerging plants are those that are rooted through rhizomes and roots, and the stem emerges towards the surface, so the leaves are the aerial part of the vegetation. Emerging plants require a support medium or substrate; hence, they are used in SS-CW. Table 8.1 shows different types of submerged vegetation distributed in the tropics and the most used in constructed wetlands.

Most of the research agrees that SS-CWs are the most appropriate wetlands for treating mining tailings and AMD due to the plant interactions (Pat-Espadas et al. 2018; Ma et al. 2015). Indeed, there are numerous detailed studies on the heavy metal accumulation capacity of plants, and that information can be considered in the design of SS-CWs (Das 2018; Bailey-Serres and Colmer 2014; Hadad et al. 2018; Li et al. 2019). First, it is necessary to differentiate between accumulation, translocation, and hyperaccumulation. All three are mass transfer phenomena and are not necessarily mutually exclusive. Accumulation indicated by the bioconcentration factor

Table 8.1 Different types of vegetation used in constructed wetlands implemented to treat mining tail/AMD in tropics

Scientific name	Vegetation type	Type of CW
Utricularia sp.	Submerged	S-CW
Chara vulgaris	Submerged	S-CW
Cabomba sp.	Submerged	S-CW
Riccia fluitans	Floating	S-CW
Nymphaeaceae	Floating	S-CW
Eichornia crassipes	Floating	S-CW
Pistia stratiotes	Floating	S-CW
Phragmites sp.	Emergent	S-CW and SS-CW
Echinochloa sp.	Emergent	SS-CW
Polygonum lapathifolium	Emergent	SS-CW
Typha sp.	Emergent	S-CW and SS-CW

refers to transferring some metal(s) from the aqueous phase to the vegetation, mainly in the rhizome and roots. Translocation is when the vegetation transfers the accumulated metals to the aerial part of the plant (leaves and stems). Ideally, the plants used should be hyperaccumulators, which means that the accumulated metal is mainly found in the aerial part of the plant (leaves and stems), but mainly that the vegetation can accumulate high concentrations of metal. Specifically, a plant is considered a hyperaccumulator when it contains above 1000 μg/g of the pollutant (Van der Ent et al. 2013). Translocation and hyperaccumulation would favor the recovery of metals through harvesting. For calculating any of the three phenomena, Eq. 8.1 can be used as the base equation, the result of which is the fraction of total metal contained in the wetland. Likewise, the bioconcentration factor is calculated by dividing the concentration of the metal in the plant (all tissues) by the concentration of the metal at equilibrium, and it is therefore calculated in controlled tests (Renoux et al. 2001).

In AMD, the bioconcentration and translocation values are dependent upon the type of vegetation and the heavy metal; hence, different values are reported. Table 8.2 shows the minimum, maximum, and average values accumulated by aquatic vegetation in the tropics.

A literature review of phytoremediation of AMD-contaminated sites in temperate ecosystems shows that the values for removal, survival, accumulation, and translocation by vegetation are lower than those compiled in Table 8.2. In botanical terms, this is expected due to the tropical climate effect, i.e., temperature, humidity, precipitation, evaporation, and weather seasons. For example, in Germany, a country with significant progress in applying CW, the differences in removal between summer and winter are remarkable. There are high statistical correlations between solar irradiance and removal efficiency (Zhu et al. 2017). One of the benefits of implementing CW in the tropics is that enough irradiance is guaranteed for plant photosynthetic processes.

$$\text{Accumulation} = \frac{\text{Metal contained in plant}, \frac{\text{mg}}{\text{kg}} \times \text{Plant dry mass, kg}}{\text{Metal in the influent}, \frac{\text{mg}}{\text{L}} \times CW\ \text{volume, L}} \qquad (8.1)$$

Table 8.2 Bioconcentration and translocation factors for ten species of tropical vegetation values taken or calculated from Amabilis-Sosa et al. 2015; Clemens 2017; Hadad et al. 2018; Li et al. 2019; Ma et al. 2015; Tauqeer et al. 2016

Metal	Bioconcentration factor			Translocation factor			Observation, n
	Min	Max	Average	Min	Max	Average	
Zn	92	572	132	4.5	86	32.6	22
Cd	7	1020	321	8.3	1050	89	15
Pb	13.5	1100	352	4.6	780	202	22
Cu	23	1650	803	19	116	71	22
Hg	76	2300	769	71	1700	482	8
As	195	2400	1150	1.8	47.2	376	12

Hydrophytic plants are not usually adversely affected by summer conditions (high temperatures and irradiation) in the Tropics unless their habitat dries out. Hence, the temperature is not a factor to consider when evaluating vegetation performance in CWs implemented for mining tailings and AMD treatment. Although temperature and atmospheric pressure directly influence evapotranspiration and precipitation values, these are considered in the hydraulic aspects of CW design, as will be addressed in Sect. 8.8.

8.6 Microorganisms for Bio-augmented Systems

Microorganisms, as ubiquitous inhabitants of the environment in general, have adapted to conditions prevailing in metal-rich environments. Microbes are well known for their capability to (bio)-transform and tolerate or resist metals through different mechanisms, i.e., biosorption, precipitation, extracellular sequestration, transport mechanisms, and/or chelation (Haferburg and Kothe 2007). Moreover, they have important roles in the biogeochemical cycling of all elements, including metals and radionuclides (Lloyd and Lovley 2001). Consequently, using microorganisms, mainly bacteria, for bioremediations purposes is a widespread strategy.

Plant-microorganism interactions are known to be complex processes of great interest since benign microorganisms promote plant health and growth, enhancing stress tolerance and pollutant detoxification, while plants provide habitat and protection for microbial populations (Weyens et al. 2009; Lugtenberg et al. 2002). Bacteria associated with plants may be found on the exterior (rhizosphere or phyllosphere) or in the interior (endosphere) of plants (Bulgarelli et al. 2013). For instance, in the rhizosphere, there is a particular group of interest, the plant growth-promoting rhizobacteria (PGPR), while bacteria are the principal colonizing microbes in the phyllosphere (as epiphytes). Finally, endophytes (colonizing internal tissues) are also considered for biotechnological use since they can benefit plant growth by assisting plants to overcome contaminant-induced stress responses, hence improving phytoremediation (Weyens et al. 2009).

The addition of some specific microorganisms (bioaugmentation) in addition to a careful section of plants may lead to better performance and enhance the treatment of metal-containing waters such as AMD. Certainly, it is important to consider details, for instance, specific groups or species of bacteria need a particular substrate, the growth rate is of importance as some microorganisms grow faster than others, and the initial concentration of microorganisms (inoculum) used to start the bioaugmentation must be adequate.

Regarding this last point, the progress in microbial communities has been documented for some CW. Hallberg and Johnson (2005) studied the microorganisms of a composite wetland ecosystem for remediating metal-rich, acidic mine drainage over a 30-month period. Their results revealed that in the aerobic wetlands, the heterotrophic acidophils dominated, but moderately and extremely acidophilic iron-oxidizing bacteria were also present in significant numbers. In waters draining the anaerobic compost bioreactors the dominant microbial isolate was an iron- and sulfur-oxidizing moderate acidophil closely related to *Thiomonas intermedia*. Other studies have reported on the performance of small experimental CW and short-term monitoring after bioaugmentation with bacteria (Ashraf et al. 2018; Yu et al. 2020).

A plausible strategy for microorganism selection is isolation or enrichment, starting from samples collected from polluted environments. In this context, metal-tolerant and resistant bacteria or consortia can be considered and studied. Also, sulfate-reducing bacteria are of special interest since they are the main contributors to metal removal in AMD-impacted environments, which includes wetlands (Johnson and Hallberg 2005). In addition, there are many bacteria associated with plants that can be considered, such as plant-growth-promoting and endophytic bacteria, as mentioned before. Hence, in these cases, the plant benefits, with potential knock-on improvements for metal removal and overall CW performance.

More studies are needed to explore the response of the bio-augmented CW systems at large scales and for long-term performance. Until now, evidence and theory suggest that under the appropriate conditions and application, this approach can be used to enhance AMD treatment. However, the influence of environmental factors such as climate conditions also plays a role in the activity of microorganisms. For example, Streten-Joyce et al. (2013) studied the seasonal behavior of the chemical composition and bacteria communities in acid and metalliferous drainage from the wet-dry tropics. Their results indicated that the bacteria community changed depending on the season (wet and dry) and time (year on year), while iron-oxidizing bacteria such as *Leptospirillum* and *Acidithiobacillus*, typically associated with AMD in temperate regions, were not prevalent in the tropical sites studied by the authors.

Microbes adapt and acclimate to their surroundings even to AMD pollution (Aguinaga et al. 2018); a good example is the number of extremophiles known. Consequently, it is important to consider all the factors, including temperature, season, plant species in the CW, chemical oxygen demand loading rate, and competition between microorganisms which may vary in time and impact the water treatment outcomes.

8.7 Metal Pathways in Constructed Wetlands

As described in the previous sections, heavy metals in the AMD are removed through the different compartments of the CW (vegetation, microorganisms, and substrate). However, in many investigations, the system's total efficiency is mentioned. The main potential of CWs to deal with potentially toxic elements (PTE) is the synergism that occurs between the compartments, and therefore SS-CWs have higher efficiency (higher synergism). Pat-Espadas et al. (2018) mentions that the components that interact the most are vegetation and microorganisms due to the symbiosis that promotes growth and translocation by plants. In particular, the exchange of substances between these two components is what reduces the toxicity of metals. However, vegetation depends on the substrate to establish its rhizomes and roots.

Likewise, microorganisms attach to surfaces and grow leading to biofilm formation, which adheres to the substrate. Any behavior is essential for synergy and tolerance, considering the acidic and toxic characteristics due to the heavy metal content. Synergism between CW components involves physical, chemical, and biological reactions acting together to remove heavy metals. This removal means immobilization. Thus, heavy metals may be precipitated, absorbed, adsorbed, or assimilated by vegetation (in any of its tissues), and even incorporated into bacterial cells in suspension. Studies on this aspect are scarce, but isolated studies have identified the involvement of genetic transcription processes, enzymatic processes, cell phytology, chemical kinetics, sorption theory, hydraulics, and water chemistry in general. Fortunately, the general acidity in AMD permits the prediction of the chemical speciation of heavy metals, and thus, we can know whether they are in colloidal, ionic, or elemental form (Amábilis-Sosa et al. 2015). These chemical characteristics reduce the time needed to evaluate the accumulation and distribution of metals within the CWs. Figure 8.3 shows the possible metal pathways in CW during the treatment of acid mine drainage. Mass balances can be performed based on this scheme to evaluate the accumulation and distribution of heavy metals in CW.

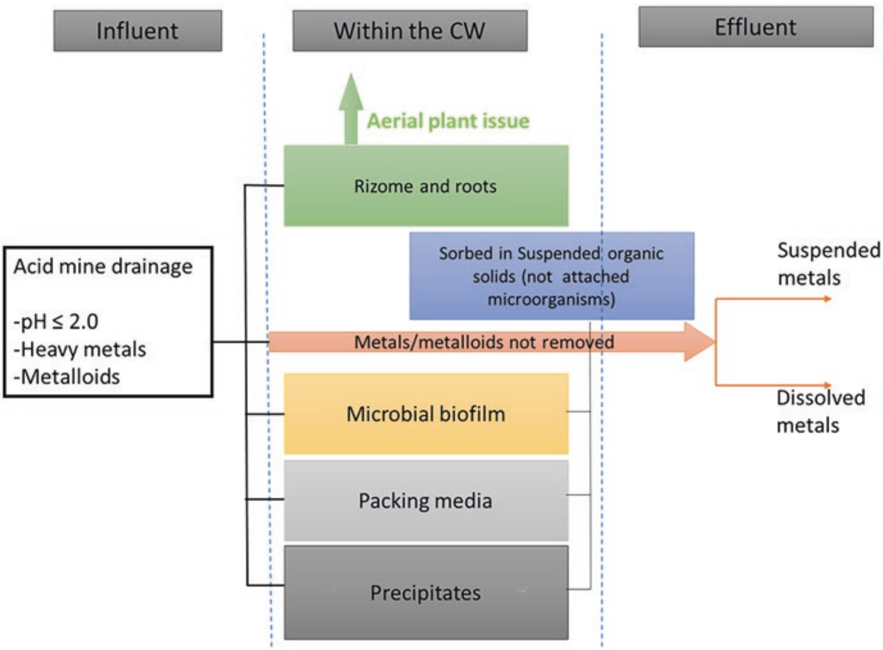

Fig. 8.3 Suggested distribution of heavy metals accumulated during the treatment of acid mine drainage by constructed wetlands

8.8 Theoretical Application of CW Design Based on Experimental Data

Based on the analysis of literature information on the use of CW for the treatment of Mining tail/AMD, it is noted that there is a vast amount of information, but at the same time, it is very diverse. So, to implement a CW at any scale, what is the best procedure to follow? If the same substrate or vegetation has been reported to have good but also not favorable results, which one should I use? These types of questions have not yet been answered beyond scientific investigations at the laboratory level and some at pilot scale (Singh and Chakraborty 2021; Zhang et al. 2020). There are guidelines and operation manuals for municipal wastewater treatment. Even the theoretical efficiency of CWs is calculated, which is still far away for the removal of heavy metals and the treatment of AMD and mining tailings.

Nevertheless, it is possible to combine three main aspects: the similarities with CW used for municipal wastewater, the mechanisms of heavy metal removal in CW, and the logical sequence of project maturation. In general terms, this means applying the hydraulic aspects for CW sizing, physicochemical characterization of the water to be treated, and subsequently using the substrate and vegetation with the best results for the physicochemical characterization, but with the restriction that these elements should be easy to acquire (in the study area). Figure 8.4 shows the

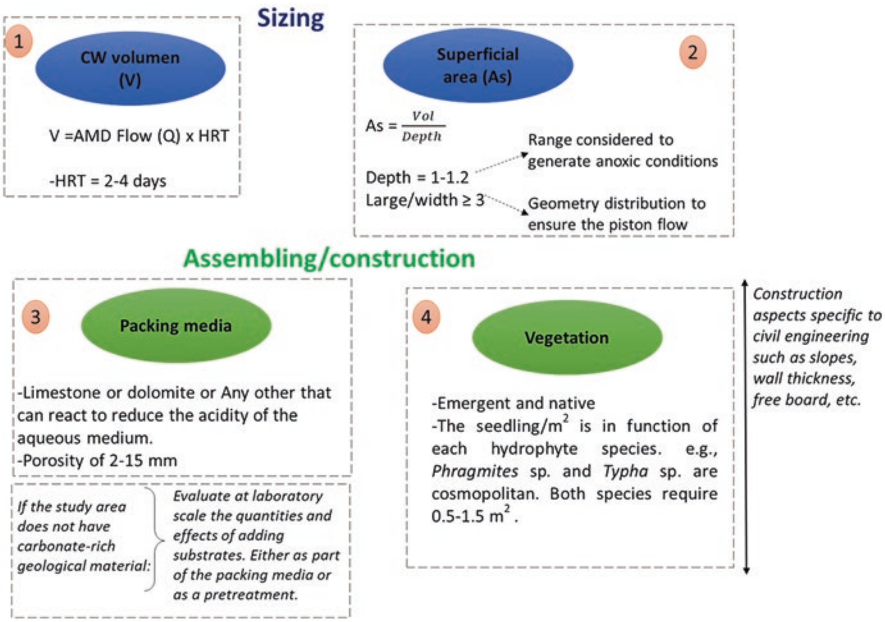

Fig. 8.4 Suggested design considerations for the use of constructed wetlands in the treatment of AMD/Mining tailings. Applicable for laboratory scale, pilot scale, and large scale. As the scale gets larger, it is necessary to incorporate aspects related to civil, economic, social, and environmental impact

flow diagram of the activities suggested for the design of CW for treating AMD and mining tailings. This diagram can be applied to CW design at any scale and is only part of the process to be carried out for full-scale implementation and to have standardized guidelines and manuals for the design and configuration of CW for treating AMD and mining tailings.

8.9 Conclusions, Perspectives, and Future Recommended Research

Wetlands as natural systems provide several benefits, among them, the retention of contaminants hence the engineered systems based on nature have been used for remediation and the experiences and results obtained, up to now, suggest their use for sites such as abandoned mine and brownfield. Design and planning are crucial aspects to achieve successful implementation of the systems; for instance, the correct selection of vegetation and the anticipation of perturbation caused by rain events and dry season are also recommended in the Tropics. In general, the efficiency and performance of CW under tropical conditions are expected to be high, making these systems potentially highly suitable for tropical regions.

Moreover, CW as a nature-based solution for remediation has emerged in recent years with increasing importance. Currently there is an increase in the number of publications, and more importantly, different sectors (industrial and governments) are more interested on support and impulse CW implementation. Additionally, it is important to remember that CW as an integrated solution also provides benefits in social aspects as well as ecological advantages.

Other important aspects to address in future works are related to the innovations in CW systems, from adding microorganisms to substrate or supporting media. For instance, the latter can be selected based on the available resources of the zone, i.e., rocks, and biochar obtained from specific biomass waste, among others. The use of microorganisms to enhance CW can be carefully considered but also more studies are needed to address the effect in long-term conditions. Area requirement is another relevant aspect to designing and constructing the systems, but it can be managed to adjust the engineering and configurations as options when necessary.

Finally, it is important to highlight that metal-containing water treatment by CW is still less known or popular than CW implemented for wastewater treatment. Hence, this field provides a great area of research opportunity.

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Chapter 9
Bioremediation and Biofuel Production Using Microalgae

Wei-Ta Fang, Chia-Hsuan Hsu, and Ben LePage

Abstract Constructed wetlands and the associated flora and fauna have had important roles in human history, such as providing food, water purification, flood control, biological habitat, recreation, and education functions. As the human population increases and living space becomes limited, the integration of constructed wetlands into the urban landscape will become important. At the most basic ecological level, microalgae are important primary producers, providing an important source of energy for wetland organisms and their respective food webs that extend to terrestrial ecosystems. They are a potential renewable energy source, and as a nature-based solution, microalgae offer considerable environmental advantages over other types of renewable energy sources. Microalgae grow rapidly, sequester atmospheric carbon dioxide, nitrogen, and phosphorous and some species are rich in oils that can be used to produce different types of biofuel. We explored the bioremediation and renewable energy roles these simple, unicellular organisms could play in the future.

Keywords Biofuel · Biogas · Biomass energy · Bioremediation · Constructed wetlands · Microalgae · Renewable energy · Wise use

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9.1 Introduction

Whether we are making full use of constructed wetlands and their natural resources to provide the wide variety of ecological services that improve our quality of life and practices that target sustainability goals around water use are important questions. The ability to construct and manage wetlands at the minimum, directly supports the United Nations' (2015) 6th – Clean Water and Sanitation and 7th – Affordable Clean Energy Sustainable Development Goals and indirectly the other 15 interlinked goals. The ability of aquatic and terrestrial micro- and macrophytes beneficial to remediate volatile organic compounds such as benzene, xylene, and toluene, the semi-volatile organic compounds such as polyaromatic hydrocarbons, and inorganic (metals and metalloid) contaminants in water, sediment, and soil substantially reduces the time, effort, and cost associated with implementing engineered approaches (Ayangbenro and Babalola 2017). With the development of human settlements during the Bronze Age, about 3200–1000 BC, human waste accumulated, and the people realized water management strategies were needed. Presumably, without knowing anything about the principles of bioremediation, the practices that the people implemented to manage their waste and stormwater were in fact early types of bioremediation (Angelakis et al. 2018). Today, these bioremediation practices would fall under ex situ applications (Azubuike et al. 2016). Around 600 B.C., the Romans specifically noted and used microorganisms to remediate their wastewater streams (Iyobosa et al. 2020). In the last 70 years, the use of genetically based techniques has advanced bioremediation technologies substantially (Azubuike et al. 2016; Malla et al. 2018). The ability to customize organisms such as algae and bacteria to target and eliminate specific contaminants from contaminated media and, in some cases, harvest certain types of contaminants such as metals and metalloids is showing great promise (Morais et al. 2021).

The number of species of algae varies from 30,000 to more than 1 million and possibly up to 350 million (Guiry 2012). Their classification is still under debate, but the color of the pigments in the chloroplasts plays a large role in the initial subdivision of the group (Sahoo and Seckbach 2015). The large forms, what we call seaweed or macroalgae, are clearly visible to the human eye, and species are assigned to the Phaeophyta (brown), Chlorophyta (green), and Rhodophyta (red) based on their color. The microalgae are obviously the much smaller microscopic forms that are also called microscopic algae, microphytes, unicellular, photosynthetic organisms, and/or phytoplankton (Figs. 9.1a–h). Despite their small size, they too can sometimes be seen by the naked eye, but only by the color of the water when trillions of individual algal cells occupy a small space in the water (Figs. 9.1i, j).

Fig. 9.1 (continued) 11 to 297 μm in length, (**g**) *Phacus* sp., a euglenoid genus that is about 30 to 40 μm long, (**h**) *Scenedesmus armatus* (Chodat) GM Smith, green algae that form colonies with individual cells that can be up to 10 to 20 μm long, (**i**) Algal reef in Taoyuan, Taiwan where the aforementioned microalgae were obtained, (**j**) a natural wetland showing microalgae accumulating in the center of the pond, and (**k**) a former water storage structure showing thick accumulations of microalgae that can be generated in simple reactors. (Photo credits: (**a**–**h**) by Jui-Yu Chou, (**i**) by Szu-Ju Wu, and (**j**–**k**) by B. LePage)

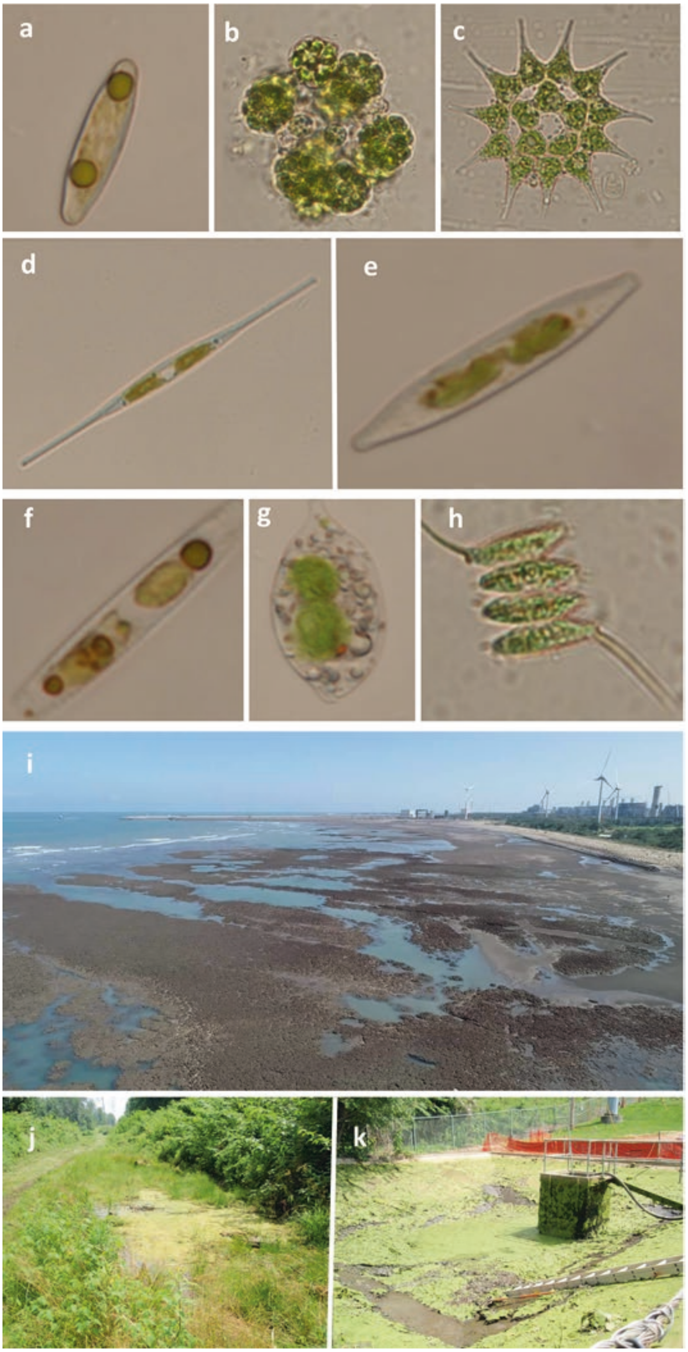

Fig. 9.1 (**a**) *Caloneis* sp., a genus of diatom with species that range from 8 to 20 µm in length, (**b**) *Coelastrum microporum* Nägeli, green algae that form colonies that can be up to 40 µm in diameter, (**c**) *Pediastrum simplex* Meyen Lemmermann, green algae that form colonies with individual cells ranging from 20 to 30 µm in length, (**d**) *Nitzschia* sp., a large genus of diatoms with species that range from 9 to 375 µm in length, (**e**) *Navicula* sp., a large genus of diatoms that range from 7 to 176 µm in length, (**f**) *Pinnularia* sp., a large genus of diatoms that range from

This occurs when high levels of nitrogen or phosphorous are present in the water, which then can trigger harmful red tides and algal blooms (Guy 2014). Macro- and microalgae are essential for life. They form the base of aquatic food webs, produce about 50% of the world's oxygen, serve as a carbon dioxide sink, sequester minerals and nutrients, and can also be used as food (Lembi 2003). Despite the immense taxonomic diversity of this group, we still don't fully understand the full range of ecological services that the algae provide or are capable of providing.

Microalgae have been used to remediate contaminants in wastewater streams, and are now being considered an important feedstock for biofuel production (Sharma et al. 2018; Hossain et al. 2019; Zewdie and Ali 2020). Microalgal biomass is one of several renewable energy sources and may become important in the future (Robertson et al. 2017). From 2019 to 2020, bioenergy accounted for 12–14% of the total energy consumed globally (World Bioenergy Association 2020). Biodiesel is one of many bioenergy products that can be produced from a microalgal feedstock in addition to animal fat and soy, canola, and palm oils (Chisti 2007). The chemical properties and performance characteristics of biofuel are similar to petroleum-based diesel fuel, and these types of fuels can be easily blended and/or replace petroleum-based diesel fuel, requiring few equipment modifications (Huang et al. 2012). Compared to fossil fuel production, biofuel can be produced inexpensively from a variety of feedstocks in large oil refinery-sized plants, natural and constructed wetlands, or using simple technologies at the village level (Wongsawaeng et al. 2019). Microalgae are also good sources of carbohydrates, lipids, proteins, and vitamins and can be used as slow-release green fertilizers because they contain high concentrations of nitrogen, phosphorus, and potassium in their tissues (Pan et al. 2014; Coppens et al. 2016; Bijay-Singh and Craswell 2021).

In this paper, we examine the use of microalgae for bioremediation and its potential as a renewable energy source. We then consider the role of using constructed wetlands in concert with certain species of microalgae to take advantage of the microalgae's inherent capability to produce oil. By better understanding these abilities, we can then use and better manage or control their growth for the purpose of harvesting the oil that they produce using constructed wetlands. Furthermore, techniques such as horizontal gene transfer, which is the lateral (non-sexual) transfer of genetic material between organisms, also provide opportunities to develop microalgal lineages that exclusively target existing and emerging contaminants and capitalize on microalgal oil production as a renewable nature-based energy technology.

9.2 Bioremediation

Bioremediation is an efficient and suitable alternative for removing contaminants from water compared to conventional wastewater treatment methods. The use of microalgae is an elegant and cost-effective use of nature to clean contaminated water (Touliabah et al. 2022). Microalgae can bioremediate pesticides (Singhal et al. 2022), metals such as arsenic, cadmium, chromium, lead, and mercury (Leong and Chang 2020), endocrine disruptors (Singh et al. 2019; Wang et al. 2019), and low- and high-molecular weight polyaromatic hydrocarbons (SureshKumar et al. 2018; Duarte et al. 2021; García and de Llasera 2021). Not only are they capable of removing contaminants from wastewater streams, they are carbon dioxide, nitrogen, and phosphorous sinks because these elements are needed for their growth. Their ability to remove pollutants from wastewater eliminates a step in conventional wastewater treatment systems, and the contaminants that they sequester could in fact be harvested (Nagarajan et al. 2020). Ultimately, microalgae can be used to produce biofuels, biofertilizers, high-value chemicals, and even the next-generation of organically grown microalgal biomass that is targeted at zero-waste policies and contributes to a more sustainable circular bioeconomy (Nagarajan et al. 2020).

9.3 Constructed Wetlands

Constructed wetlands in the tropics are becoming more prevalent to manage and clean stormwater and wastewater (Mander and Mitsch 2009; Zhang et al. 2015) and remediate contaminated ground and surface water (Afzal and LePage, Chap. 4 of this volume). By 2050, about 50% of the world's population will live in the tropics (https://www.science.org/content/article/expanding-tropics-will-play-greater-global-role-report-predicts), and water and waste management will become key issues and constructed wetlands along with sound water recycling and sustainability strategies will need to be part of global solutions. In Taiwan, for example, Yeh and Wu (2009) constructed a hybrid wetland system composed of an oxidation pond and two surface flow wetlands arranged in a series, which helped improve groundwater recharge. The practice of cleaning water in wetlands and then putting it back into the ground to recharge aquifers is the embodiment of sustainability. When wetlands are properly designed, built, operated, and maintained, they will often function like natural wetlands, providing the full range of ecosystem services such as those seen in native wetlands (Kadlec and Knight 1996; Otte et al. 2021). Constructed wetlands can then be built nearly anywhere where there is suitable hydrology, and they are becoming more widespread in and near cities (Fang et al. 2020). Increased biodiversity (Cao et al. 2007; de Martis et al. 2016), buffering of weather extremes (Huang et al. 2015; Turner et al. 2016), water storage (Omondi and Navalia 2020), improved water quality (Zedler and Kercher 2005; Mander and Mitsch 2009), and providing recreation and education opportunities (Anderson and Moss 1993; Faccioli et al. 2015) are some of the benefits that they provide. Of these, improving water quality is becoming more important as our population grows, cities become larger, and water use increases (Afzal and LePage, Chap. 4 of this volume). Moreover, bringing nature, such as wetlands, back to the city contributes to improved human health and wellness (Keniger et al. 2013; Hartig et al. 2014). In constructed wetlands, the wastewater that we generate can be treated using technologies that are modeled on natural processes and the interactions occurring between the vegetation, soil, and associated microorganisms that comprise the bacterial consortia, which include microalgae (Kadlec and Knight 1996; Kadlec and Wallace 2009).

9.4 Microalgae and Biofuel

The green muck that we often see in wetlands is microalgae, which is also a potential feedstock that can be used to produce biodiesel. Through a process called photosynthesis, microalgae and most plants have the ability to convert water and atmospheric carbon dioxide into the oxygen that we breathe, the carbohydrates (sugars) that are used by the plants, and sources of biomass energy that are in the form of carbon, lipids (fats), and oil (Singhal et al. 1999). Depending on one's point of view, the social, economic, and environmentally unsustainable costs of using carbon-based petroleum sources of energy continue to be justified even though we now know that the consumption of this type of energy has negative impacts on global climate and human health (Lelieveld et al. 2022; Rode et al. 2021). Currently, about 81–85% of the energy that is produced and used globally is from burning coal, crude oil, and natural gas (Bull 2001). Given our relatively recent acknowledgment that the environmental damage associated with the use of these types of fuel sources cannot be undone easily, we've focused our attention on other technologies that are capable of generating energy in ways that are renewable, affordable, and environmentally friendly. Owusu and Asumadu-Sarkodie (2016) provide a review of the types of renewable sources of energy, their sustainability, and ability to mitigate for climate change while considering energy security, energy access, social and economic development, and the reduction of environmental and human health impacts. Although each type of renewable energy technology has social, economic, and environmental benefits, they too all impact the environment negatively in a variety of ways. It is evident that a thorough review highlighting the positive and negative impacts of the types of renewable sources of energy on society, economic development, and the environment is needed, but it's well outside the scope of this paper and theme of this book. Nonetheless, we do provide a brief high-level overview later. Here we focus on a type of energy source that is nature-based, renewable, and the organism producing this source of energy can also be used to bioremediate contaminated water.

Nature-based solutions are the new buzzword and fast becoming an emerging approach to address societal challenges, biodiversity loss, human well-being, and climate change (Hanson et al. 2020; Seddon et al. 2020a, b). In and of itself, the concept is simple and philosophically Gaian. That is, the planet and its natural biotic and abiotic systems adapt and/or evolve to changing conditions. We've just not taken the time to study and understand what nature has to offer and how these systems work. We seem to be focused on definitions, classification systems, and how these approaches will contribute to meeting the United Nations' Sustainability Development Goals in the next 8 years (Hanson et al. 2020; Herrmann-Pillath et al. 2022). Humanity has become dependent on carbon-based fossil fuels to live, and we are slowly exploring other sources of energy, such as biomass, to replace our fossil-derived carbon sources (Ruiz et al. 2013; Gupta and Verma 2015; Adeniyi et al. 2018). The initial fear of replacing fossil fuels with biofuel is not unfounded and, in fact, may create environmental issues such as the loss of land and decreased biodiversity. Replacing one source of energy with another at the expense of the environment could impact future food supplies and create security issues because the land area needed to grow these energy-producing crops is usually at the expense of food-producing crops, which also contributes to environmental degradation (Herrmann et al. 2018). In addition, disease, pathogens, and/or poor weather could put energy-producing monocultures at risk, further contributing to global food and energy security issues. The "Water-Energy (Food) Nexus" concept identifies the need for water to produce energy, but energy is needed to produce clean water, and the water needed to grow the food is competing directly with that needed to produce energy. For example, 1700 liters of water are needed to grow the corn needed to produce 1 l of biofuel (Piementel and Patzek 2008; Piementel 2012). In 2005, approximately 36,000 million liters of ethanol and about 4000 million liters of biodiesel were produced, replacing about 2 and 0.3% of the gasoline and diesel used globally. Between 2000 and 2004, the amounts of ethanol and biodiesel produced grew at 10 and 25%, respectively (REN 212005). Between 2019 and 2020, 12–14% of the energy consumed globally was based on bioenergy (World Bioenergy Association 2020). Thus, while bioenergy may provide respite from our dependence on carbon-based fossil fuels, we need to exercise caution and/or common sense when considering alternative energy sources because of the negative impacts to the environment.

The advantages of using constructed wetlands and microalgae to produce biofuel are substantial. By constructing wetlands for biofuel production in different parts of the world, the risk of losing entire crops due to disease, pathogens, poor weather, and/or poor management practices is reduced. Furthermore, constructed wetlands can be built on brownfields, which puts land with little environmental or development value back into productive use. In addition, constructing wetlands for biofuel production can be managed so that lands that are currently used to grow food or provide other wetland functions that are not related to biofuel production are not eliminated or repurposed. Furthermore, the residual organic algal biomass that is not used for biofuel production, the leftovers, could be used as an organic fertilizer. The Sustainability-Peace Nexus is another new concept that is gathering attention and is closely related to the Water-Energy (Food) Nexus. The United Nations indicate sustainability is framed around the ecological, social, and economic elements of society (de Lucio and Seijo 2021). Virji et al. (2019) indicate sustainability and peace are closely tied to one another, and the future of global food and water security is linked to resource sustainability. Therefore, our Sustainable Development Goals must also consider the interactions between environmental change, socioeconomic development, global change, and peace.

9.4.1 Biotic and Abiotic Factors That Affect Oil Production

The potential of using microalgae to produce high-quality diesel to replace current fossil-derived fuels is high (Adeniyi et al. 2018; Gao et al. 2020). In general, the triglyceride (oil) content of oil-producing microalgae is 20–50% of their weight, and it may even approach 80% in some species (Adeniyi et al. 2018; Gao et al. 2020). By reacting the triglycerides with alcohol, a process called transesterification, diesel is produced (Demirbas 2008). Therefore, selecting species of microalgae that are capable of producing high oil content under different environmental conditions is important because the amount of oil produced depends on the microalgae's growth rate (Dickinson et al. 2017; Jang et al. 2012). Microalgae can grow up to twice their weight in a 24-h period, under a wide range of water types and environmental conditions (Hall and Benemann 2011). Robertson et al. (2011) showed that the annual production of triglycerides from microalgae can be as high as 140,000 l/ha.

An obvious disadvantage of using natural and constructed wetland ecosystems as reactors is that they are susceptible to change in the biotic and abiotic conditions that include temperature, light intensity, photoperiod, and contamination by other algal species, bacteria, and protozoa. These variables are not easy to control, which introduces difficulties, especially in large-scale operations. To minimize such problems, one can select species of microalgae that grow quickly under a wide range of environmental conditions yet still have the potential to be used as a renewable energy source (Chisti 2007; Pulz 2001; Rodolfi et al. 2009). Carbon dioxide is the main source of carbon that the microalgae use, so its solubility in water is a limiting factor for growth and oil production. Other limiting factors include nutrient availability, the activity of the micro-and other species of algae in the water column, and water clarity (Gani et al. 2019). Most of the light spectrum needed for photosynthesis is absorbed (absent) approximately 10 m below the water surface. In addition, the density of algae in the water column as well as the percent of total suspended solids/sediment will affect the depth of light penetration or dispersion and the depth at which algae can grow. However, the photosynthetic rate, which equates to the amount of carbon fixed and algal growth rates, can be managed by raising the water temperature or adding carbon dioxide to the water where the microalgae are being grown (Rao et al. 2007).

9.4.2 Energy Production from Biomass

The simplest way to generate energy from micro- and macroalgae is to collect, dry, and burn the material to produce heat. High-temperature and pressure methods such as pyrolysis, gasification, or hydrothermal upgrading can be used to produce gas or liquid fuels, but substantial amounts of energy are needed to operate the equipment (Onwudili et al. 2013; Tsita and Pilavachi 2013). These technologies all use dried algae as a feedstock and the drying process requires energy, which makes it difficult to balance or justify the energy produced and instrumentation costs (Wijffels 2008). Thermochemical liquefaction is a high-temperature, high-pressure process that also can be used (Banerjee et al. 2002; Dote et al. 1994; Tsukahara and Sawayama 2005), but this technology is still in the developmental stage. Nonetheless, the results of some studies suggest that this technology can be commercialized (Su et al. 2017).

The thermochemical reactions are carried out in an anaerobic environment, thus allowing the direct use of fresh, undried algae, which reduces production cost (Dutta 2021). One type of biogas produced is methane, which can then be used to generate heat or electricity and replace natural gas by thermal conversion (de Mes et al. 2003). The results of some experiments suggest some species of microalgae are unable to generate their maximum energy-producing potential through fermentation due to the robustness of their cell walls, so some pre-treatment of the microalgae is necessary to break down the cell walls, and the anaerobic decomposition of the algae is used to overcome this problem (Reith 2004).

9.4.3 Types of Biofuels

Biodiesel is composed of fatty acid methyl ester and ethyl ester molecules with a molecular weight of about 290 grams per mole. The triglycerides, also sometimes called lipids or fats, are stored in a type of cell called liposomes in the cell wall membranes and can be the main component of the cell membranes. Thermodynamically, the microalgae produce oil through photosynthesis using the sun's energy. That oil, which is now chemical (potential) energy, is stored in the microalgae's tissues. Depending on the species of microalgae, the lipids (oil) can constitute 2–60% of the dry weight of a cell. The oil can be used as pressurized fuels, such as straight vegetable oil for combustion engines (Mondal et al. 2008), and the triglycerides, free fatty acids, and lipid derivatives in the cell's cytoplasm (the liquid or solution that fills each cell) can be converted to biodiesel (Halim et al. 2012).

For more efficient biodiesel production, it is necessary to select microalgal species that possess high growth rates and produce high amounts of oil. If the microalgae are grown in open environments such as constructed wetlands compared to closed reactor systems such as tanks, then the species of oil-producing microalgae must have a growth advantage over the other microalgal species that may be present in that ecosystem. If one is taking advantage of the local environmental conditions, then the endemic species should be considered first as the feedstock source (Sheehan et al. 1998). Lipid accumulation in microalgae is usually an adaptation to external adversities or local conditions, so the availability of nutrient sources in a wetland environment is also a necessary consideration (Klok et al. 2013). There is a trade-off, however. Microalgae grow faster in nutrient-rich environments and accumulate fewer lipids compared to species of microalgae that grow in nutrient-poor waters, which equates to less potential energy. However, Rodolfi et al. (2009) showed that the primary productivity of microalgae did not decline when grown under natural conditions, even in nutrient-poor environments and that the microalgae accumulated twice as much lipid as is seen in cultures grown indoors. Of all of the microalgal byproducts, biodiesel is currently the most valuable, but its development is still in the early stages. Oil-rich microalgae include *Chlorella* sp. (28–32% oil content of the total weight), *Dunaliella primolecta* Butcher[1] (23%), and *Ettlia oleoabundans* (S Chantanachat *et* HC Bold) J Komarek (35–54%; Chen et al. 2011; Gouveia and Oliveira 2009).

9.4.3.1 Ethanol

Bioethanol is being used to replace the fossil fuels that we currently use and is produced by fermenting sugars through the hydrolysis of starch. Some microalgal species can contain more than 50% starch by weight and current technologies use cellulose and hemicellulose hydrolysis to produce sugar (Hamelinck et al. 2005), which makes it feasible to convert dried microalgae into bioethanol. Some properties of microalgae, such as the lack of lignitic tissue (an organic polymer that provides plant tissues with mechanical strength), make them optimal feedstock for bioenergy because they don't possess the recalcitrant lignified tissues that are more difficult to break down. In addition, microalgal cell walls are mainly composed of polysaccharides, which can be efficiently converted into sugar to produce bioethanol. Currently, the production of bioethanol from microalgae is being developed using molecular biotechnology to modify green algae (Deng and Coleman 1999). To learn more on how to produce bioethanol from treatment wetland plants.

9.4.3.2 Biogas (Hydrogen)

Gaffron (1944) discovered the ability of green algae to produce hydrogen in the presence of light. Hydrogen is considered an ideal fuel for mitigating air pollution and slowing global warming (Melis and Happe 2001). It's a convenient fuel source whose only combustion product is water (Haryanto et al. 2005). Greenbaum (1988) and Li et al. (2022) have shown that microalgae can photosynthetically produce hydrogen with a photon conversion efficiency of 6–24%, which was the highest ever recorded for hydrogen production by algae at that time. More recently, conversion efficiencies of 60–100% have been obtained (Bao et al. 2008; Kalisman et al. 2016). Large-scale hydrogen production can occur through water electrolysis, but the process is energy intensive (Stojić et al. 2003). Many purple non-sulfur bacteria are known to convert carbohydrates into hydrogen in the dark (Lee et al. 2002), and green sulfur bacteria can convert hydrogen sulfide or sulfur trioxide into hydrogen (Warthmann et al. 1992). While microalgae can use sunlight to convert water into hydrogen, an anaerobic environment is needed, which means that biogas production may not yet be economically feasible (Kapdan and Kargi 2006). Therefore, a better understanding of biogas production by microalgae and/or the use of technologies

[1] Taxonomic authorities follow https://www.ncbi.nlm.nih.gov/Taxonomy/Browser/wwwtax.cgi?mode=info&id=118562

such as horizontal gene transfer to tailor microalgae to produce biogas are needed (Melis and Happe 2001; Nagarajan et al. 2017; Li et al. 2022).

9.4.4 Biofuel Potential in Taiwan

Based on the available data on microalgae and biofuel production, we examined whether the constructed wetlands in New Taipei City could be used for commercial biodiesel production. Currently, not all of the microalgal species in the New Taipei City constructed wetlands have been identified. Nonetheless, the species of freshwater microalgae possessing a high oil content in their tissues were identified from the literature (Ratledge 1984) and the taxa (species) that were identified were then compared with the microalgal species that are known to be present in Jade Reservoir and recorded in the Taipei Algal Database (http://proj1.sinica.edu.tw/~dbalgae/). The microalgal species that could then be present in the New Taipei City constructed wetlands and wetlands from northern Taiwan were then identified based on these selection parameters (Table 9.1). If these or other microalgae possessing a high oil content are present, they could then probably be cultivated at a large scale in the New Taipei City and other constructed wetlands in Taiwan.

Botryococcus braunii Kuetzing is a freshwater microalga that can grow under a wide range of salinities, with the highest known being up to 3 moles of sodium chloride per liter of water (Qin 2005). The molarity of seawater is 0.6 moles of sodium chloride per liter of water. This species is one of the most common in Taiwanese waters and one of the best-known hydrocarbon producers (Hillen et al. 1982). The hydrocarbons that these microalgae produce can be converted into gasoline, kerosene, and diesel and the oil accumulates in the outer layer of the cell wall, making oil extraction easy. By adjusting the salinity of the water, the desired lipid content of the microalgae living in the water can be controlled (Wijffels 2008). Other factors that affect *B. braunii* growth and oil production include the amount of phosphorus and nitrogen, light, and pH of the water (Qin 2005). The only drawback of using *B. braunii* compared to other microalgal species is its slow growth rate. The time it takes for this species to double in number is about 72 h in the field and up to 48 h in the laboratory setting (Qin 2005). Other species of microalgae, such as

Table 9.1 Species of microalgae with high lipid content that are present in northern Taiwan's brackish waters/wetlands

Algae	Lipid content (percent)	Familial-level classification
Botryococcus braunii	Approx. 53–70	Chlorophyceae
Oocystis sp.	Approx. 35	Chlorophyceae, Oocystaceae
Scenedesmus sp.	Approx. 26	Chlorophyceae, Scenedesmaceae
Euglena sp.	Approx. 14–20	Euglenophyceae, Euglenaceae
Peridinium sp.	Approx. 36	Dinophyceae, Peridiniaceae

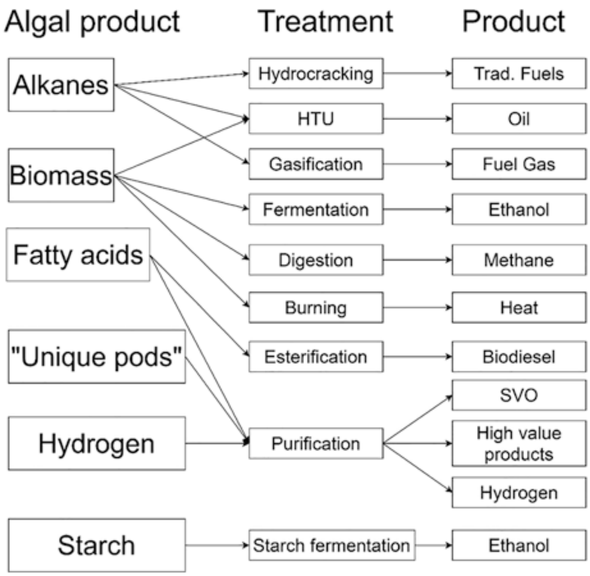

Fig. 9.2 Refining process of algae as raw material and the resulting end product, adapted and modified from van Iersel et al. (2009)

Spirulina platensis (Gomont) Geitler and *Dunaliella tertiolecta* Butcher, can produce the same amount of oil in one-third to one-half of the time. Therefore, it is necessary and important to study how to utilize the growth environment to achieve the best conditions that benefit *B. braunii* cultivation (van Iersel et al. 2009; Fig. 9.2).

9.4.5 Renewable Energy Sources

Current renewable sources of energy such as wind, photovoltaic, geothermal, and hydroelectricity are being touted as the future of energy production. The public has bought into various technologies without considering the cradle-to-grave aspects of the technology. In response to public demand/pressure, energy companies are adding renewable energy sources to their portfolios and sustainability models to maintain stock prices and investor satisfaction. There is no doubt that the environmental and human health risks associated with renewable energy sources are lower than fossil-based fuel sources, but a mix of renewable energy sources will need to be regionally specific. During the operational phase of their lifecycles, the benefits of renewable sources of energy include low maintenance and operating costs and reduced greenhouse gas emissions compared to coal. A detailed study that considers the cradle-to-grave environmental costs and risks of all renewable sources of energy is required. NIMBY (not in my backyard), however, plays a substantial role in what the public considers acceptable. Finding solutions to the environmental problems we currently face may never be resolved unless we have a well-developed and detailed stakeholder engagement plan so that the stakeholders understand the benefits and limitations, of each renewable energy source.

Even though renewables are being framed as environmentally friendly, they all negatively impact the environment, and these impacts start in the supply chain when the raw materials used to construct the assets are obtained and continue throughout the lifecycle of the asset, which includes maintenance and monitoring, decommissioning, and disposal. While this topic is clearly well out of the scope of this paper and book, it is important to point out that there is no perfect solution to any technology that we implement. For example, the loss of land, which then impacts environmental biodiversity, is common to all of the currently used renewable energy sources. Bird and bat strikes, mechanical and aerodynamic noise, and viewscape aesthetics are associated with wind turbines (Wang and Wang 2015; Pasqualetti 2000). There are no greenhouse gas emissions during their operation and the wind is clean, free, and as long as there is a sun to heat the Earth, the supply of wind may be inexhaustible (Jaber 2013). The most obvious disadvantage is that when there is no wind, energy cannot be generated. Solar energy is the use of photovoltaic cells that convert the sunlight into electrical energy and until our sun goes supernova, solar energy is inexhaustible. However, energy generation using photovoltaic panels is limited to regions where there is enough sunlight, and when it gets dark, energy cannot be generated. In addition, the human health and environmental risk of landfilling photovoltaic panel cells are a concern.

9.5 Conclusions

Cultivating microalgae in natural or constructed wetlands reduce the amount of useable land needed to produce biofuels, less water may be needed, and the environmental benefits of wetlands can be increased. Increasing biofuel production is attractive, but water supplies could be further stressed at a time when the water needed for agricultural purposes is already depleted. However, the use of microalgal alternatives to minimize human impacts related to energy use and production, as well as contamination of water resources, may provide attractive alternatives. Constructed wetlands, together with the use of microalgae, could play key roles in disaster mitigation, water purification, food supply, and education and recreation opportunities. Microalgae and plant macrophytes can also play an important role in sustainable energy production and increasing soil fertility. There are still many microalgal species in wetlands that have not been well studied and could be valuable for biofuel production. Microalgae have a huge potential to be developed as a technology to produce biofuel.

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Chapter 10
Wetlands for Remediation in Africa: Threats and Opportunities

Oscar Omondi Donde, Austine Owuor Otieno, and Anastasia Wairimu Muia

Abstract Natural wetlands in Africa continue to decline in surface area and water quality. Still, there is scant information about the nature and causes of the decline, despite African countries' increasing efforts to sustainably manage the wetlands. African countries have embarked on efforts to promote sustainable pathways to balance the development of wetlands for sustaining livelihoods with conservation and maintenance of ecosystem services, such as improvement of water quality and biodiversity. However, there is still inadequate information about the application of wetlands in remediation and the threats to wetlands under the pressures of changes in climate and population. This chapter reviews the status of Wetlands in Africa and their potential use in remediation to highlight emerging opportunities for sustainable management of ecosystems and uncovers the gaps and challenges that may deter proper implementation and application of wetlands for remediation of environmental problems in Africa. We recommend that constructed wetlands be integrated into management plans for conservation of both soil and water resources of watersheds and as parts of major restoration projects in Africa.

Keywords Wetlands in Africa · Remediation opportunities · Remediation processes · Wetland conservation

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10.1 Introduction

10.1.1 Background on Wetlands in Africa

Worldwide, wetlands have continued to experience degradation, with only 13% of their former extent still in existence since the 1700s (Gardner and Finlayson 2018). Within Africa, the same trend applies, but there is little information about the extent of decline on the continent (Davidson 2014). Based on existing indicators, there is increasing pressure on wetlands in Africa (Gardner and Finlayson 2018), driven by anthropogenic activities such as the development of hydropower dams (Zarfl et al. 2015), continued population growth, and agricultural and urban development (Beuel et al. 2016). Approximately 4.33% of Africa's total land mass, estimated to constitute over 131 million ha, is covered by wetland ecosystems which play a significant role in the provision of vital services for the survival of millions of communities (Rebelo et al. 2010). As climate change will exacerbate the decline of wetlands, African countries have made remarkable progress with developing policies for wetland conservation and management, as evidenced by the number of African nations which have become signatories to the Ramsar Convention on Wetlands with a total of 415 Ramsar sites (Moomaw et al. 2018). Nevertheless, embracing wise-use strategies for wetlands remains challenging due to inadequate capacity for policy implementation (Ostrovskaya et al. 2013), as well as knowledge gaps as far as wise-use is concerned.

Natural wetlands have been used as convenient wastewater discharge sites for as long as sewage has been collected (Kadlec and Reddy 2001). In Africa, some wetlands are also currently being used for that purpose. With wetlands being subjected to the Ramsar Convention of 1972 on biodiversity conservation, the use of natural wetlands for wastewater treatment became limited at the end of the twentieth century. Although some natural wetlands in the past have been effectively used for water quality improvement, this has been discouraged (Reed 1991) instead of promoting their many alternative values and functions while eliminating long-term consequences of wetland destruction (Kadlec and Reddy 2001).

African countries are continuing efforts to find sustainable pathways to balance sustainable development of wetlands in support of the livelihoods of millions of African people with conservation of wetlands, maintaining ecosystem services such as improvement of water quality and biodiversity (Simaika et al. 2021). Therefore, through a literature review using search terms such as African wetlands, bioremediation and gaps, knowledge, threats, and opportunities, this review article discusses the status of wetlands in Africa and their potential use in environmental remediation, considering wetland hydrology and ecological characteristics that are key to the efficient functioning of the wetlands based on the type of pollutants. We highlight gaps and challenges that may deter proper implementation and application of the wetlands in Africa in remediation.

10.1.2 Types and Location of Major Wetlands in Africa

Despite wetlands covering a significant proportion of Africa's land mass, most of them are yet to be studied for a complete comprehension of their environmental attributes (Amler et al. 2015). In terms of their distribution across the continent, the most prominent concentration of wetlands is between 15°N and 20°S. These include wetlands of the four major river catchments, including the Nile, Niger, Zaire, and the Zambezi. Other important wetlands are Lake Chad, the wetlands of the Inward Niger Delta in Mali, the Rift Valley lakes Victoria, Tanganyika, Nyasa, Turkana, Mweru, and Albert, the Sudd in southern Sudan and Ethiopia, and the Okavango Delta in Botswana, all of which are rich and unique in terms of biodiversity (Bailey 1989). Other types of wetlands are found in saline and brackish coastal and marine regions around the continent (Rodrigo 2021). These include the mangrove woodlands of eastern Africa extending from the coastal cities of Kisimayu in Somalia to Maputo in Mozambique and broken but wealthy stands along the West African coastline from northern Angola to Tidra Island in Mauritania, covering a total area of around 1.7 million hectares (Sieben et al. 2022). Coral reefs and seagrass beds are found along the coasts as mangroves, but mainly along the warm Indian Ocean coastline and less commonly along the Atlantic Sea. In addition, there are coastal pans, lagoons, and swamps, such as the Ébrié and Tadio tidal pond complexes of Cote d'Ivoire (Brenon et al. 2004). Further north of the 15°N latitude and south of the 20°S latitude, there are many more wetlands, including the inland desert springs and chotts of northwest Africa, the Oualidia and Sidi Moussa tidal ponds in Morocco, the Limpopo Stream floodplain in southern Africa, the Banc d'Arguin of Mauritania, and the St. Lucia wetlands in South Africa (Brenon et al. 2004; Denny 1993).

10.1.3 The Isolated Wetlands in Africa

The term "isolated wetlands" is relatively recent and is used loosely to define wetlands or ponds that lack a surface outlet to rivers and bays (Krieger 2003). Such wetlands and ponds typically form in depressions in the landscape and are "isolated" because the higher elevation of the land around them keeps water from flowing further downhill and downstream through even small rivulets. In their report on wetland characteristics and boundaries, the National Research Council defines an isolated wetland as a "wetland not adjacent to another body of water." Isolated wetlands are among the most seasonal ecosystems in semi-arid Africa. Their role is essential but varies throughout the year for different user groups. For example, the fishermen fish more in these ecosystems during flooding time of the year while the farmers undertake more farming on the wetlands during drought. Such wetlands are under threat from global climate change, and thus, the chains they form part of are under threat (Krieger 2003; Marton et al. 2015). Within Africa, such as in Niger alone, there are more than 1000 isolated wetlands of some size (1000–2000 ha), some of which have only been formed recently (De Roeck et al. 2008; Marton et al. 2015).

In Africa, peatlands are the most common type of isolated wetlands. African peatlands consist of moors, bogs, mires, peat swamp forests, and permafrost. Peatlands in Africa typically cover small areas. Estimates of the continent's totals vary between 4,856,500 and 5,853,400 ha of peatland (Grundling and Grootjans 2018). The African countries with the largest areas of peatland include the Democratic Republic of the Congo, Uganda, and Zambia. In Uganda alone, there is 64,000 ha of permanent swampland and as much land temporarily inundated in the wet seasons (Hesslerová and Pokorný 2010). Arid and semi-arid areas of Africa are also often characterized by seasonal rainfall and wetlands that retain water long after the rest of the landscape has dried out. These wetlands include rivers, swamps, lakes, and springs that dry up for certain portions of the year and the small rivers that feed Lake Turkana in Kenya (Hesslerová and Pokorný 2010).

10.2 Natural Wetlands in Africa and Remediation

10.2.1 African Wetlands for Remediation

Globally, the land cover of natural wetlands is rapidly dwindling, especially in inland areas. Data and information about trends in wetlands in Africa are limited (Davidson 2014). Human activities negatively impact the wetlands (Mitchell 2013; Willbroad and Kiyawa 2019). In Kenya, an attempt has been made by the National Environmental Management Authority (NEMA) to develop a Wetland Monitoring and Assessment Strategy (WEMASK) that will provide scientifically sound strategies for monitoring and assessing the status and trends of wetlands countrywide. Various guidelines for system restoration, clean-up, and habitat protection for marine and freshwater wetlands have been suggested to protect natural wetlands in Africa from destruction. Some potential remediation strategies for restoration are discussed (Feka and Morrison 2017; Schuijt 2002).

Natural wetlands are often referred to as "earth's kidneys" because of their high and long-term capacity to filter, sequester, and attenuate pollutants from the water that flows through them (Sharifi et al. 2013; Mitsch and Gosselink 2015). They have a high potential for improving water quality, for example, from agricultural runoff, oil spills in coastal areas, and acid mine drainage (AMD) that are often diffuse and of nonpoint origin and form a significant component of water pollution in Africa and other parts of the world (Reichenberger et al. 2007).

Investigations on the remediation potential of wetlands in Africa have been carried out at research facilities, nurseries, and a few field trial levels. However, broad field tests that cover diverse wetland types, from African coastal salt bog to riparian forested wetlands, have not been reported. The use of African Wetlands in

Fig. 10.1 Major aquatic plants serving as buffer zones within Lake Naivasha wetland

remediation depends on the presence and types of vegetation, microorganisms, and other physical and chemical properties. The part they play in the degradation of specific pollutants has not yet been broadly studied (Bruton 2021).

Apart from acting as buffer zones (Fig. 10.1), wetland vegetation directly decreases water pollutants through plant uptake and indirectly by providing a soil environment amenable to microbial degradation. Wetland plants enhance the degradation of pollutants by oxidizing the substrate in the rhizosphere by radial oxygen loss from the roots and by producing root leachates that act as organic substrates for microbes (Larkum et al. 2006). Within the generally chemically reduced soil of wetlands, oxygen may constrain microbial growth, but the oxygen provided by plant roots creates a heterogeneous microenvironment inducive to the degradation of pollutants. However, studies to provide an in-depth understanding of the characteristics of African wetlands in the remediation process are still rudimentary and need to be improved (Rebelo et al. 2010).

10.2.2 Mechanisms of Bioremediation Processes in Wetlands

Although there are similar values and functions offered by both constructed and natural wetland types in Africa, the latter tend to offer a wider range of benefits as opposed to constructed wetlands that are designed and fitted in the landscape to achieve a specific target. Most of these wetlands (Table 10.1) have hydrological, plant, soil, and microbial characteristics that have given them a better natural ability to control pollution, hence are considered better ecosystems for achieving bioremediation efforts in Africa. Under natural wetlands, bioremediation uses biological organisms such as bacteria, fungi, algae, and plants to degrade or detoxify

Table 10.1 Selected African wetlands with remediation potential

Potential functions	Examples of African wetlands	References
Recreation, tourism, and bioremediation	Saiwa Swamp, Yala swamp of lake victoria basin, Kenya	Okeyo-Owuor and Raburu (2016)
Biomass export and bioremediation	Lake Victoria	Kayanda et al. (2010)
Nutrient (effluent) retention and bioremediation	Nakivubo and Luzira swamps, Kampala	Willbroad and Kiyawa (2019)
Sediment retention and bioremediation	Awetu and Boye wetlands, Southwest Ethiopia	Mereta et al. (2020)
Groundwater recharge and bioremediation	Ondiri swamp in Kenya	Awuor (2008)
Microclimate stabilization, bioremediation	Kabale district valley swamps, Uganda	Willbroad and Kiyawa (2019)

contaminated environments' organic and inorganic pollutants. Thus, the process of phytoremediation is a bioremediation process that uses green plants and associated microorganisms (bacteria, fungi, and algae) along with proper soil amendments and agronomic techniques to remove pollutants from the environment or render them harmless (Reichenauer and Germida 2008). Phytoremediation processes are due to wetlands' characteristics, which decrease concentrations of pollutants through degradation and transformations, rendering them less harmful. These processes differ based on the type of pollutant (Azubuike et al. 2016). The reduction of metal concentrations in water and sediments, among other processes is achieved through accumulation and storage in the rhizospheres of the wetland plants (Kong and Glick 2017).

Phytoremediation approaches have been applied to organic and inorganic pollutants in soil and water. It provides promising approaches for the remediation of polluted sites due to its competitive performance, cost-effectiveness, and environmental friendliness. This is because it derives energy from daylight, preserves soil properties, and sustains high levels of microbial biomass (Yan et al. 2020). Five phytoremediation processes are recognized based on the contaminant removal mechanism: phytodegradation, phytostabilization, phytotransformation, phytoextraction, and rhizofiltration (García et al. 2010). Underlying phytodegradation mechanisms are uptake directly by plants and subsequent degradation or storage inside plant tissues, or the plants secrete enzymes that degrade or transform pollutants externally (García et al. 2010). Phytoextraction or phytoaccumulation is the accumulation of pollutants in the plant tissues in some species associated with subsequent excretion (Ghori et al. 2015). Phytotransformation, also called phytodegradation, is the breakdown of natural contaminants taken up by plants through metabolic processes inside plants (Newman and Reynolds 2004). Phytostabilization is the use of plants to decrease mobility and transport to the groundwater or air (Bolan et al. 2011). Rhizodegradation is the breakdown of pollutants within the rhizosphere (the soil encompassing the roots of plants) by microorganisms. Rhizofiltration is the assimilation or precipitation onto plant roots (or retention into the roots) of contaminants (Dushenkov et al. 1995). More than 400 plant species, mainly in the families, Asteraceae, Fabaceae, Lamiaceae, and Brassicaea, have shown potential for accumulating metals such as Co, Cu, Cd/Zn, and Se (Lone et al. 2008).

10.2.3 Bioremediation Opportunities Under African Wetlands

Wetlands and their aquatic plant communities play an important role in improving water quality. The African larger emergent plants such as cattail (Typha spp.), take up and remove nutrients (i.e., phosphorus and nitrogen) and break down contaminants and toxins from the sediment and water, incorporating them into their plant material or biomass. The role of natural wetlands in improving water quality (bioremediation) is also widely experienced in Africa, especially for the treatment of stormwater runoff, municipal wastewater effluent, and treating contaminants from landfill leachate and mine tailings ponds (Table 10.1).

For effective bioremediation with wetlands in Africa, a thorough understanding of the primary biotic and abiotic factors controlling pollutant degradation in wetlands and how nutrients, oxygen, temperature, and their interactions limit remediation of pollutants is required (Salimi et al. 2021). It is essential to understand the processes in the plant rhizosphere, i.e., that zone in the soil in which plant roots affect the biogeochemistry, including microbial activities in removing pollutants. Knowing whether there are plant species-specific differences in capacities to accelerate pollutant degradation and tolerances, soil oxidative capacity, root architecture and distribution, root exudate release, and rhizosphere development would benefit optimization of bioremediation with wetlands (Abhilash et al. 2009). The impacts of opposing and synergistic processes on pollutant removal, such as binding to substrates and mobilizing pollutants, require further examination. However, much has already been known from other parts of the world, and such information can be used to support the bioremediation approaches in Africa to avoid delaying the application. Moreover, awareness must be created to improve people's acceptance of remediation approaches.

To maximize remediation processes in wetlands, additions to the substrate of materials that enhance binding and degradation (biostimulants) may be considered. Nutrient amendments may enhance such processes. For instance, the nutrient that most limits microbial degradation in wetlands may be added to stimulate microbial activity (Greer et al. 1998). Slow-release fertilizers may be more effective than soluble fertilizers, perhaps in combination and with repeated applications (Xu et al. 2005). Utilization of non-fertilizer applications, such as soil oxidants, surfactants, and dispersants, may also be considered (Fragkou et al. 2021; Nikolova et al. 2021).

The underlying principles are the same regardless of location worldwide, but some conditions are unique to Africa. These include climate conditions, high pressure from dense and increasing population densities in places rich in wetlands (for example, the countries surrounding Lake Victoria), expanding deserts resulting in increasing demands for water along their boundaries, and threats to iconic African wetlands such as the Okavango, and the deltas of rivers such as the Niger, Zambezi, and Nile.

10.2.3.1 Remediation in Oil Spills

Oil pollution is a common occurrence in marine and freshwater environments and the most significant pollution to be found in all coastal areas of the African continent, with the Niger delta being the most impacted (Ordinioha and Brisibe 2013). Sources of oil spills are tanker accidents, rupture or leakages, loading and unloading of oil in ports or offshore facilities, and construction sites that have caused much environmental damage (Adeola et al. 2022). Oil causes harmful effects on the flora and fauna of wetlands. It may cause damage to fur and feathers on animals, and toxic residues may settle in sediments, potentially causing further damage (Ordinioha and Brisibe 2013). Oil remediation may include flotation and washing, coal agglomeration, thermal desorption, ultrasonic desorption, bioremediation, chemical oxidation, and ionic liquids extraction (Agarwal and Liu 2015). Bioremediation is considered to be one of the most promising clean-up processes of oil spills (Balba et al. 1998) and could be a suitable option to clean up freshwater and coastal wetlands in Africa, including those countries experiencing rapid industrialization and increased oil and gas production (Kabenge et al. 2017). Bioremediation through bio-stimulation has been identified to have a high potential for cleaning polluted sediments in the Niger Delta. Ex-situ trenching and treatment is recommended for groundwater treatment, while bioremediation is recommended for contaminated soils (Nuhu et al. 2021). Adding nutrients in the form of fertilizers to contaminated environments may accelerate the natural biodegradation processes (Thavasi et al. 2011). Such approaches should be considered in other African coastal and inland wetlands with frequent oil spills (Kabenge et al. 2017).

10.2.3.2 Remediation in Acid Mine Water

Highly acidic and greatly concentrated metallic streams are characteristics of metal and coal mining operations and pose a threat to the health of humans and other organisms (McCarthy 2011). There are still no effective methods to clean acid mine drainage (AMD) pollution. Sediment bacteria in natural wetlands are key in immobilizing metals such as iron and trace metals in natural wetlands subjected to acid mine drainage (AMD) pollution (Dean et al. 2013). For example, sulfate-reducing bacteria (SRB) render metals immobile by producing insoluble sulfides, and thus, wetlands receiving acid mine water at inlets of lakes passively improve water quality by increasing pH and removing of SO_4^{-2} and metal ions, as in the case in the immediate vicinity of mine tailings in and around Johannesburg in South Africa (Tutu et al. 2008). Along Klip River, a natural peat wetland that has received acid mine contaminated water from gold mining operations in Johannesburg, South Africa, for a long time (130 years), showed that wetland systems, possibly through biofilm formations on plant roots, can accumulate large quantities of metals, and thus remediate polluted waters (Humphries et al. 2017). The wetland system performs a vital ecosystem service in the environment by trapping metals that would otherwise enter the Vaal River system downstream. Such wetlands utilize naturally available energy sources such as topographical gradients, microbial metabolic activity, and photosynthesis to precipitate the metal ions by reducing the acidity of the water (Humphries et al. 2017). Bioremediation approaches may include the use of indigenous South African grasses such as *Hyparrhenia hirta* and *Setaria sphacelata* which display high potential for the treatment of this kind of pollution (Ramla and Sheridan 2015).

10.2.3.3 Remediation in Nutrient and Sewage Pollution

Natural wetlands around the world have been used as convenient wastewater discharge sites for sewage disposal for many centuries, mainly because wetlands were generally regarded as places useful only for dumping wastes. However, because it was recognized that wetlands improve water quality, they have been used and constructed for that purpose specifically for decades (Kadlec and Reddy 2001). The use of natural wetlands for this purpose has been discouraged since the establishment of the United Nations Ramsar Convention in Ramsar in Iran in 1971, which designates and stimulates the sustainable management of wetlands of international importance. Nevertheless, some natural wetlands are still used to control water pollution in Africa (Kansiime and Van Bruggen 2001; Mwanuzi et al. 2003). It is also worth noting that that natural wetlands still play an important role in the improvement of water quality as they act as buffer zones surrounding water bodies. Indeed, ecological studies in riverine wetlands along the densely populated areas in Ethiopia showed significant pollutant attenuation of organic and inorganic pollutants and high diversity and low abundances of tolerant taxa of macroinvertebrate indicators in sites after joining the riverine wetlands (Sileshi et al. 2020).

Modern agriculture utilizes fertilizers and pesticides to secure high crop yields and contributes to diffuse pollution, also referred to as a non-point source of pollution, to wetlands (Reichenberger et al. 2007). This negatively affects receiving water bodies, aquatic ecosystems, and human health (Li et al. 2011). Fisheries may also be affected if fish and invertebrates suffer poisoning and/or mortalities due to the biological degradation of organic matter, which can lead to hypoxia, anoxia, and anaerobic conditions (Njiru et al. 2018). As such, there is a need to remediate African wetlands from such effects to avoid degradation and ultimately destruction. It has been found that the redox conditions in most natural wetland soil/sediment zones enhance degradation pathways, requiring conditions with wetland plants remediating pollutants and contaminant loads in water and sediments (Williams 2002). This is more so by using expansive rhizosphere of wetland herbaceous shrub and tree species that provide an enriched culture zone for microbes involved in degradation (Williams 2002).

10.2.4 Elimination of Microbial Pathogens in Wetlands in Africa

The survival and multiplication of human pathogenic and antibiotic-resistant bacteria in ecosystems are of increasing concern but have been little explored. Wetlands can be contaminated by water fluxes from rivers and may present environmental conditions leading to bacterial survival and multiplication. Indeed, it has been shown that rivers can carry pathogenic microbes to wetlands making them a source of health risks. These pathogens can reach wetlands through other pathways such as hillslope groundwater or leaching from contaminated fields (Henriot et al. 2019). Therefore, wetland contamination does not fully depend on river input, and other sources such as sewage sludge and manure spreading that could directly contaminate wetlands through rainwater runoff and infiltration need to be considered to better understand the pathogen introduction and dispersion processes in wetlands (Henriot et al. 2019). However, natural wetlands are helpful in the removal of microbial pathogens and indicator bacteria as well as viruses, protozoa, and parasitic worms, with a removal rate of 78% of coliphages and up to 100% of helminth eggs (Reinosoa et al. 2008).

10.3 Remediation Barriers for Wetlands in Africa

10.3.1 Threats Facing Wetlands in Africa

Wetlands are biodiversity hotspots with environmental, social, and economic significance (Mitsch et al. 2015). In Africa, wetlands have continued to be productive ecosystems that not only offer direct goods and services to the population, such as food, energy, medicine, building materials, and transportation, but they also offer indirect benefits such as water quality improvements, flood control, and erosion control (Nonterah et al. 2015; Okeyo-Owuor and Raburu 2016; Willbroad and Kiyawa 2019). However, the importance of wetlands in the provision of various goods and services has resulted in human pressure on wetland resources. The rising population will increase pressure on wetland resources. This calls for strategic actions to be put in place to sustain wetland resources. Gradually, climate change and eutrophication of wetlands as a result of the inflow of agricultural runoff and municipal wastewater have become a serious concern since they impair the functioning of wetlands.

The most important prerequisite of successful and sustainable wetland management, utilization, and preservation is information about the degree and status of wetlands in a nation or locale. However, wetland inventories in Africa are frequently inadequate, and monitoring them is uncommon. Hence, various wetland scientists have pointed out that the barriers to successful wetland studies run from information accessibility and quality to eagerness and capacity to utilize information. There is agreement that the most critical tasks are applying standard, policy-relevant markers, scaling up conventional and new management tools and conventions, and capacity and partnership building. However, the value of wetlands for ecosystem services such as environmental remediation is generally not understood (Moomaw et al. 2018). Legitimate administration of wetlands in Africa is hampered by a general lack of skills and funding. Few if any wetlands in Africa are satisfactorily overseen and managed for sustainability. A few initiatives to develop this have incorporated the geomorphic forms of wetland arrangement in a modern hereditary geomorphic classification framework (Grenfell et al. 2019). The ecosystem nature of the African streams has also contributed to the nature, distribution, and abundance of wetland organisms (Pavón-Jordán et al. 2020).

A growing human population also forces people to claim natural wetlands for agricultural purposes, leading to environmental degradation due to agronomic practices that use synthetic agrochemicals and cause siltation of water sources. Changes in land use, as well as changes in hydrology and climate, lead to further degradation and losses. Climate change leads to flooding and drought and affects plant and animal communities by bringing in new competitors that are not adapted to their new environment. More frequent incidences of drought and increased temperatures result in high evaporation and increased evapotranspiration rates affecting plants and animals. Increased temperatures result in increased microbial respiration and oxygen consumption, in turn affecting plant growth and microorganisms that contribute to bioremediation activities. Disposal of non-biodegradable materials such as plastics in natural wetlands is another bottleneck that challenges the performance of the wetlands (Akindele and Alimba 2021).

10.3.1.1 Major Threats to Lake Victoria Wetlands

Lake Victoria is the largest freshwater lake in Africa, lying mainly in Tanzania (49%) and Uganda (45%) but bordering Kenya (5%). The lake is of social and economic importance to the local communities and globally. It is mainly known as a source of fish protein for the East African communities; however, it also serves as a source of fresh water for domestic use, irrigation, and hydro-electric generation (Njiru et al. 2018). Other benefits include being a medium of transportation, tourism, employment, and dispersal of wastes. Additionally, cichlids from the lake are exported as ornamental fish for aquaria worldwide (Muli 1996).

Despite the social and economic importance of the Lake Victoria wetland, several factors have led to significant disruption of its ecosystem since the 1920s. The most regrettable one among the scientific community has been the introduction of alien fish species, notably, Nile perch (*Lutes niloticus*) and Nile tilapia (*Oreochromis niloticus*), which has led to a reduction of fish stocks, including species of particular interest like the ornamental cichlids (Muli 1996). Nutrient enrichment of the lake as a result of the inflow of domestic and industrial sewage, and agricultural effluents has also exacerbated the growth of water hyacinth (*Eichornia crassipes*), die-off of fish, and zones with hypoxic conditions (Okeyo-Owuor and Raburu 2016; Willbroad and Kiyawa 2019). Siltation of the lake due to deforestation of the watershed for settlement and agricultural production has also been on the rise (Okeyo-Owuor and Raburu 2016).

10.3.1.2 Major Threats to the Niger Delta

The Niger Delta makes up 7.5% of Nigeria's land mass, and it is located where the Niger River discharges into the Atlantic Ocean. Mangrove forests in the coastal region of the Niger Delta occupy 17% of the total landmass of the delta (Mmom and Arokoyu 2010). The delta is petroleum-rich and is mainly known for oil exploitation (Ordinioha and Brisibe 2013). Other economic activities within the delta are fishing, and commercial agriculture (oil palm) (Mitchell 2013). The delta has gradually undergone degradation due to human-induced factors. Human activities affecting the delta include dam construction upstream of the Niger River, which has resulted in a reduction of sediments reaching the delta by an estimated 70%, proliferation of the water hyacinth (*Eichhornia crassipes*) in the delta due to inflow of wastewater, deforestation, and harvesting of juvenile fish. Of great concern recently has been the pollution of the delta by oil spills due to oil exploration, which poses severe threats to both humans and aquatic life (Ordinioha and Brisibe 2013). Climate change has resulted in a reduced amount of rainfall received in the Niger Delta. The resultant effect has been reduced flow in the Niger Delta, reducing fishery yields (Oguntunde and Abiodun 2013).

10.3.1.3 Major Threats to the Zambezi Delta

The Zambezi Delta is situated in Mozambique at the downstream terminus of the Zambezi River, where it discharges into the Indian Ocean. The Zambezi is southern Africa's most extensive river system (Moore 2017). The delta is the habitat for the largest diverse concentration of wildlife found in African floodplain systems, in addition to supporting a diverse mosaic of grassland, palm savannah, woodland, and mangrove communities (Beilfuss et al. 2016). The delta is not only rich in biodiversity but provides a range of goods and services that is important for the local, regional and national economy of Mozambique (Beilfuss et al. 2016). These ecological services include, among others, provision of forest and woodland products, grazing lands for livestock, nutrient-rich lands for flood recession agriculture, floodplain freshwater fisheries, freshwater for domestic use, carbon sequestration by woody species, storm surge and coastal erosion protection, flood storage and mitigation, and wildlife for sustainable meat supply.

The Zambezi delta continues to face several threats despite conservation efforts. The key challenges reported (Beilfuss et al. 2000) include reduction of sediments reaching the delta due to the construction of dams for hydropower generation along the river, eutrophication of floodplain waterways from agro-industrial drainage and pollution from commercial sugar cane farming, reduction of biodiversity due to uncontrolled fires, and deforestation.

10.3.2 Gaps and Opportunities for Improved African Wetlands in Remediation

Many issues hamper the understanding and management of wetlands in Africa, including limited knowledge about taxonomy to determine the abundance and identity of plant communities in wetlands and the occurrence of endemic as well as exotic and invasive species that may pose a risk to wetlands biodiversity. There further is a lack of remediation technologies as well as legislation in the area of groundwater pollution management in African countries (Gaye and Tindimugaya 2019).

Bioremediation alone is often not the complete solution to pollutant clean-up, for example, because it may require more space than is available. However, it is an under-utilized approach that can often be incorporated at different stages of treatment, in combination with engineered physical and chemical treatments. Knowing the limitations and how to address those must be included in any remediation work (Salimi et al. 2021). In addition, all stakeholders must be provided with all information to alleviate concerns that the public may have. Such concerns arise from a general lack of understanding of how the natural processes work, in combination with the belief that somehow methods engineered by humans are better than natural processes. This is also because people live more and more removed from nature, often in cities made from concrete and full of artificial smells. Ignoring people's lack of education about wetlands and their strong convictions, traditional, cultural, or historical, makes the widespread application of wetlands in remediation difficult, not just in Africa but worldwide.

Wetland macrophytes produce high amounts of biomass, and pollutants accumulate on the roots and may be taken up quickly. Stabilizing plant tissue after dieback to avoid recycling accumulated pollutants may be necessary, particularly if the wetland is connected to other waters. The vegetation may be harvested and used as biofuel or incorporated into building materials. Bottomland hardwood forests can also inhabit well-drained alluvial soils, while lower-elevation marshes can be organically rich with peat layers. Plume convergence with surface water flows could occur further down-gradient, out of the expected wetland remediation zone. Conversely, planted stands of poplar and willow trees have been shown to alter subsurface flow fields, creating a significant degree of plume containment (Kadlec and Reddy 2001).

Wetlands are experiencing immense pressure from human activities such as agriculture and settlement, excessive exploitation by local communities, and improperly planned development activities. Moreover, challenges that may limit the application of wetlands in Africa include plant sensitivity to climate and seasons, lack of resistance to toxic compounds, and suspended solids. For instance, the most commonly used chelates for phytoextraction can increase the toxicity of groundwater and affect soil microfauna; therefore, environmental-friendly forms of chelates need to be developed (Ghori et al. 2015). Another strategy to detoxify soils is to create transgenic plants with increased hyper-accumulation activity against a particular pollutant (Gunarathne et al. 2018). Such plants improved execution concerning the digestion system of trichloroethylene and the expulsion of a range of other poisonous unstable natural pollutants, including vinyl chloride, carbon tetrachloride, chloroform, and benzene, suggesting that transgenic plants may be able to contribute to the more extensive and more secure application of phytoremediation (Gunarathne et al. 2018).

Another concern is that the use of wetlands for remediation purposes may also lead to the degradation of the wetlands themselves. It is not acceptable to solve one problem (removal of pollutants) while creating another (degradation of wetlands). Therefore, advancing remediation using wetlands in Africa requires strategies that improve pollutant removal while advancing environment rebuilding. Restoration and creation of wetlands in Africa may achieve both enhancements to the general environment as well as the removal of pollutants (Cavicchioli et al. 2019). This can be achieved not by using existing natural wetlands to treat polluted water but by incorporating modifications on the natural ones specifically for that purpose. It would expand, not reduce, the total area of wetlands, with the benefits of providing additional ecosystem services other than improving water quality, such as creating habitat for plants and animals, and opportunities for recreation.

Human activities posing threats to the existence of wetlands in Africa can be mitigated by a stronger political will to protect them. It should be based on sound wetland policies and communities' engagement in the wetlands' management. Formulating policies by African governments in accordance with the recommendations from the Ramsar Convention will be a major step in demonstrating their commitment to protecting endangered and fragile wetlands and those outside protected areas. Recently, the government of Uganda launched a policy for conserving wetlands in protected and non-protected areas as proposed by the Ramsar Convention which is an excellent example of political goodwill to conserve wetlands and their biodiversity (Willbroad and Kiyawa 2019). Guinea Bissau, with the assistance of the World Conservation Union (IUCN), is currently implementing an Integrated Coastal Zone Planning for safeguarding coastal wetlands (Lopes et al. 2021). The examples from Uganda and Guinea Bissau demonstrate Africa's commitment to conserving wetlands, their biodiversity, and other functions.

10.4 Conclusions and Recommendations

To address the challenges associated with establishing nature-based solutions to remediate environmental damage, such as the use of wetlands for improvement of water quality, joint efforts are needed, including transformation to a green economy, innovation technologies, improving operations and maintenance, harvesting energy, improving governance and management, and promoting public participation by communities, as well as establishing and ensuring water quality standards are adhered to. Successful bioremediation will depend on ecosystem services that the wetland can be exploited for. The use of the plants will depend on what the community desires. For example, plants like *Typha* species and *Papyrus* can be utilized for making baskets, ropes, and all types of mats. Other plants can be used for generating timber for the construction of buildings and furniture. Thus, suitable plant selection and application of appropriate agronomic techniques are paramount in promoting the use of wetlands in bioremediation within Africa. In most cases, locally available and environmentally adapted native plants are the most suitable phytoremediators. Most of the plant biomass obtainable from the macrophytes can be used for the production of biofuel, biogas, and biochar, among other uses. Some bioremediation processes can also help in the recovery of precious metals such as nickel and copper that have been realized in water and sediments within some African Aquatic ecosystems. Moreover, many African wetlands are sanctuaries for a wide range of flora and fauna; in that connection, they can serve in their conservation and nature attraction.

Although wetland technology is an economical and environmentally sound approach as a wastewater management option, its adoption still faces challenges in Africa. Notable challenges are lack of awareness of the technology, lack of technical expertise, limited spaces, especially in urban areas, the existence of few policies that promote sustainable wetland management and conservation, poor understanding of constructed wetland potential, and lack of crucial data and information about the state of wetlands in African countries. Therefore, for effective ecological use of African wetlands, the following recommendations are suggested for promoting and applying them in bioremediation:

- There is a need for research institutions to collaborate with industries to promote the full-scale application of research findings on constructed wetlands.
- Education and advocacy concerning integrated implementation of both natural and constructed wetlands should be pursued.
- There is a need for increased financial support for research, monitoring, and training of wetland specialists to understand the appropriate designs and treatment mechanisms occurring in constructed wetlands.
- Constructed wetlands should be integrated into management plans for conserving a watershed's soil and water resources or as part of major restoration projects.
- The performance of constructed wetlands should be monitored, and information about their state should be shared with the public or relevant authorities to advocate for their adoption.
- Conservation strategies that better integrate the priorities of indigenous communities to avoid conflicts should be encouraged.

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Chapter 11
Cost and Benefits of Treatment Wetlands in the Tropics

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Abstract Treatment wetlands have proven to be an effective technology for treating wastewater in tropical regions due to their capacity to treat a wide range of polluted waters from different origins and compositions. Additionally, treatment wetlands can provide ancillary benefits beyond the removal of pollutants, including leisure, education, conservation, and climate change mitigation. Depending on the context where the technology will be implemented, treatment wetlands may demand similar or higher capital cost investments compared with other wastewater treatment technologies, but with lower operation and maintenance costs; this represents an opportunity for low-income countries and regions in the tropics, especially if the cost structure is analyzed in the long term, where the treatment wetland technology is more cost-efficient in comparison with conventional mechanical-based treatment systems. Treatment wetlands integrated into the local landscape can become biodiversity sanctuaries, and besides the water reclamation and reuse, the systems can produce plant-based, high-value products, integrating into a circular economy model. In this chapter are reviewed the different components of the cost structure of treatment wetlands, are presented examples of the capital, operation, and maintenance cost of treatment wetlands vs. conventional treatment technologies, and are addressed the environmental, and ecosystemic benefits of implementing treatment wetland technologies, and the side potential economic benefits of plant biomass use.

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11.1 Introduction

For more than a century, wastewater management has been based on the operation of sewage networks to collect the wastewater generated in populated areas and transport it to centralized facilities to be treated before its final discharge into surface waters or the sea. This concept has been effectively implemented mainly in Europe, the USA, and other developed countries, but it faces challenges when it is intended to be implemented in developing countries with economical and technical limitations, as well as with scenarios of rapid urban growth and water-scarcity (Maurer et al. 2005). Latin America and the Caribbean countries and many others in tropical regions around the world fall under this category. Latin America and the Caribbean is the most urbanized region in the world, with an estimated population of 650 million inhabitants that produces more than 30 km^3 of wastewater each year, only counting the urban settlements (FAO 2017).

According to Hernández-Padilla et al. (2017), in Latin America and the Caribbean, only 20% of the produced wastewater receives adequate treatment before discharge into nature, with the resulting detrimental effects on the receiving waters. As a consequence, in the region, up to one-third of the rivers and one-seventh of the total length of them are polluted (UNEP 2016). Some of the reasons for the low wastewater treatment efficiency and coverage in developing countries are associated with the adoption of technological options that involve complex and expensive systems, that imply high capital and operation and maintenance costs. For example, mechanical-based wastewater treatment systems such as the activated sludge process, typically implemented in most centralized sewage systems and although effective in terms of treatment, demand high operation and maintenance resources. Plant personnel must be specialized to guarantee the proper operation of the system, and the systems require a high-power supply both in quantity and continuity to guarantee successful treatment (Kumar and Tortajada 2020; Tchobanoglous et al. 2003).

If the operators of technically complex wastewater treatment facilities do not have the financial muscle and the adequate knowledge to administrate these systems, their operation and maintenance become neglected, consequently endangering the system's sustainability in the long term. Therefore, the implementation of a technological option for wastewater treatment must be based on the context of where it is going to be implemented and should consider aspects such as efficiency, reliability, sludge disposal, land requirements, environmental impacts, capital costs, operation and maintenance costs, sustainability, and simplicity (von Sperling and de Lemos 2005). In 1997, the U.S. Environmental Protection Agency (EPA) performed a contextual cost comparison of centralized vs. decentralized wastewater management for a hypothetical 135-home rural community. The study revealed that decentralized systems are generally more cost-effective for managing wastewater in rural areas than conventional centralized wastewater treatment systems where the transport distance via sewer is more than 5 miles (EPA 1997).

In many developing countries, like those found in Latin America and the Caribbean region, the budget allocated to the maintenance of the public sewage and wastewater treatment systems is surpassed by population growth and maintenance needs, therefore, most wastewater treatment systems are either neglected or even abandoned. According to Rodriguez-Dominguez et al. (2020), some Latin American and the Caribbean countries reported high coverage of wastewater treatment, but most of these systems do not work properly, or they are not working at all, mostly a consequence of the high operational and maintenance cost. Then, it is clear that in regions where technical and mechanical treatment technologies cannot be effectively maintained, the implementation of less energy- and maintenance-intensive wastewater technologies could contribute to improve water quality, redounding in better local and regional environmental conditions, as well to uplift health conditions of the region (Arias and Brown 2009).

Treatment wetlands, also known as constructed wetlands, phytoremediation systems, reed beds, soil infiltration beds, engineered wetlands, root systems, or biofilters (Carvalho et al. 2017), is a nature-based solution where the natural processes occurring in natural wetlands are emulated and optimized through engineered designs to improve water quality (Rodriguez-Dominguez et al. 2020). Treatment wetlands are effective due to the interaction of three primary elements: (1) water, (2) plants, and (3) media (soil, gravel, sediment, etc.) (Kadlec and Wallace 2009). The combination of treatment wetlands' water depth, influent-effluent disposition, flow rate, aspect ratio, and plant spacing and density determines the hydraulic performance (Guo et al. 2019). Treatment wetlands are hydraulically characterized by having mostly laminar flow and are a highly productive ecosystem, where the organic matter and pollutants are transformed by the symbiotic interaction of biofilm, plant biomass, and media to elemental compounds such as CO_2, CH_4, and N_2O, removing pollutants and therefore producing an effluent of better quality.

Treatment wetlands have been extensively evaluated for their capacity to improve the water quality of different concentrations and origins, and among others have shown to be effective for treating wastewater of domestic origin (Brix and Arias 2005; Konnerup et al. 2009; Molle et al. 2005; Peñacoba-Antona et al. 2022; Vymazal 2011; Wallace 2015), urban run-off (Malaviya and Singh 2012; Istenič et al. 2011), slaughterhouse origin (Gutiérrez-Sarabia et al. 2004), industrial origin (Afzal et al. 2019b; Maine et al. 2019; Tara et al. 2019), and urban sewage (Afzal et al. 2019a; Stefanakis 2019). Treatment wetlands have relatively low establishment costs, are robust, can be easily operated and maintained, and have a high potential for application in developing countries (Molle et al. 2005). The technology is proven, and they seem to be an adequate option for treating wastewater in Latin America, the Caribbean, and tropical regions, where the existing conditions with mostly warm temperatures, extensive light radiation periods, and available land can even enhance treatment wetlands performance (Machado et al. 2017).

In this chapter, the costs and benefits of treatment wetlands systems will be described, with particular emphasis on the importance of the technology for countries in the tropics and warm regions, where construction, maintenance, and operational costs are issues that may limit the establishment and operation of efficient wastewater treatment. Then, the use of effective and less expensive wastewater treatment systems can improve life quality and ensure an environmentally sustainable wastewater treatment of polluted water before they are discharged into the environment.

11.2 Cost Structure of Treatment Wetlands

Treatment wetlands for wastewater treatment have been regarded as one of the best sustainable alternatives to conventional wastewater treatment systems due to their low global cost and simple maintenance requirements, facts that constitute them as a good investment regarding performance for value return (Arias and Brown 2009; Zhang et al. 2015). The cost structure of treatment wetlands is based on two general components: capital costs, and operation and maintenance costs (Dotro et al. 2017).

The **capital costs** are those related to the establishment of the treatment wetlands system and can be grouped into four main factors: (i) land acquisition (which can be a significant fraction of total capital cost), (ii) installation of pumping, aeration, and/or recirculation systems (depending on treatment wetlands design), (iii) the availability of local building materials, and (iv) labor force. Even though the associated capital costs for the construction of treatment wetlands could be within the same order of magnitude as conventional wastewater treatment technologies (Dotro et al. 2017), these costs have regional variations that depend on the particularities of each market and differences can even be found when comparing worldwide geographical locations (Kadlec and Wallace 2009).

The **operation and maintenance cost** can be grouped into four main factors that include: (i) monitoring of the operation and performance of the systems, (ii) general maintenance of the system, surrounding areas, and electrical/mechanical components (if required), (iii) vegetation management, and (iv) energy supply (if systems include assisted aeration or require pumping). Treatment wetlands are typically passive technologies, with operation and maintenance costs related to electricity and chemicals drastically reduced and with almost negligible costs of sludge management and specialized workforce. In general terms, treatment wetlands require less operation and maintenance efforts in comparison with conventional wastewater treatment facilities, with variations by a factor of 2–10 (Dotro et al. 2017; Kadlec and Wallace 2009).

It is not possible to establish a universal guideline for the cost structure of treatment wetlands, given the geographical variations, selection of materials, design characteristics and variations, legal restrictions, and other intrinsic variables of the regional markets, however, there are some common cost components that a treatment wetlands system includes (Kadlec and Wallace 2009; Wu et al. 2011). A broader explanation of the main items of the cost structure of the implementation of treatment wetlands for wastewater treatment is summarized in Table 11.1.

Table 11.1 Summary of components of capital and operation and maintenance cost of treatment wetlands

Cost type	Concept	Description
Capital	Land acquisition	Generally is the highest capital cost, due to the high land surface demand of treatment wetland facilities
	Site evaluation	Cost associated to the characterization soil, groundwater elevations, and topography of selected site for construction of treatment wetland facilities
	Earthworks	These includes excavation and soil grading to create basins of appropriated size according to wetland design, enclosing with earthen berms, and clearing and grubbing undesirable vegetation, and obstacles. This cost varies according to the complexity of the project, in terms of surface area n required, and site topography
	Construction materials	Cost associated to liners (synthetic, clay, etc.), support media of appropriate granulometry, pipes, pumps (if required), aeration systems (if required), plants, transportation, and labor on site
	Hydraulic structures	Construction/implementation of inlet distribution, perforated piping of appropriate size, water level control structures, and outlet collection, final discharge structures
	Site work	Construction of access roads and construction facilities, fencing of working areas, installation of electrical power supply (if needed), and erosion control/surface restoration (if needed), landscape beautification
	Auxiliary infrastructure	Construction of complementary structures (e.g. parking areas, visitors centers, observation sites, walking paths) when the system has an added ecosystem benefit value in terms of environmental education, cultural, aesthetic, leisure or ecosystemic
	Indirect costs	These include engineering-related items (e.g. conceptual and final design, preparation of blue prints and material characteristics and specifications, preparation of operation and maintenance manuals, etc.), non-construction contractor costs (e.g. insurances, construction surveying and staking, traffic control, etc.), construction observation and start-up services (e.g. inspections, testing, start-up assistant and operation training), and contingency and escalation costs (to cover costs due to human judgment errors, and allowance for inflation), environmental studies
Operation and maintenance	Monitoring	These include compliance monitoring (external sampling and analysis) to be presented to environmental authorities, as well as for internal use of operators to know the operation state and performance of the system
	General maintenance	These include the maintenance of access roads and berms; maintenance, repairing or replacement of hydraulic/mechanical components; sludge removal from pre-treatment units
	Vegetation management	Cost associated to vegetation harvesting, litter removal, plague control, and replanting (if necessary)
	Energy	Cost associated to operation of pumping systems (if needed by the treatment wetlands system)

Based on Dotro et al. (2017), Kadlec and Wallace (2009) and WSP-LAC (2008)

11.2.1 Costs of Treatment Wetlands Reported Around the World

As mentioned in previous paragraphs, treatment wetlands are very competitive in terms of capital costs and are frequently very advantageous in terms of operation and maintenance costs, compared with conventional wastewater treatment systems (Langergraber et al. 2019). The cost structure of treatment wetlands is based on the same basic working principles of the technology, no matter where these are implemented. However, at the regional level, the intrinsic conditions of the market, legislation, and general environmental characteristics have an impact on the cost structure, making it almost impossible to determine standardized values worldwide. Regarding this, Gunes et al. (2011) estimated the prices of land according to the location in the United States; for the installation of treatment wetlands in low-density rural regions, the values are around US\$0.3/m^2 and can reach up to US\$10/m^2 for land planned to be urbanized. When comparing countries with similar social-economic development, also can be identified regional variations that have an impact on the cost structure of treatment wetlands. For example, the capital cost for implementing a horizontal flow wetland is around US\$86/m^2 in the USA, whereas, in countries like Portugal, Italy, and Belgium, the average capital cost is US\$123/m^2, US\$161/m^2, and US\$331/m^2, respectively.

DiMuro et al. (2014) investigated the replacement cost and life cycle assessment of treatment wetlands built instead of a sequencing batch reactor to solve a regulatory compliance issue in meeting suspended solids requirements for a wastewater treatment system at the Union Carbide Corporation (a subsidiary of The Dow Chemical Company) plant in Seadrift, Texas. The financial results indicated that the total net present value savings calculated for implementing the treatment wetlands instead of the sequencing batch reactor were US\$282 million over the project's lifetime (2014 prices). The life cycle assessment demonstrated that the lower energy and material inputs to the treatment wetlands resulted in lower use of fossil fuel and consequently less acidification, smog generation, and ozone depletion and likely led to lower impacts for global warming and marine eutrophication. Wu et al. (2011) compared the cost of hypothetical integrated household treatment wetlands with different conventional alternative technologies analyzed by EPA (1997), finding that the total capital cost and the annual operation and maintenance were US\$34,965 and US\$700, respectively. These values represent less than 10% of the total capital and annual operation and maintenance of the technologies named "alternative small-diameter gravity sewers," and "collection and small on-site systems," and less than 1% of the costs of centralized systems.

Parde et al. (2021) reported a capital cost of US\$446/m^3 (inflow to be treated) for free water surface wetlands, US\$578/m^3 for vertical flow wetlands, US\$1434/m^3 for horizontal flow wetlands, and US\$1047/m^3 for combined treatment wetlands system. Despite treatment wetlands being more demanding in terms of surface area in comparison with other conventional technologies, it is still a low-cost treatment alternative, especially in terms of operation and maintenance. The operation and maintenance cost of moving bed biofilm reactor represent 20–25% of their capital cost; for activated sludge process, trickling filter, up-flow anaerobic sludge blanket reactor, and stabilization ponds, the operation and maintenance represent 10–15% of their capital costs; for sequential batch reactor and a duckweeds systems, the operation and maintenance represent 3–5%; whereas, for treatment wetlands, the operation and maintenance represent 1–2% of the capital cost. Liu et al. (2009) reported that in China, the construction cost of treatment wetlands was between US$164 and US$460/m^3 (inflow to be treated), which is one-third to half of the capital cost of a mechanical wastewater treatment system, that ranged from US$246 to US$657/m^3. Regarding operation and maintenance costs, the values were extremely low for treatment wetlands (US$0.01–US$0.04/m^3) compared with mechanical-based wastewater treatment systems (US$0.12–US$0.25/m^3).

The differences in costs can also be found among treatment wetlands technologies. Tsihrintzis et al. (2007) reported that the costs for a free water surface wetland compared to a vertical flow wetland for treating domestic wastewater in Crete can be 30% lower. Their calculation shows a construction cost of US$364/PE (PE: population equivalents) for the free water surface treatment wetland, while for the vertical flow wetland the cost amounted to US$521/PE. The same is true for the operation and maintenance costs, Tsihrintzis et al. (2007) reported that the total operation and maintenance cost of the free water surface wetland system was US$0.04/m^3 (inflow to be treated), while the vertical flow wetland system had a cost of US$0.14/m^3, which is 3.6 higher than for the free water surface flow wetland.

11.2.2 Costs of Treatment Wetlands Reported in Warm and Tropical Regions

The performance of treatment wetlands depends on the environmental conditions of the establishment site. In warm-climate regions, the high sunlight irradiance and high environmental and wastewater temperatures might promote the conditions for the growth of vigorous vegetation and the development of active and abundant microbial communities that accelerate most biochemical reactions and physical processes (Varma et al. 2021). This could be advantageous in aspects related to the wetland design and can impact the cost structure, given that, for reaching a demanded effluent quality, the footprint could be smaller, or for a given surface area, the removal efficiencies are expected to be higher allowing treatment wetlands systems with higher inlet pollutant loads, therefore diminishing the surface area required for its implementation leading to a decrease in the capital and operation and maintenance costs (Alarcón et al. 2018; Langergraber et al. 2019).

Table 11.2 presents a compilation of study cases that show the differences among conventional wastewater treatment systems and treatment wetlands in terms of capital, operation, and maintenance costs in warm and tropical regions, as well as depict some variations that exist even among regions with similar

Table 11.2 Summary of average cost (capital, and operation and maintenance) of conventional and nature-based technological solutions (ponds/lagoons, and treatment wetlands) for wastewater treatment in warm regions

Technology	Type of system	Location	Surface (m^2/PE)	Power requirement	Capital cost (US$/PE)	O & M costs (US$/PE-year)	Reference
Conventional	Activated sludge	Brazil	0.19	Yes	85.05	13.00	von Sperling and de Lemos (2005)
	Activated sludge*	Colombia	0.19	Yes	108.42	18.28	Castaño-Herzig and Ramirez-Vargas (2008)
	Activated sludge - extended aeration	Brazil	0.19	Yes	69.06	13.00	von Sperling and de Lemos (2005)
	Trickling filter	Brazil	0.21	No	89.37	9.75	
	Trickling filter *	Colombia	0.20	No	167.07	14.47	Castaño-Herzig and Ramirez-Vargas (2008)
	Rotating biological contactor	Brazil	0.15	No	89.37	9.75	von Sperling and de Lemos (2005)
	Septic tank + anaerobic filter	Brazil	0.28	No	64.99	6.50	
	Septic tank + infiltration	Brazil	1.13	No	52.81	3.25	
	Sequencing batch reactor	Brazil	0.14	Yes	69.06	13.00	
	Sequencing batch reactor*	Colombia	0.59	Yes	33.49	5.45	Arias and Brown (2009)
	UASB reactor + anaerobic filter	Brazil	0.10	No	40.62	3.57	von Sperling and de Lemos (2005)
Ponds/lagoons*	Anaerobic pond + facultative pond	Brazil	2.10	No	34.12	2.44	von Sperling and de Lemos (2005)
	Anaerobic pond + facultative pond*	Colombia	2.42	Yes	17.19	1.60	Arias and Brown (2009)
	Anaerobic + facultative + maturation ponds	Brazil	4.00	No	48.75	3.25	von Sperling and de Lemos (2005)
	Facultative aerated lagoon	Brazil	0.38	Yes	44.69	5.69	
	Facultative ponds	Brazil	3.00	No	36.56	2.44	
	Facultative pond*	Colombia	2.02	No	500.26	14.51	Castaño-Herzig and Ramirez-Vargas (2008)

Treatment wetlands	Horizontal flow	Brazil	4.00	No	40.62	2.44	von Sperling and de Lemos (2005)
	Horizontal flow*	Central America	1.42	No	105.89	5.24	WSP-LAC (2008)
	Horizontal flow*	Colombia	2.42	Yes	17.59	1.67	Arias and Brown (2009)
	Horizontal flow*	Colombia	0.55	No	217.88	4.07	Castaño-Herzig and Ramirez-Vargas (2008)
	Vertical flow*	Greece†	10.23	Yes	864.58	13.28	Tsihrintzis and Gikas (2010)
	Free water surface*	Greece†	7.90	Yes	752.62	10.39	
	Horizontal flow*	Spain†	3.60	Yes	1035.48	119.40	Puigagut et al. (2007)
	Vertical flow*	Spain†	2.90	Yes	607.29	57.64	
	Combined systems*	Spain†	4.80	Yes	452.89	160.57	
	Combined systems*	Nepal	1.68	No	117.60	1.69	UN-HABITAT (2008)
	Vertical flow*	Nepal	1.45	No	103.35	n.a.	

Costs are given in 2022 US$

PE Population equivalents, *Including pre-treatment units, †Mediterranean conditions social-economic conditions. Brazil is a classic example of the differences that exist in the cost structure of conventional wastewater treatment systems and treatment wetlands; the capital cost of treatment wetlands is around US$40/PE, which is in a similar range to other natural-like systems based on lagoons but lower than conventional treatment technologies like activated sludge (US$69–85/PE), sequential batch reactors (US$69/PE), or septic tanks with complementary units (US$53–69/PE). Also, that trend can be observed in operation and maintenance costs, where treatment wetlands have an average value of US$2.44/PE-year, which is similar to other nature-based treatment systems but much lower than other conventional technologies.

The case of Colombia is an interesting example to show the local differences that may exist in the cost structure of treatment wetlands in comparison with other conventional technologies, as well as the effects of the economy of scale. Arias and Brown (2009) reported that the capital cost of the treatment wetlands evaluated was around US$17.6/PE, similar to the per capita value of one of the alternative systems used in the comparison (US$17.2/PE for a combination of anaerobic and facultative lagoons), and much lower than a system based on a sequencing batch reactor technology (US$33.5/PE); the operation and maintenance costs followed a similar trend, with average values of US$1.7, US$1.6, and US$5.5/PE-year; based on a global analysis that considers the performance, resource investment, and cost, the researchers concluded that the use of treatment wetlands is a promising alternative that should be considered for wastewater treatment in Bogotá Savannah. Whereas Castaño-Herzig and Ramirez-Vargas (2008) in a study carried out in the coffee triangle of Colombia, reported that the average capital cost of treatment wetlands (US$218/PE) is lower than other nature-based alternatives such as facultative ponds (US$500/PE) but higher than conventional treatment systems analyzed in the comparison (trickling filter – US$167/PE; activated sludge – US$108/PE); that trend in the capital costs is a consequence of the high cost of land, earthworks, and filter material in the study zone. In the long term, those costs are compensated by the low operation and maintenance costs of treatment wetlands (US$4/PE-year), in comparison with the same costs for the other treatment systems analyzed (US$14–US$18/PE-year), making the treatment wetlands technology the most favorable in the long term based on a cost-effectiveness analysis.

Based on the literature and studies mentioned in this section, it is possible to conclude that exists an effect of the sizing of treatment wetlands systems on their costs/benefits structure. Indeed, systems with larger surface areas benefit from economies of scale in comparison with conventional treatment systems (Kadlec and Wallace 2009). This effect is especially seen in systems designed for small populations, (i.e., <1000 PE), but for larger systems (e.g., >8000–10,000 PE), this advantage becomes limited due to higher land requirements compared to other conventional systems (Stefanakis et al. 2014). An example of the effect of economies of scale can be seen in Fig. 11.1.

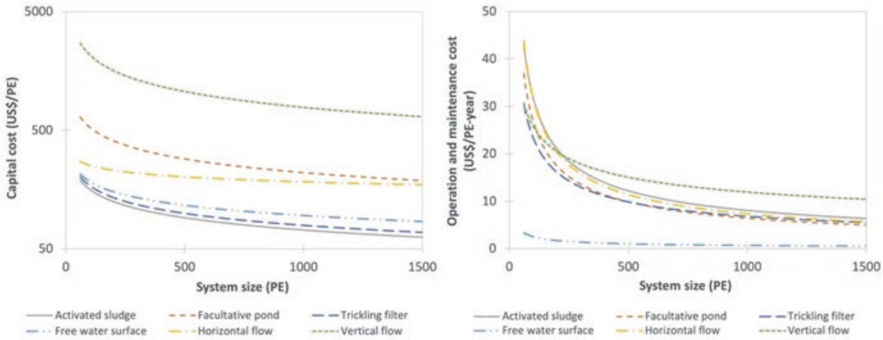

Fig. 11.1 Impact of the economy of scale based on system size (population equivalents – PE) over capital cost (left), and operation and maintenance cost (right) for different wastewater treatment conventional options, and treatment wetlands variants (free water surface; horizontal flow; vertical flow). Costs are given in US$ for the year 2022. The figure is based on information from Castaño-Herzig and Ramirez-Vargas (2008); Salas-Quintero et al. (2007); Stefanakis et al. (2014); von Sperling and de Lemos (2005)

11.3 Treatment Wetland Benefits

Nature-based technological solutions for wastewater treatment, as in the case of treatment wetlands, provide a range of benefits that surpass the intrinsic objective of pollution removal. These benefits fall in the category of ecosystem services (provisioning, regulatory, and cultural services) and go beyond the classical valuation in terms of capital and operation and maintenance costs.

11.3.1 Treatment Wetlands' Robustness for Pollutant Removal

The performance of anaerobic and aerobic reactors is affected by changes in external factors such as temperature, loading rates, and inlet water quality. In anaerobic reactors, the changes cause accumulation of volatile fatty acids, changes in pH and alkalinity that affects biogas production and composition, or create a sludge washout (Leitão et al. 2006). In aerobic reactors, for example, sequencing batch reactor or moving bed biofilm reactor, the TN removal capacity decreases when temperatures drop from 30 °C to 18 °C (Lackner and Horn 2013), however, treatment wetlands seem to manage better changes in the external conditions.

Compared with other wastewater treatment technologies, treatment wetlands have demonstrated to be a robust technology, able to keep its capacity for pollutant removal even after drastic changes in temperature, flow rates, salinity, and pollutant loads, among others. Paing et al. (2015) evaluated 169 different French vertical flow wetlands for 12 years and found that around 97% of the analyzed samples at the outlet of the plants were fulfilling the French discharge limits for organic matter (125 mgCOD/L and 25 mgBOD$_5$/L). Additionally, the research showed that the removal efficiencies were not affected by changes in hydraulic and organic loads, temperature, or lack of proper maintenance, and if the system had been established for two years or more, the performance was consistent. The survey confirms the good performance and the robustness of the French vertical flow wetlands. In another experiment, Masi et al. (2007) demonstrated that treatment wetlands located in remote areas in Italy were not sensitive to peak flows, maintaining their capacity for treating pollutants, achieving COD reductions between 83% to 95%, and pathogen indicators removal from 3 to 5 logs during the 2 years study.

11.3.2 Positive Effect of Treatment Wetlands on the Local Biodiversity

Classically, treatment wetlands have been designed to be used as a wastewater treatment system, however, the presence of plants and freshwater (in the case of free water surface wetlands) provides a suitable environment for enhancing biodiversity and creating new habitats where a wetland has not been established to promote and sustain wildlife. The largest treatment wetlands reported were constructed in Oman, in the middle of the desert, for treating co-products from oil production. In addition to the water quality improvement, the site has become an "oasis" in the desert, providing valuable habitat for bird life (>117 species identified at the site) (Al-Rawahi et al. 2014). Hsu et al. (2011) evaluated the biodiversity of integrated free water surface wetland in subtropical Taiwan by analyzing water quality, habitat characteristics, and biotic communities of algae, macrophytes, birds, fish, and aquatic macroinvertebrates in the treatment cells. Additionally, to achieve good performance in reducing the concentrations of TN, TP, BOD$_5$, and COD, 58 bird species, 7 fish species, and 34 aquatic macroinvertebrate taxa were recorded, and it was observed that the richness, abundance, and diversity of birds, fishes, and aquatic macroinvertebrates increased in the wetland area.

Wiegleb et al. (2017) reviewed 21 scientific papers related to treatment wetlands, finding conclusive results regarding increased bat activity, associated with the increase of insects over the wetlands, resulting in more food available for the bats. Additionally, they reported an increase of birds in numbers and species including rare birds, invertebrate richness, more plant species compared with reference sites, and a higher α-, β- and γ diversity of macroinvertebrates. However, even though treatment wetlands can provide potential beneficial habitat for many species, the purpose of the establishment is not ecological enhancement but water pollution management. The presence of fauna and flora can affect the removal processes as well as affect the endemic flora and fauna, given that some key characteristics of engineered ecosystems vary from natural wetlands, including some fundamental ecological processes. Without proper management, or selection of plants according to the local species, treatment wetlands can promote the expansion of invasive, unwanted biological material and can even become a form of an "ecological trap" for native species (Zhang et al. 2020).

The "best practice" criteria related to biodiversity protection are not regularly applied during the construction and monitoring stages of treatment wetlands (Wiegleb et al. 2017). Management options, such as basin-wide integrative management and building the treatment wetlands resembling the natural ones, can partially offset the adverse impacts of treatment wetlands on the surrounding biodiversity. Also, other initiatives to avoid the potential negative impacts on the surrounding biodiversity should be concentrated on defining strategies on how to balance the interests among different stakeholders, for example, among wastewater managers and conservationists (Zhang et al. 2020).

11.3.3 Treatment Wetlands as Carbon Sinks

Due to rapid human urban development, natural wetlands are being destroyed. Treatment wetlands emulate the functions of those natural wetlands, however, their use for sequestrating carbon is yet to be explored (Rosli et al. 2017). Treatment wetlands can be either a sink or a source of CO_2 depending on the time scale of research and the environmental and management conditions involved (De Klein and Van der Werf 2014). Treatment wetlands' biomass could be considered a sink of CO_2, but also it is well known that wetlands also produce substantial amounts of greenhouse gasses like CH_4 and N_2O, due to the anaerobic, nitrification, and denitrification processes occurring there (Rodriguez-Dominguez et al. 2021).

According to Rosli et al. (2017), treatment wetlands are capable of sequestrating carbon like natural wetlands do. De Klein and Van der Werf (2014) evaluated the capacity of carbon sequestration of treatment wetlands located in the Netherlands, finding that, after converting the fluxes of CH_4 and N_2O to CO_2 equivalents, it was concluded that the treatment wetlands were most likely a sink of CO_2. However, many factors determine how much CO_2 can be stored or released into the environment. According to Maucieri et al. (2017), the wastewater flow and composition, the retention period, environmental conditions (like temperature and humidity), and plant species used to vegetate treatment wetlands can affect this CO_2 sink capacity. Maucieri et al. (2017) stated that CH_4 emissions in horizontal flow wetlands are higher than in free water surface wetlands, and N_2O emissions are higher in vertical flow wetlands than in free water surface wetlands. Intermittent feeding of treatment wetlands bed allows the decrease of CH_4 and the increase of CO_2 and N_2O emissions. A rise in the temperature increases the CO_2, CH_4, and N_2O emissions, and higher solar radiation increases CO_2 and CH_4 emissions. Lastly, plant presence significantly increases the CO_2 emission in comparison to an unvegetated condition in all treatment wetland types and increases N_2O and CH_4 emissions in vertical flow wetlands, however, according to the author, in the horizontal flow wetlands, plant presence significantly reduces the CH_4 emissions.

11.3.4 Treatment Wetlands as a Source of Biomass for Plant-Based Materials and Fuels

Treatment wetlands, apart from producing treated wastewater, also generate other subproducts, such as greenhouse gases and plant biomass. The balance of the greenhouse gases has been already mentioned in this chapter, but its value as a subproduct can be near zero, mainly because they diffuse and escape into the atmosphere. However, plant biomass seems to be a potential sustainable source of resources for producing a wide range of different plant-based products and fuels. The production and later use of treatment wetlands biomass is also attractive due to the high primary productivity reported for plants that grow in those systems (Vymazal 2013) and the enhanced capacity for removing pollutants if treatment wetlands are harvested (Zhu et al. 2007; Zheng et al. 2015). The use of treatment wetlands biomass as a value subproduct has been approached by different authors; Kouki et al. (2016) evaluated the potential of the treatment wetlands biomass for producing compost, Chiarawatchai et al. (2008), Vera-Puerto et al. (2021), Vera-Puerto et al. (2020), and Sandoval et al. (2019) reported the use of treatment wetlands biomass to produce different handcraft products and ornamental flowers for commercial use.

In recent years, resource recovery has developed to a more technical approach looking to recover material with higher value. Rodriguez-Dominguez et al. (2021) compared the potential of five of the most common treatment wetlands plants (*Iris pseudacorus*, *Juncus effusus*, *Phragmites australis*, *Typha latifolia*, and *Salix viminalis*) as a source of biomass to produce biocrude using a hydrothermal liquefaction process. The research showed that the treatment wetlands environment receiving polluted waters does not affect the capacity of the plants to produce biofuels using this specific technology. Additionally, Rodriguez-Dominguez et al. (2022) reported the potential of the same five treatment wetlands plants for producing high-value materials through a green biorefining platform. The results showed that treatment wetlands biomass has a high potential for producing soluble protein for animal feeding and has the potential to be used as a source of lignocellulosic material for dissolving cellulose pulp applications, which can be used for producing advanced materials, such as plant-based textiles or packing materials.

11.4 Conclusions

Treatment wetlands are a cost-effective wastewater treatment system that can offer an affordable and suitable solution for wastewater challenges in warm and tropical regions around the world. The available information provides evidence that the capital cost of treatment wetlands systems is within the same range and, in most cases, lower than conventional wastewater treatment systems based on mechanical designs. That, in combination with the lower operation and maintenance cost of treatment wetlands, creates a competitive global cost structure that, in the long term, favors the implementation of treatment wetlands under certain scenarios. The robustness, simplicity, and reliability of treatment wetlands for wastewater treatment can guarantee an efficient operation in the long term with minimal issues that promote the premature shutting-down of these systems, especially in developing countries, where the scarcity of resources and qualified workforce is common. In addition, the advantages of implementing treatment wetlands also include an increase in the quality of life for the surrounding area of the system, improving the biodiversity that is not currently evaluated in terms of economic return while improving water quality. Finally, from the perspective of a circular economy, treatment wetlands constitute a new source of materials, compounds, and resources, since the high primary productivity represents a new opportunity to produce high-value plant-based products and if monetarized can alleviate the economic burden of the investment.

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Conclusions

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Using constructed wetlands for phytoremediation in the tropics has exceptional potential as a sustainable nature-based solution, but going by the existing literature, that potential is not commonly utilized. Additionally, broad differences exist in history, culture, regulations, climate, water fluctuations, and vegetation, which suggest that although lessons can be borrowed from temperate regions, distinct approaches should be developed in the tropics.

It is essential to explore native species that grow with high productivity in natural wetlands, which could have a high capacity to accumulate or metabolize pollutants. Non-native species should be avoided because they may become invasive and disturb ecosystems of natural wetlands. Identifying new phytoremediator species that are endemic to tropical regions, as opposed to common crop species, can further support and protect the biodiversity of valuable and vulnerable tropical wetlands. In addition, it is necessary to extend studies to address the treatment of new or emerging pollutants through phytoremediation. Such studies are particularly valuable in tropical and subtropical regions because of the high diversity and availability of plants combined with the typically high human population densities and often poor economic conditions. Nature-based solutions addressing environmental problems are more viable in tropical regions.

Many examples from urban, rural, and industrial environments were discussed in this book, including the Sponge City concept. This nature-based solution could be applied to tropical and highly urbanized areas worldwide. The production of microalgae for bioremediation and biofuel may be combined with pollutant removal.

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Overall, treatment wetlands are cost-effective, offering affordable solutions for wastewater treatment in warm and tropical regions worldwide. The robustness, simplicity, and reliability of treatment wetlands provide efficient operation for the long term. The easy and efficient operation helps prevent premature shutdown, especially in developing countries, where scarce resources and a lack of qualified workforce are common. However, proper design and planning are crucial to successfully implementing the systems. For example, the correct selection of vegetation and the anticipation of the hydrological cycle regarding periods of high rainfall and droughts are crucial.

Finally, this book also highlighted some issues and gaps in knowledge associated with the use of wetlands for remediation in the Tropics, as follows:

- More examples from specific regions are needed to convince policymakers that wetlands are appropriate remediation systems.
- More information is needed about the efficacy of tropical aquatic and emergent plants in treatment wetlands, their response to different pollutants, their tolerance, and their ability to accumulate pollutants in tissues, among other variables.
- There is a general lack of collaboration among and between academics and industry worldwide, especially in the tropics.
- Many studies on treatment wetlands, particularly in the tropics, are done in small-scale mesocosms without any attempt to scale up to full-scale applications.
- There is a severe lack of education, awareness, and advocacy about the need to protect natural wetlands as well as about the many benefits of constructing wetlands, not just for remediation of pollution but also about how these may compensate for past loss of wetlands and their many ecosystem services.

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