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Review

Wetland Construction, Restoration, and Integration: A Comparative Review

Douglas J. Spieles

McPhail Center for Environmental Studies, Denison University, Granville, OH 43023, USA; spielesd@denison.edu

Abstract: In response to the global loss and degradation of wetland ecosystems, extensive efforts have been made to reestablish wetland habitat and function in landscapes where they once existed. The reintroduction of wetland ecosystem services has largely occurred in two categories: constructed wetlands (CW) for wastewater treatment, and restored wetlands (RW) for the renewal or creation of multiple ecosystem services. This is the first review to compare the objectives, design, performance, and management of CW and RW, and to assess the status of efforts to combine CW and RW as Integrated Constructed Wetlands (ICW). These wetland systems are assessed for their ecological attributes and their relative contribution to ecosystem services. CW are designed to process a wide variety of wastewaters using surface, subsurface, or hybrid treatment systems. Designed and maintained within narrow hydrologic parameters, CW can be highly effective at contaminant transformation, remediation, and sequestration. The ecosystem services provided by CW are limited by their status as high-stress, successional arrested systems with low landscape connectivity and an effective lifespan. RW are typically situated and designed for a greater degree of connection with regional ecosystems. After construction, revegetation, and early successional management, RW are intended as self-maintaining ecosystems. This affords RW a broader range of ecosystem services than CW, though RW system performance can be highly variable and subject to invasive species and landscape-level stressors. Where the spatial and biogeochemical contexts are favorable, ICW present the opportunity to couple CW and RW functions, thereby enhancing the replacement of wetland services on the landscape.

Keywords: treatment wetlands; ecological restoration; socioecological systems; coupled ecosystems; integrated landscape approach



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1. Introduction

Wetlands provide ecosystem services to a degree that is proportionately greater than their geographic extent [1]. Even so, wetlands are among our most endangered ecosystems, having been drained, filled, diked, flooded, and converted to other land uses with impunity for much of modern history [2,3]. Socioecological systems around the world have suffered from wetland loss and degradation, as manifested in deteriorating fisheries, reduced water quality, loss of coastal storm abatement, biodiversity decline, increased flood intensity and frequency, aquifer depletion, and reduced carbon storage [4]. The remaining wetlands have consequently achieved a heightened degree of protection, and efforts around the world have attempted to replace some of the wetlands we have lost.

Wetland replacement generally occurs in two broad categories, which are the focus of this review. The first is wetland construction. Constructed wetlands (CW) are typically designed to treat a particular wastewater stream (see [5–7] for representative designs). Often, but not always, CW are artificial ecosystems, in the sense that they are designed and managed primarily for wastewater treatment, and not intended to go through the adaptive cycle of succession [8,9]. CW are commonly engineered to treat particular wastes that are introduced at controlled concentrations, with carefully managed substrate, hydraulic path, retention time, oxidation, and a trophic structure. The second category, restored wetlands

(RW), differs in several respects. RW are intended to reestablish multiple ecological functions on the landscape (see [10–12] for representative designs). In the broadest sense of the term, the restoration of wetlands may involve the rejuvenation of a wetland where it once existed, creation of new wetland habitat, or enhancement of a wetland that exists in a degraded state. The goals of wetland restoration vary by project, but they are typically not focused solely on water quality improvement. For example, habitat provision, flood water retention, aquifer recharge, carbon sequestration, and cultural services are all common desirable outcomes in RW [13]. In special cases, there are also political and economic goals, as wetlands are built for the compensatory mitigation of wetland losses [14,15]. Beyond their goals and objectives, RW differ from CW in that they are intended as ecosystems that self-organize, respond to disturbance, and change through succession [16].

Despite the different intentions of CW and RW, they have many commonalities. First, they have followed a similar historical trajectory, with early trials in the first two-thirds of the twentieth century, implementation beginning in earnest in the 1970s, and global growth over the past 50 years [17,18]. Second, both types of wetlands have evolved in terms of design and management technique over this period [19,20]. Third, both CW and RW have performance criteria by which they are evaluated, and critical parameters that influence performance [5,21]. Recently, some scholars and practitioners of wetland replacement have advocated for the use of CW and RW in combination where practicable, noting that the design and performance of these systems are complementary. Coupled CW and RW systems, known as Integrated Constructed Wetlands (ICW), offer the potential for holistic replacement of wetland services on the landscape [22,23].

During this era of wetland replacement, numerous studies and comprehensive reviews have been prepared on both CW and RW systems. However, the CW and RW literature have largely occurred in parallel, with little comparative assessment. The objective of this review is to assess the state of the science for wetland construction and wetland restoration, both individually and in combination as ICW systems. Specifically, CW, RW, and ICW systems are considered in terms of their objectives, design, management, performance, and limitations, with particular attention to ecosystem development and ecological services. Ecological services, including provisional, cultural, regulating, and supporting services as described by the Millennium Ecosystem Assessment [4], are ultimately what human-made wetlands are intended to replace. Through a comparative analysis of the services provided by different types of replacement wetlands we may better understand the potential for renewing the full complement of wetland functions that once existed on our landscapes.

2. Constructed Wetlands

2.1. CW Properties and Context

Wetlands have attributes that promote useful ecological functions. Among these are the capacity to transform, assimilate, or sequester influent nutrients and contaminants, thereby reducing the effects of potential pollutants on the landscape. In this way, natural wetlands have long alleviated stress on adjacent and downstream ecosystems [24]. Unfortunately, the accelerated production of anthropogenic stressors, such as nutrients, synthetic chemicals, eroded sediments, toxic metals, and oxygen-demanding organic wastes, has coincided with the era of wetland destruction and degradation. Our collective production of stressors thus exceeds the ecological capacity of the remaining wetlands [25].

One response to this problem is to construct wetlands for the explicit purpose of receiving and sequestering or altering wastewater before it can contaminate other ecosystems. For the purposes of this review, the term “constructed” wetlands refers to wetlands built for wastewater treatment. As Zhi and Ji [26] note, other terms are used interchangeably with “constructed wetlands”, including “engineered wetlands”, “treatment wetlands”, “artificial wetlands”, and “reed beds”. By all these names, CW have indeed become part of our ecological infrastructure. Vymazal [27] traces the evolution of CW from conceptual origins in 1901 [28,29] to the first documented experimental treatments in 1950s Germany [30]. In the late 1960s, wetlands were constructed for municipal wastewater treatment in the

Netherlands and in Hungary [27,31,32]. North American applications began at about the same time, as HT Odum experimented with natural estuarine wetlands and cypress swamps for the treatment of wastewater [27,33,34]. CW for wastewater treatment followed in the Houghton Lake Project in Michigan [35]. Research publications on CW were sparse until 1990, and then increased dramatically through 2020 [17,26]. The geographic extent of CW research sites is now global, featuring thousands of publications from more than 60 nations [17,36,37].

2.2. CW Objectives and Design

From early experiments to modern systems, CW have been used to treat a wide variety of wastewaters, including pre-treated municipal wastewater, livestock waste, industrial effluent, biomedical and pharmaceutical waste, agriculture and aquaculture runoff, landfill leachate, greywater, acid mine drainage, food processing waste, lake and river water, stormwater, and others [27]. The specific contaminants intended for treatment vary by wastewater source [38]. Nutrients, notably nitrogen and phosphorus, are contaminants of primary concern in municipal and agricultural waste streams. Oxygen-demanding wastes are also a concern in municipal and agricultural wastewater, and in some industrial effluent, such as paper mill sludge [5]. Industrial waste streams may include halogens, metals, metalloids, and synthetic organic compounds. Biomedical waste, as well as waste from municipal, greywater, and livestock sources, is apt to contain pathogenic microorganisms and pharmaceuticals. Acid mine drainage may be a concern due to its acidification of receiving waters. In addition, many of these wastewater streams contain high levels of dissolved and suspended solids [5].

To mitigate these various contaminants, CW takes a few basic forms [39]. Free water surface (FWS) constructed wetlands receive surface waters from a stream, swale, ditch, or effluent pipe. As water flows through the sealed wetland basin or series of basins, the water column contacts the substrate surface, vegetation, and atmosphere. FWS systems rely on algal mats and macrophytes, both for their ability to assimilate and metabolize contaminants and for the binding sites they provide for microbial biofilms. In FWS wetlands, the primary mechanisms for contaminant treatment include biomass assimilation, microbial metabolism, substrate and organic detritus adsorption, ultraviolet irradiation, volatilization, and sedimentation [38,39]. Through biomass assimilation, contaminants are taken up and sequestered, primarily by vegetation. Adsorption is a process by which contaminants adhere to the surface of detritus, vegetation, or substrate particles; these surfaces are also sites of biofilm formation and microbial metabolism [38,39]. Through these processes, FWS can be particularly effective at removing organic contaminants, suspended solids, and nitrogen from influent wastewater [39]. Floating treatment wetlands (FTW) are an innovation that suspend macrophytes on buoyant mats so that the roots dangle into the water column [40]. FTW have shown some promise for the removal of nutrients, metals, and organic contaminants from surface waters [41].

Subsurface CW include horizontal flow (HF) and vertical flow (VF) systems. In HF designs, wastewater is introduced to a bed of porous substrate and rhizosphere of emergent plants, through which it moves above an impervious liner until it exits the system [42]. This design maximizes substrate and rhizosphere exposure and minimizes atmospheric exposure, making it effective for the microbial metabolism of organics and nitrogen, filtration of suspended solids, and sorption of ions [39]. Vertical flow (VF) subsurface systems are similar but intermittently introduce wastewater to the surface of a substrate bed, through which the water percolates to drainage structures below [43]. VF systems allow for enhanced oxygenation, which can provide advantages for the aerobic treatment of organic contaminants and for nitrification [39]. A variation is the “tidal flow” (TF) CW, in which wastewater is allowed to flood a subsurface bed completely, held for a period, and then drained completely [44,45]. This intermittent fluctuation allows for alternating periods of oxidation and reduction, thus improving the treatment efficiency for some contaminants. Many CW are hybrid designs, which use more than one type of CW in succession to provide

a complex network of aerobic and anaerobic zones, settling ponds, filtration, sorption sites, and microbial communities to promote multiple removal pathways [46,47].

2.3. CW Performance and Management

Physical, chemical, and biological treatment mechanisms in CW vary by design, and thus CW conformations are specific to contaminant treatment objectives. Essentially, CW effectiveness relies on the interactions of the waste products with the vegetation, substrate, microbes, and water column of the CW [47]. Physical waste removal processes include flocculation, precipitation, sedimentation, and filtration, and are thus reliant on water column interactions with the substrate [6]. The substrate is also the primary locus of chemical removal processes, including interactions with ions and adsorbent surfaces, ion exchange, and redox processes. For this reason, substrate has been a focus of CW research, testing the relative efficacy of various adsorbent substrate materials, including alum sludge, limestone, coal slag, sand, rice husks, biochar, and many others [47,48]. The different physicochemical characteristics of substrate materials can result in a range of removal efficiencies for different contaminants; according to Patyal et al., substrate materials varied removal efficiency for oxygen-demanding wastes from 71.8–82%, for total phosphorus from 77–80%, and total nitrogen from 52–82% [47].

Biological processes, including microbial metabolism, phytoremediation, biosorption, and predation, also occur in the substrate, as well as in the water column [6]. Macrophyte stems, roots, and leaves reduce the flow rate and provide additional surface area for microbial biofilms. Macrophytes also exude carbon compounds and oxygen, particularly in the rhizosphere, stimulating chelation and aerobic microbial metabolism [49]. Macrophyte metabolism contributes to waste retention and removal through phytoaccumulation, phytodegradation, volatilization, and sequestration [17,50].

The physicochemical characteristics of CW are subject to careful management. For instance, hydraulic parameters are critical aspects of CW design [51]. The expected hydraulic loading rate (HLR) and contaminant concentration determine the necessary surface area and volume of the CW [52]. Equally critical is the frequency and duration of flooding and drawdown events, as these regulate the oxygen availability in the water column and substrate. The CW design, HLR, and hydrologic regime determine the hydraulic retention time (HRT), the average length of time that influent wastewater remains in the CW [53]. Generally, a longer HRT results in higher percentage of contaminant removal for a given system [54]. For example, Toet et al. [55] found a significant increase in retention of N and coliform bacteria as HRT was increased from 0.3 to 9.3 days in a FWS municipal wastewater treatment wetland. In a HF CW system, Ghosh and Gopal [54] similarly report a greater retention of nitrogen (83–100%) with an HRT of 4 days over an HRT of 1 day (21–77%). Many CW designs use long, sinuous paths, baffles, or multiple cells to lengthen the HRT and maximize the system's efficiency [56].

Other factors can confound CW performance, however; season, temperature, pH, oxygen availability, changes in biomass productivity, bioturbation, and weather events can all influence the efficacy of contaminant removal [50]. These, too, can be managed to an extent. In cold-weather climates, temperatures may be increased with greenhouses [50] and bed heating [57]. Artificial aeration, pH buffering, insulation, and bio-augmentation are all used to maintain critical parameters to achieve the most efficient waste removal [50,58]. Vymazal et al. [48] and Ingrao et al. [59] review emerging challenges in new wastewater streams and novel pollutants amid the tighter regulation of effluent contaminant concentrations.

CW biota are also managed. The ideal macrophyte species for CW systems vary by geography, design, and waste stream, but generally the plants must be robust, with quick establishment, rapid growth, large biomass, and tolerance to the stressful conditions that wastewater presents [60]. While plant species selection is deemed an important part of CW design, the advantages or disadvantages of one macrophyte species over another are inconsistent in the literature [60,61]. Vymazal [61] surveyed over 640 studies on FWS systems that introduced a total of 150 macrophyte species. The most used species are

Typha latifolia, *Phragmites australis*, *Typha angustifolia*, *Juncus effusus*, *Scirpus lacustris*, *Scirpus californicus* and *Phalaris arundinacea*. Others [62] have experimented with woody species in FWS CW. Subsurface CW often use emergent macrophytes for the enhanced treatment that the presence of a rhizosphere provides. Vyzmal [27] found that macrophytes of the genera *Phragmites*, *Typha*, and *Scirpus* are most used in subsurface systems. Of course, both FWS and subsurface CW provide habitat for volunteer species; Knight [63] notes that more than 600 plant species occur in CW systems in the US.

Nuisance animals, the subject of removal or control in many CW (particularly FWS) systems, include burrowing mammals (e.g., muskrat, beaver), mosquitos, bioturbators, and aggressive grazers [6,64]. However, wetland animals can also have a positive effect on CW performance. Li et al. [64] review an extensive list of invertebrates, fish, birds, reptiles, and mammals that can enhance contaminant retention through adsorption and bioaccumulation, by increasing the diversity of the microbial community, consuming pathogens, and stimulating plant growth.

2.4. CW Ecosystem Services and Limitations

CW are primarily evaluated by their capacity to retain, remove, or reduce influent contaminants (Table 1). For example, Varma et al. [50] summarize CW design efficacy by the removal of nitrogen, phosphorus, and oxygen demanding wastes; depending on CW type, the study cites average efficiencies ranging from 45 to 89% removal. Other CW are similarly evaluated for their amelioration of metals and metalloids, pathogens, dissolved and suspended solids, acidic waters, and synthetic organic compounds [6,65]. Data from thousands of studies demonstrate the effectiveness of CW nutrient retention and removal, though the success is not without caveat. CW can be highly efficient, but efficiencies can fall with excessive HLR, low temperatures, low HRT, poor oxygenation, and unsuitable pH [50].

Table 1. Examples of Ecosystem Services provided by constructed wetlands (CW) for wastewater treatment. Based on Ecosystem Services categories of the Millennium Ecosystem Assessment as applied to wetlands [4,66,67].

Ecosystem Service	Representative CW Performance	Selected Sources
Provisional	Biomass harvest, microbial fuel cell Biomethane, ethanol production	[48,68] [69]
Cultural	Recreational benefits \$580 to \$9160 USD per hectare	[70]
Supporting	Regional nutrient and water cycling Regional ecosystem connectivity	[71] [72]
Regulating	73–99% retention of BOD, COD, TSS, TN, TP from wastewater sources 45 phyla of bacteria and archaea supported in CW >600 plant species supported in CW 36 macroinvertebrate taxa; >60 bird species, including species of concern Stormwater retention Carbon sequestration 676 g CO ₂ eq m ⁻² yr ⁻¹ Net GHG emission	[6,48,50,65] [48] [63] [73–76] [77] [48,78] [79]

If these critical parameters are maintained within reasonable levels, CW removal pathways such as denitrification and respiration can occur for an indeterminate time. However, CW performance may decline over time in cases where the target pollutants are to be sequestered [80–83]. CW systems have physical limitations; for example, sedimentation pools can fill, flow paths can become channelized, and substrate pore space can become clogged [84]. Chemical limitations occur as sorption sites become saturated, ion exchange capacity becomes depleted, and buffering capacity is exhausted. Biological processes such as phytoremediation and biosequestration are similarly finite. Management effort can extend the useful life of a CW with energy inputs and re-fitting: removal and replacement

of vegetation and substrate, aeration and heat, pH buffering, and management of HRT and HLR. Finally, each CW is a function of its wastewater source, which is likely impermanent. Carefully managed CW have been shown to be effective for decades [48]. For instance, Vymazal [85] reviews 17 HF systems in the Czech Republic and finds that “the treatment performance is steady for more than 20 years with outflow concentrations $< 15 \text{ mg L}^{-1}$ of BOD_5 and TSS and $< 50 \text{ mg L}^{-1}$ COD”. Another study of the long-term efficacy of HF CW finds 84–90% removal of TSS and organic contaminants and 65–67% removal of nitrogen and phosphorus after 20 years of operation [48,86]. Brix et al. [83] review several hundred Danish reed bed CW and find that these systems maintain TSS and BOD effluent of less than 20 mg/L and 30–50% P and N removal for two decades, after which many were reconditioned or decommissioned.

CW are seldom evaluated according to their status as ecosystems. Such an assessment illustrates several key differences between CW and natural wetlands. First, while some contaminant removal pathways (like denitrification, carbon sequestration, and methanogenesis) may improve with time, many are at their most efficient when macrophyte and microbial growth are vigorous, when pore spaces are open, when binding sites and sedimentation spaces are available, and when redox environments are interspersed. These conditions are maximized in early succession, and CW are commonly managed to maintain the conditions of early succession by vegetation harvest and substrate replacement [70,84]. Second, the introduction of wastewater makes a CW a high-stress environment, and the expectation of a continuous performance precludes natural disturbance events such as droughts, floods, extensive herbivory, or fire. High-stress, low-disturbance conditions can occur in natural wetland ecosystems, of course, but they are generally not maintained for years or decades. Long-term stress typically yields systems of comparatively low diversity, predominantly featuring stress-tolerant organisms [87]. Third, CW are hydrologically dominated by wastewater inflows, with only limited hydrological or biological exchange with adjacent ecosystems. Wastewater, as a potential source of pollution, is carefully maintained within the CW system by impervious liners and berms from inflow to outflow, even as external water sources (e.g., flood waters) and the organisms and material they carry are excluded from the CW system. This artificial hydrologic regime may limit CW diversity and the seasonal patterns of germination and colonization [88,89].

Knight [63] issued an early review of the potential ancillary services of CW systems, noting that the abundant water source and ubiquitous presence of emergent macrophytes make CW attractive to both aquatic and terrestrial fauna. Ghermandi and Fichtman [70] further report that CW are used for both provisional and cultural services—the annual monetary value for which ranges from 580 to 9160 USD per hectare—although they are not without the potential for nuisances and human hazards. Vymazal et al. [48] note that many CW are small, and are thus limited in terms of provisional services, but also review emerging potential for biomass and microbial fuel cell energy production from CW. Many studies have confirmed the capacity of CW to support microorganisms, macroinvertebrates, amphibians, reptiles, birds, fish, and diverse vegetation [73], though Wiegleb et al. [72] note that very few published CW studies systematically document the biodiversity and wildlife use of CW. Zhang et al. [73] review such studies and find evidence that CW can support native wildlife. For instance, various CW have been found to support 90% of macroinvertebrate species and 54% of regional flora found in natural reference wetlands, as well as abundant native birds, amphibians, reptiles, and mammals [74–76,90]. Zhang et al. suggest, however, that CW often lack the habitat and hydrological heterogeneity of natural wetlands, while also intentionally incorporating nonnative plant species [73]. These nonnative macrophytes tend to limit native biodiversity in CW. Further, CW have been shown to act as ecological traps, luring native species into an environment that is less than ideal for its toxicity and poor habitat quality. In this way, CW may actually decrease regional biodiversity over time [73].

In summary, CW have demonstrated success at replacing the ecosystem services for which they are designed, namely water quality improvement (Figure 1). They are generally

not intended to replace all the other ecological services of natural wetlands in the landscape, though they may provide a habitat for early successional wetland-dependent species, productivity, educational opportunities, and an aesthetically pleasing environment. The management of CW in a state of high stress, with a relatively closed hydrologic regime and low degree of landscape connectivity, limits the capacity to provide a greater range of ecosystem services. Furthermore, the CW lifespan means that any service rendered is temporary.

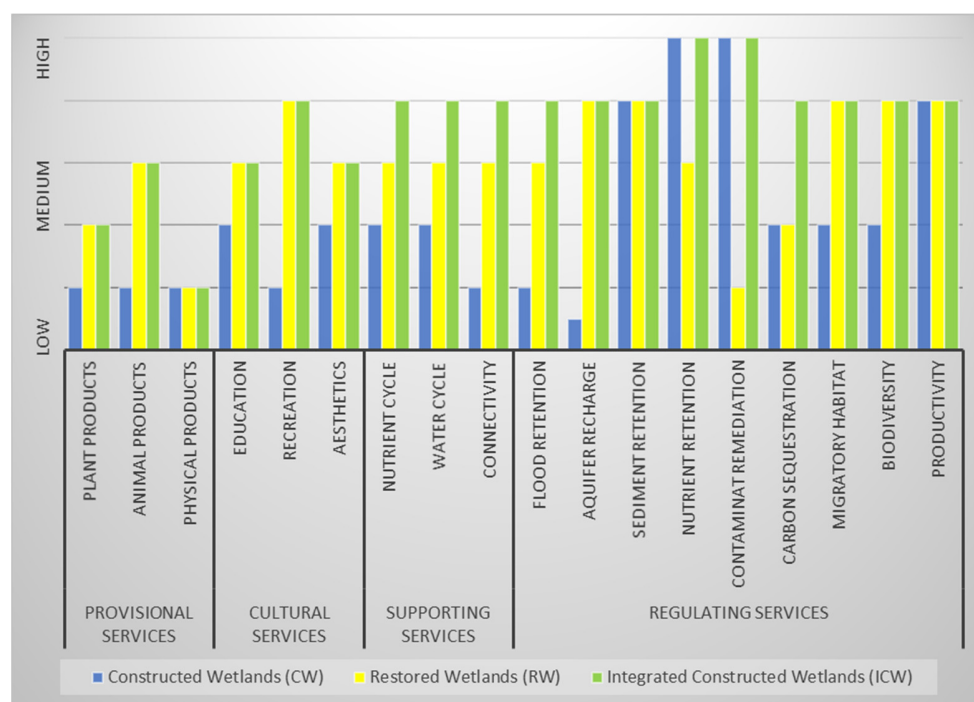


Figure 1. Idealized Ecosystem Services of Ecosystem Services provided by constructed wetlands (CW) for wastewater treatment, restored wetlands (RW), and integrated constructed wetlands (ICW). Based on Ecosystem Services categories of the Millennium Ecosystem Assessment as applied to wetlands [4,66,67].

3. Restored Wetlands

3.1. RW Properties and Context

While CW do, in fact, restore certain wetland functions to the landscape, the term “wetland restoration” has come to mean something fundamentally different than CW in the scientific literature. Restored wetlands (RW) are intended to reintroduce a broad suite of ecological services to the landscape [25]. Water quality improvement is among these, though most RW are not specifically designed to treat wastewater. Other wetland functions, such as biodiversity enhancement, floodwater control, storm abatement, carbon sequestration, and aquifer recharge, are just as, if not more, important as RW goals and objectives [25]. Another difference is that RW are designed to encourage successional processes; that is, RW are designed for change [16]. Thus, the landscape context—proximity to and connectivity with other ecosystems, topography, hydrologic sources, and disturbance regimes—as well as in-system features such as microtopography, habitat interspersions, seed bank, and biodiversity are critical aspects of RW [91].

The term RW is used here in the landscape sense, as in restoring wetland functions to landscapes in which they once existed. This is to minimize jargon, for, in this broad sense, RW can be taken to include wetland creation (systems built where none ever existed), wetland enhancement (improvement to the structure and function of existing wetlands), and wetland mitigation (wetlands built as legal compensation for wetland functions lost elsewhere) [12,92]. Collectively, these RW efforts have followed a similar historical trajec-

tory to CW. While indigenous peoples around the world have been managing and restoring wetlands for millennia [93], documented cases do not arise until the end of the nineteenth century. One early effort was conceived by Frederick Law Olmsted in Boston in the 1880s, as he incorporated a salt marsh restoration into parkland called the Back Bay Fens [2]. Olmsted's "fen" still exists today, and his methods would be recognized by modern restoration ecologists—basin reconstruction, reconnection to the tidal hydrologic regime, planting native macrophytes, and replanting to establish the desired community. Unfortunately, such projects were rare during a systemic national effort to drain and fill wetlands. The first concerted efforts to restore wetlands were initiated by private organizations like the Izaak Walton League and Ducks Unlimited. Ducks Unlimited, established in 1937, had completed over 100 wetland restoration projects by 1943 [2].

The re-introduction of wetlands to the landscape became a bigger business as the US federal and state governments began to incentivize and mandate it. The no-net-loss policy, derived from the Clean Water Act of 1972 and subsequent amendments and court cases, mandated the compensation of wetlands lost to development. Other programs, such as the US Fish and Wildlife Service Partners for Wildlife (1987), the Wetland Reserve Program in the 1990 Farm Bill, and the 1986 North American Waterfowl Management Plan, encouraged the restoration of millions of wetland hectares [2]. Unsurprisingly, academic research on wetland restoration has blossomed since, notably in North America, China, Australia, Brazil, and Europe [18].

3.2. RW Objectives and Design

RW serve a different purpose than CW, and thus have different design considerations. While many RW change the regional water quality, this is typically not the sole objective. Olmsted's Back Bay Fens, for instance, were primarily intended for aesthetic appeal [2]. Most of the wetland restoration projects of the early 20th century were undertaken for the provision of wildlife—particularly waterfowl—habitat [2]. The objectives of modern wetland restoration range over a wider collection of services, including flood water and silt retention, soil conservation, biogeochemical management, cultural significance, education, and academic research. In some cases, financial incentives are the primary driver of a restoration project. This is particularly true for government-subsidized conservation programs and the establishment of mitigation wetlands, which can be a lucrative business [94]. Naturally, objectives differ by project. However, as wetland restoration has become more closely tied to legal requirements and money, a basic objective has emerged: the RW needs to exhibit the characteristics that define a wetland. In brief, this means that the RW must have a hydrologic regime that is sufficient to develop hydric soils and support hydrophytic vegetation [95].

The requirement of a suitable hydrological regime means that an RW must be situated in such a way that it receives periodic inundation from some combination of surface flow, tidal action, groundwater discharge, and precipitation. Landscape setting is thus a critical consideration for RW [96]. Wetlands built where no wetland previously existed—called created wetlands in the literature—often result in low diversity and a high incidence of nonnative species, particularly when they are geographically isolated [97]. Greater success has been achieved with RW located in topographic depressions, floodplains, or catchments that have historically flooded with sufficient duration to develop hydric soils [97]. Another consideration for landscape setting is connectivity with other natural habitats, which can serve as a source of propagules and organisms for the new RW. Middleton [11] makes the case that landscape setting dictates the hydrologic regime of the RW, and that these patterns of flood depth, duration, and recurrence interval set the stage for all other physical, chemical, and biological processes within the ecosystem.

Soil characteristics can be useful in the identification and selection of a restoration site and are critical factors in RW success [98]. Remnant hydric soils from long drained and filled wetlands are an indicator of a historic hydrologic regime. Such sites are not always a great candidate for an RW, as land-use changes and regional hydrologic alterations may

have rendered them unsuitable for wetland restoration. Even so, the presence of remnant hydric soils indicates (1) connection with hydrologic sources, or past connection that may be restored; (2) soils with the physical characteristics to sustain periodic inundation; and (3) soils that may have a remnant seed bank of hydrophytic propagules [98,99]. Remote sensing data from aerial and satellite images and topographic surveys have proven useful in the identification of remnant hydric soils and suitable locations for RW, though remote sensing can be problematic for forested habitats [100–102]. Of course, sites with ideal landscape and soil characteristics must also be available for acquisition, or have a willing landowner, for the RW project to proceed.

The soil conditions of an RW site can exist over a wide range in terms of hydraulic conductivity, nutrient and organic matter, toxicity, microtopography, and seed bank. These attributes can influence the physical character (e.g., substrate permeability, water flow path and depth), chemical character (e.g., nutrient availability, contamination), and biological character (e.g., habitat heterogeneity, vegetation, trophic status) of the RW. In cases of severe degradation, soil may need to be amended with topsoil or other substrate to alter nutrient availability, increase organic matter, and enhance the microbial community [103,104]. Some RW projects have attempted to improve soil quality by transplanting soil from an existing wetland. This “donor soil” approach has been demonstrated as an effective way to quickly establish a depleted wetland seed bank, leading to the more rapid establishment of native wetland macrophytes [105].

Given a particular site, selected for its landscape context, hydrologic regime, and soil characteristics, the next objective of an RW project is to plan wetland habitats [11,106,107]. Depending on location and conditions, RW may provide marine, estuarine, riverine, lacustrine, or palustrine habitat [107]. The habitat that ultimately develops on a restoration site is dependent on the hydrologic regime and microenvironments of the wetland basin. Many RW projects involve strategic earthmoving to re-contour flooding zones and habitats [11,107]. Excavation may also be necessary to reconnect hydrological sources. Ditch filling or rerouting, berm or floodwall removal, runoff redirection, and drainage tile removal have all been used to restore a hydrologic regime. In many cases, microtopography can be introduced into the new wetlands at this stage. Spatial diversity in elevation, woody debris, slope, and aspect can yield a wetland community that is more diverse and more interspersed than a uniform wetland basin [11,107].

Once selected, planned, and contoured, an RW must be re-vegetated. Much field research has been devoted to the question of passive revegetation (i.e., the extent to which a newly restored wetland will vegetate itself through natural recruitment and selection) versus active revegetation (the need for managed macrophyte selection and planting) [11]. Certain RW projects may indeed re-vegetate themselves with native hydrophytes, particularly if they are connected to existing high-quality wetlands, in close proximity to propagule sources, and/or in possession of a high-quality, viable seed bank. However, the results of passive revegetation are highly variable and context-specific, particularly regarding abiotic factors [108–110]. Passively revegetated RW can be vulnerable to invasive species [111] that can preempt and outcompete native wetland vegetation and result in an undesirable successional trajectory [16]. Active revegetation—seeding, donor soil, and planting, along with re-planting and invasive species removal in the first few seasons following restoration [11]—is more costly and labor-intensive but has been shown to facilitate the establishment of desirable species in some cases [112–114]. The sourcing and selection of seed can play a critical role in success, as the genetic character of propagules can be used to establish the local or historic ecotypes of a diversity and functionality suitable to thrive in current and future environmental conditions [115]. All things considered, the approach to revegetation should fit the objectives and acceptable outcomes of the RW project and should, therefore, not be seen as a one-size-fits-all model [110].

3.3. RW Performance and Management

The choice of revegetation strategy is part of a larger philosophical debate on the degree to which anthropogenic management should be used to establish and maintain RW [11]. One approach, which has been called “self-design”, emphasizes the role of the biogeochemical and hydrological environment in determining how plant communities will develop in the RW [116,117]. In the extreme view, self-design can be implemented as passive management, relying solely on natural revegetation, successional development, and the capacity of ecological communities to self-assemble. However, self-design does not preclude plant introduction or vegetation management as a means of increasing the rate of development [118]. At the heart of the self-design approach is the idea that “the system itself will optimize its design by selecting for the assemblage of plants, microbes, and animals that is best adapted for existing conditions” [117]. An alternative perspective, sometimes called the “designer” approach to restoration [11], calls for attention to the life history strategies of target species in the restoration project. Through the careful introduction (and reintroduction) of plants and propagules, ecological engineering, and management of biotic and abiotic factors early in the restoration process, the restorationist can select individual species to be part of the assembled community [119]. The goal is not to achieve a particular species complement as an endpoint, but rather to encourage the establishment of desirable species that would otherwise be overwhelmed by undesirable invasive species [119]. As Middleton [11] notes, these two views are not mutually exclusive, but they do conceptualize ecosystem development in different ways. Ultimately, the degree of human intervention in a RW may depend upon the goals and performance standards of the project.

Many RW projects use one or more reference wetlands, or historic reference conditions, to evaluate success [120,121]. A reference wetland ostensibly represents the desirable structure and function of similar ecosystems in the region of interest. The idea has emerged from the Leitbild concept of river restoration in Germany and Austria; Leitbild refers to the ideal or undisturbed state of an ecosystem [122]. The identification and measurement of one or more such benchmarks can be used to set the parameters and expectations for the restoration work. Middleton [11] notes that, while the reference ecosystem approach can be useful in RW, the goal of matching RW conditions to reference conditions may be unachievable. This is because the RW and reference sites may have different land use legacies and different site impairments—leading to irreconcilable differences in ecological development—or changes in regional biota and climate that could make historical reference conditions untenable [11]. Moorhead [121] adds that the reference ecosystem concept can be problematic, given the differences in successional state (reference systems typically being mature, while restored systems are early successional), and the inherent variability that exists even among potential reference sites. According to Moorhead, RW goals should not be to duplicate the conditions of reference sites, but rather to establish “self-supporting and self-maintaining” ecosystems. The maturation of these systems, Moorhead suggests, is better measured with the general structural and functional attributes of early and late-stage ecosystems than with specific points of comparison to a reference ecosystem [121].

Even so, many RW—particularly mitigation wetlands—are undertaken with specific performance indicator goals. Targets for RW evaluation can be placed in several categories [123]. Structural metrics consider physical aspects of the wetland environment, such as the vegetated percentage, degree of habitat interspersion, or macrophyte biomass. Structural metrics can also be used to quantify landscape-level patterns of land use and connectivity. Measures of taxonomic diversity and evenness, particularly of macrophytes, are common, and often include indices of species quality. For example, the floristic quality index and associated coefficient of conservatism can be used to evaluate vegetation quality, based on species tolerance [124]. Indicator species are also commonly used to denote the presence or absence of particular communities or site conditions. Functional metrics are measures of ecosystem processes, such as nutrient processing, organic matter accumulation, and productivity, as well as process surrogates, such as trophic composition and functional

diversity. Finally, taxonomic composition can be used as a direct comparison of species abundance with reference wetland composition [123].

Brudvig et al. [125] suggest that these attribute categories occur with different degrees of variation in RW systems. Structural metrics, according to Brudvig et al., may be established with the greatest degree of certainty. Diversity and function are subject to stochastic events and are more difficult to predict. The metric that is most subject to variation, according to Brudvig et al., is taxonomic composition. Developmental variation thus makes RW performance evaluation a challenge. RW are designed as open systems that are expected to progress through succession. The RW target goals are likely to be characteristics of late-successional reference systems that will emerge, if they emerge at all, many years after the initial RW design and management phases. Indeed, the ultimate goal of a self-maintaining ecosystem may well be incompatible with the rigid metrics of particular functional rates or species lists.

3.4. RW Ecosystem Services and Limitations

In contrast with CW, the ecosystem services offered by RW are both broader and more variable (Table 2; Figure 1). While CW are intended to ameliorate specific contaminants for the duration of their existence, RW are ecosystems with attributes, structures, and functions that will change over time. RW also occur in a much wider contextual range than CW, as they are situated in different landscapes with different levels of degradation, degree of connection, and stress and disturbance regimes. Even in similar RW projects, successional trajectories and attributes can be highly variable, and this translates to variable ecosystem services [125].

Table 2. Examples of Ecosystem Services provided by restored wetlands (RW). Based on Ecosystem Services categories of the Millennium Ecosystem Assessment as applied to wetlands [4,66,67].

Ecosystem Service	Representative RW Performance	Selected Sources
Provisional	36% greater provisioning services than degraded wetlands	[126]
Cultural	Recreational benefits >\$130,000 USD/ha·yr	[127]
Supporting	Regional nutrient and water cycling	[128]
	Regional ecosystem connectivity	[128]
Regulating	29–90% retention of BOD, COD, TSS, TN, TP from river water	[117,129]
	Thousands of plant species supported in RW	[11]
	>30 macroinvertebrate taxa; >180 of bird species, including species of concern	[130,131]
	Stormwater retention	[77]
	Carbon sequestration 84 g C m ⁻² yr ⁻¹	[130]
	Net GHG emission	[132]

Still, a great deal of experimental evidence demonstrates that RW provide a wide range of ecosystem services. In a large meta-analysis, Meli et al. [126] found that RW significantly increased the diversity of vertebrates, vascular plants, and invertebrates as compared with degraded wetlands, and that RW diversity did not significantly differ from that of natural reference wetlands. RW outperformed degraded wetlands in every category of provisional, supporting, and regulating service, while providing similar cultural services [126]. For all categories combined, RW were found to support 43% more ecosystem service than degraded wetlands, but 13% less than natural wetlands [126]. Meli et al. caution that RW ecosystem services are context-specific and largely dependent on the ecosystem type, site condition, and RW design [126].

Other scholars have provided evidence that a particular RW may not be able to provide all ecosystem services equally. Jessop et al. [128] surveyed 30 mitigation wetlands in Illinois, USA, and found a negative relationship between biodiversity and indicators of nutrient cycling such as soil organic matter abundance, decomposition rates, and potential denitrification. They conclude that “optimizing restored wetlands for nutrient storage and

removal may come at the expense of biodiversity” [128]. Hansson et al. [133] similarly studied 32 RW in Sweden and found that wetland designs that favor biodiversity do not always result in the greatest nutrient retention. Specifically, shallow RW with large surface areas and spatially complex shorelines were found to provide a high biodiversity of birds, benthic invertebrates and macrophytes, but retained phosphorus less effectively than deeper basins [133].

The extent to which wetland functions and services are fully restorable is still an open question. Zedler and Kercher [25] note that some of the factors leading to wetland loss and degradation on a landscape may inhibit the restoration of wetland functions. For example, the alteration of flood regimes, soil characteristics, nutrient dynamics, groundwater levels, and regional climate can limit the functional capacity of RW [25]. In addition, the regional complement of species may have changed since the natural wetlands were eliminated or degraded, and RW functions and services may, thus, be limited by regional diversity or inhibited by invasive species [25].

In the most general sense, a successful RW is one that responds to and changes in the face of shifting stress and disturbance regimes, and that has the capacity to support a diverse but transient assembly of species over successional time. Given time, some RW may approach the functional capacity of pristine natural wetlands. In most cases, however, the ecological services provided by individual RW should not be expected to replicate those provided by natural reference wetlands or wetlands that once existed. To fully replace the wetland services we have lost, we may need to think of multiple complementary wetland rehabilitation projects that, together, provide comprehensive ecosystem services at the landscape scale.

4. Integrated Constructed Wetlands

Thus far, this review has demonstrated that CW and RW are contrasting and parallel approaches to the re-establishment of wetland services on the landscape. They differ in their objective, design, management, performance objectives, and in the ecosystem services they can provide. CW and RW services are not mutually exclusive; however, CW, especially FWS systems, can provide habitat and other ecosystem services in addition to nutrient retention, just as RW can sequester and transform influent contaminants. In general, though, CW are not designed for the long-term successional development of wetland habitat, nor are RW designed to process highly contaminated waste streams.

Many scholars and practitioners have observed that CW and RW are not only compatible, but synergistic, such that their use in combination may be able to achieve a broader range of ecosystem services on the landscape than either could alone [23,134,135]. CW and RW, in combination, are known as Integrated Constructed Wetlands (ICW); [22,23]. The ICW concept has roots in the whole-ecosystem studies of HT Odum and KC Ewel [34,136], and in the integrated small watershed research of Bormann and Likens [137,138]. Initially developed in Ireland [22], the ICW concept has since been applied elsewhere, but has yet to become the standard for wetland replacement [139]. Harrington et al. [140] present ICW as an ecosystem approach (after the Convention on Biodiversity 2010) to water quality and land-use management on a watershed scale. The ICW attributes suggested by Harrington et al. include: that influent contaminants are maintained below the threshold of macrophyte toxicity prior to entering the system; that the ICW design incorporates multiple sequential wetland cells; that the cells predominantly feature shallow habitat with dense emergent vegetation; and that the entire system be managed for ecosystem services beyond water quality improvement [140].

4.1. ICW Properties and Context

As we have seen, CW and RW both provide important functions to ecologically impoverished landscapes. Over the past five-plus decades, however, CW and RW research and development have rarely informed one another or been combined into holistic efforts to replace wetland services. McInnes et al. [139] speculate that this lack of integration

may “stem from narrow disciplinary framing of legacy regulations or a lack of vision by, and appropriate support tools for, planners and managers”. Clearly, there are barriers to integration, including cost, land availability, regulatory requirements, technical difficulties, resistance to change, and disconnection between regulators, ecologists, engineers, land-use planners, and regulators [139,141]. Mitsch [142] suggests that the disconnect is primarily between ecologists and engineers, or more broadly between the disciplines of restoration ecology and ecological engineering. The benefits to CW and RW integration are equally clear. First, the functional capacity of integrated systems has the potential to exceed that of either system alone [143]. Second, integration allows for spatial and temporal heterogeneity, allowing for different functional loci in space and time [144]. Third, integrated systems allow for the coupling of ecological processes, such that the biological, chemical, and physical processes of one ecosystem are linked to processes of other ecosystems within the landscape [145]. Fourth, and encompassing the previous three, landscapes that are designed to integrate wetland functions optimize and maximize ecological services by supporting both human and natural systems [146,147]. These benefits of ICW systems are not merely theoretical; they have been demonstrated in practice.

4.2. ICW Objectives, Design, and Performance

The purpose of ICW systems is to ameliorate the pollutants from a wastewater stream while also providing a long-term, successional habitat for wetland-dependent species and providing provisional, supporting, regulating, and cultural services as appropriate to the setting. The goal is not to provide all ecological services at all times, but rather to couple treatment processes with wetland ecosystem functions in ways that are integrated with the socioecological character of the landscape. Scholz et al. [23] describe ICW objectives as follows: “the explicit integration of (a) the containment and treatment of influents within emergent vegetated areas using (wherever possible) local soil material; (b) the aesthetic placement of the wetland structure into the local landscape with the intention of enhancing the site’s ancillary values; and (c) enhanced habitat diversity and nature management”.

ICW can take different forms. Commonly, a CW is built to receive and treat wastewater by some combination of subsurface and/or free surface flow cells. The CW effluent then flows into an RW before entering a receiving body of water. Scholz et al. [23] describe an ICW in Ireland consisting of sequential FWS cells that intercept agricultural runoff before discharging into a stream. Similar designs are described in the US by Ludwig and Wright [148] and in Sudan by Ladu et al. [149]. Yan et al. [150] use an alternative ICW hybrid design in which domestic wastewater from a septic system enters a subsurface CW before discharging into an RW. Zhang et al. [151] similarly use an intricate series of HSF, VF, and FSW cells to treat domestic wastewater while providing a wetland habitat in China. Boets et al. [152] describe a hybrid system for treating pig manure in Belgium that incorporates eight cells of both FWS and subsurface design. Alternative ICW designs feature RW with floating treatment wetlands, either in the RW itself or in the river, lake, or lagoon water that supplies the RW [153].

The multi-cell designs that are common to many ICW share some characteristics with hybrid CW systems. Indeed, hybrid CW have been shown to support wetland biodiversity and associated ecosystem services [63,70,73,154]. ICW differ from hybrid CW in that they include cells which are: (1) hydrologically connected to ground or surface water other than the wastewater stream; (2) designed primarily to support biological diversity; (3) connected with other ecosystems in the landscape; and (4) open to the adaptive cycle of succession. While the multi-cell design facilitates the multi-faceted nature of ICW, this is not always necessary. ICW that receive pollutants in low concentrations may combine processes in a single basin. For example, Mitsch et al. [117] designed a FSW system for ambient river water that showed both effective improvement in water quality and habitat provision [90,129].

At the broadest conception of performance, ICW systems may be evaluated according to the IUCN Integrated Wetland Assessment Toolkit, developed in 2009 [155]. In this approach, wetlands are evaluated not only for their physical (e.g., hydrological and water

quality) and biological (e.g., biodiversity) attributes, but also for their contributions to local livelihoods, to the regional political economy, and to regional socio-ecological systems. Such analyses require a significant investment of time and resources and are seldom incorporated into the evaluation of a single ICW system. Most ICW systems are measured in terms of water-quality improvement, biodiversity, and/or socioeconomic benefits. For example, van Biervliet et al. [156] report a significant nutrient retention, along with a significant increase in avian species richness for an ICW system in the UK. Becerra-Jurado et al. [157] show that ICW support a macroinvertebrate species richness similar to nearby natural aquatic environments; Boets et al. [152] also report macroinvertebrate richness increases in an ICW in Belgium. Harrington and McInnes [158] note significant nutrient retention and numerous cultural, provisioning, supporting, and regulating services provided by ICW systems in Ireland.

4.3. ICW Ecosystem Services and Limitations

ICW systems seek to expand and extend ecosystem services by coupling wetland habitats and functions with the socio-ecological landscape. Ideally, ICW would contribute to the full range of ecosystem services, much as natural wetlands may have done prior to their conversion to other land uses (Table 3; Figure 1). In practice, ICW ecosystem services are not all equally amenable to assessment. The most straightforward evaluation is for nutrient and contaminant treatment, measured in terms of inflow–outflow comparison in the manner of traditional CW. For this there is abundant evidence of ICW success in the removal or retention of nutrients, oxygen-demanding wastes, and suspended solids, mostly from agricultural, livestock, and pre-treated municipal waste streams [141,156,159]. Biodiversity is also easily evaluated, and multiple studies confirm the capacity of ICW to support native plants and animals [117,152,157]. Ancillary ecosystem services are more difficult to quantify and are not as commonly measured. Everard et al. [141] and McInnes et al. [139] use stakeholder engagement to qualitatively evaluate the socioecological efficacy of ICW. Mao et al. [144] develop a comprehensive scoring system for evaluating ICW ecosystem services, using the visitation rates of local residents as a surrogate for socioecological benefits. While such studies provide positive evidence for a wide range of ecosystem services, they also indicate the need for a better system to assess these services. Many methods exist for the valuation of ecosystem services [160], but there is no standardized approach for the evaluation of ICW systems, or indeed ecosystems in general.

Table 3. Examples of Ecosystem Services provided by integrated constructed wetlands (ICW). Based on Ecosystem Services categories of the Millennium Ecosystem Assessment as applied to wetlands [4,66,67].

Ecosystem Service	Representative ICW Performance	Selected Sources
Provisional	Biomass, freshwater provisioning	[158]
Cultural	High degree of recreational value	[158]
Supporting	Regional nutrient and water cycling	[141]
	Regional ecosystem connectivity	[141]
Regulating	>90% retention of BOD, COD, TSS, TP, TN from mixed sources	[154,158,161]
	Potential for high plant diversity	[11,63]
	>17 macroinvertebrate taxa; 27 bird species	[154,156]
	Stormwater retention	[77]
	Carbon sequestration $\sim 84 \text{ g C m}^{-2} \text{ yr}^{-1}$	[130]
	Net GHG emission	[132]

It must also be acknowledged that ICW may not be appropriate for all situations. Contaminants that are particularly toxic may inhibit the ecological development of an ICW, for example. Leaching is much more likely in an ICW than in a traditional, impervious CW system, so wastewater streams with potential groundwater contaminants may not be

appropriate for an ICW. Nuisance plant and animal control and disturbance management are likely to be more complicated in an ICW, particularly in its developmental stages. In addition, ICW have greater land area, cost, and management requirements than either traditional stand-alone CW or RW. Still, where appropriate and feasible, ICW offer tremendous advantages in terms of holistic wetland replacement.

5. CW, RW, and ICW: Summary Assessment

Replacing what we have lost, and continue to lose, in terms of wetland services is no simple task, but the last 50 years of research have provided some valuable lessons. Clearly, there is a place for traditional CW, especially in urban environments, where land area is limited and/or the contaminant in question is particularly toxic. CW have proven to be quite effective in ideal conditions; however, their efficacy may decline over time, and the stressful CW environment may limit ecosystem services. Continued research is needed on the best designs for particular contaminants and best practices to maximize the CW lifespan. There is also a place for stand-alone RW, particularly when they are situated where wetlands once existed. Careful design and early successional management have proven to be quite successful in yielding a high-quality and self-sustaining wetland habitat. Stressful landscape matrices can make an invasion trajectory a concern; further research on successional variability and best management practices is needed.

Despite their individual success, CW and RW can offer only partial wetland functions and services to degraded landscapes. This is due to differences in objective, design, and management, but also a result of the ecological attributes of these two types of replacement wetlands. In the parlance of EP Odum, CW share some characteristics with early successional and stressed ecosystems [162,163]. They are dominated by r-type species, with simple trophic and habitat structure. They feature rapid nutrient turnover of predominantly extrabiotic nutrients. Additionally, CW are high-management systems, owing to their relatively low stability and narrow operating parameters. RW share some of these attributes in early succession, but, as they mature, RW ideally become more diverse and trophically complex, with a high degree of spatial heterogeneity. Nutrients in a mature RW are predominantly intrabiotic, and organic detritus play an important role in both nutrient and trophic dynamics [162,163]. As a combination of early and late successional systems—or high- and low-stress systems—ICW have the potential to offer a synergistic relationship between CW and RW (Table 4). Modular, multicell designs offer options for coupling CW cells with RW habitat, thereby providing both wastewater treatment, long-term habitat development, functional diversity, and ancillary ecosystem services.

Table 4. Comparison of idealized ecosystem characteristics for constructed wetlands for wastewater treatment (CW), young restored wetlands (RWy), mature restored wetlands (RWm), and integrated constructed wetlands (ICW). Attributes adapted from Odum [162,163].

	CW	RWy	RWm	ICW
Hydrologic regime	closed	open	open	open
Ecological Stress	high	low	low	moderate
Inorganic nutrients	extrabiotic	extrabiotic	intrabiotic	extra/intra
Dominant life strategy	r	r	r, K	r, K
Trophic structure	simple	simple	complex	complex
Habitat heterogeneity	simple	simple	complex	complex
Nutrient exchange rate	rapid	rapid	slow	moderate
Role of detritus	low	low	high	high
Stability	low	low	high	moderate
Temporal variability	low	high	high	moderate
Management effort	high	high	low	moderate
Ecosystem services	narrow	broad	broad	broad

While the ICW concept has been successfully demonstrated around the world, it is still in its infancy. There remains a disconnect between CW and RW in terms of theory,

experimental design, management, and evaluation; this gap will need to be bridged for ICW to become mainstream. Further, there is a need for more long-term, landscape-level studies on the integration of wetland services into the socioecological matrix, and for a cost–benefit analysis of wetland ecosystem services in these different systems. Finally, methods for the holistic evaluation of ecosystem services as they apply to ICW need to be refined and standardized to facilitate assessment within and among sites. These goals are attainable, and indeed are already being achieved by scholars and practitioners who seek to connect the built and natural environments for the benefit of both.

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