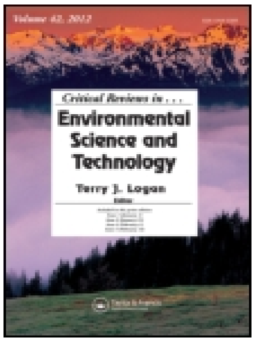

Critical Reviews in Environmental Science and Technology

ISSN: 1064-3389 (Print) 1547-6537 (Online) Journal homepage: http://www.tandfonline.com/loi/best20

Constructed Wetlands for Management of Urban Stormwater Runoff

Piyush Malaviya & Asha Singh

To cite this article: Piyush Malaviya & Asha Singh (2012) Constructed Wetlands for Management of Urban Stormwater Runoff, Critical Reviews in Environmental Science and Technology, 42:20, 2153-2214, DOI: 10.1080/10643389.2011.574107

To link to this article: http://dx.doi.org/10.1080/10643389.2011.574107

Accepted author version posted online: 20 Oct 2011.
Published online: 20 Oct 2011.

Critical Reviews in Environmental Science and Technology, 42:2153–2214, 2012

ISSN: 1064-3389 print / 1547-6537 online
DOI: 10.1080/10643389.2011.574107

Constructed Wetlands for Management of Urban Stormwater Runoff

PIYUSH MALAVIYA and ASHA SINGH
Department of Environmental Sciences, University of Jammu, Jammu, J&K, India

In urban and suburban areas, stormwater runoff has been identified as a significant contributor to water quality impairment. The quality of the stormwater and the characteristics of the pollutants present are dependent on the types of surfaces the stormwater encounters. The scientific research within this field has been focused on pollutants such as nutrients, heavy metals, organic and particulate matter, herbicides, and polycyclic aromatic hydrocarbons. These constituents may pose risks to life forms along with technical and aesthetic problems and thus require implementation of management practices for mitigating stormwater pollution. Structural best management practices are now commonplace for stormwater management in urban developments and range from simple filter strips, to the latest green technology methods such as constructed wetlands. Physicochemical and biological properties of wetlands provide many positive attributes for remediating contaminants. Specifically, wetland stormwater treatment areas offer the advantages of water storage and peak-flow attenuation, nutrient cycling and burial, metal sequestration, sediment settling, and breakdown of organic compounds along with certain ancillary benefits such as recreational facilities as well as functioning as a wildlife habitat. In this context, the authors summarize information on stormwater and different structural best management practices used for stormwater treatment. They critically examine the potential of constructed wetlands for stormwater treatment by looking at the present research initiatives toward execution of this technology. Future considerations in choosing constructed wetlands as stormwater treatment systems are also highlighted by discussing

Address correspondence to Piyush Malaviya, Department of Environmental Sciences, University of Jammu, Jammu-180 006, J&K, India. E-mail: piyushmalaviya@rediffmail.com

benefits and inadequacies as well as economic facets of constructed wetlands.

KEY WORDS: macrophyte, metal, pollution, stormwater, wetland

1. INTRODUCTION

Proliferation of urban sprawl has caused dilapidation of receiving waters owing to amplification of point as well as nonpoint source (NPS) pollutants. Unlike point source pollutants, that could be substantially reduced by applying various conventional treatment technologies, NPS pollutants remain a disquieting trouble, as they are dispersed, making collection and centralized treatment exigent. Among NPS of pollution, urban stormwater runoff (i.e., the water that originates when precipitation from rain or snowmelt flows over the ground; U.S. Environmental Protection Agency [EPA], 2003), represents one of the great challenges of modern water pollution management, and worldwide, it is a principal contributor to impairment of water bodies. In addition to its highly unsteady inflows with much greater volumes, a variety of contaminants are present in stormwater, including physical objects, from large debris to microscopic particles, and chemical constituents, both dissolved and immiscible, exhibiting technological complexities in its management (National Research Council [NRC], 2008).

Although stormwater has long been regarded as a major culprit in urban flooding, only in the past 30 years policymakers have appreciated the significant role stormwater plays in the deterioration of urban watersheds. This rise in recognition has led to a search of different effectual management programs for stormwater runoff (NRC, 2008). As the traditional use of end-of-pipe control technologies and automated effluent monitors used for point source pollutants do not work for the episodic and variable loading of pollutants in stormwater, structural best management practices (BMPs) such as filter strips and swales, infiltration systems, storage facilities, and alternative road structures (Eriksson et al., 2007b) act as crucial tools for stormwater management. Among all these BMPs, phytopurification or green technologies are gaining increased popularity due to their cost effectiveness especially suitable for developing countries. The most common phytopurification technology involves constructed wetland (CW) treatment systems, which have been successfully used for the treatment of a wide range of wastewaters (cf. municipal sewage, landfill leachates, industrial wastewaters); however, the treatment of stormwater runoff in CWs is still at its infancy (Sundaravadivel and Vigneswaran, 2001). CWs reduce nutrients, organic substances, pathogens, and various heavy metals present in stormwater runoff (Langergraber et al., 2008). Keeping in view the importance of CWs for stormwater treatment, in the present article we critically review the research pertaining to stormwater and its treatment by CWs, in addition to providing a brief description about different BMPs employed for stormwater treatment.

2. URBAN STORMWATER POLLUTION: A GLOBAL SCENARIO

Urban stormwater is the term used to describe runoff associated with a rain or snow storm that can be measured in a downstream river, stream, ditch, gutter, or pipe shortly after the precipitation has reached the ground (NRC, 2008). It also includes water that has percolated into the ground but nonetheless reaches a stream channel relatively rapidly (typically within a day or so of the rainfall), contributing to the high discharge in a stream that commonly accompanies rainfall. This diffuse, NPS pollutant load consists of litter, debris, and sediment as the visually apparent components, and nutrients, coliforms, heavy metals, and toxic chemicals (e.g., polycyclic aromatic hydrocarbons [PAHs], polychlorinated biphenyls, organochlorines) as a hidden component (Taebi and Droste, 2004). These stormwater constituents may represent risks to exposed humans, animals, or plants, as well as technical and aesthetic tribulations. For instance, suspended solids can have a detrimental impact on aquatic environments, by smothering aquatic plants, silting waterways, resulting in an increase in water turbidity and reduced light penetration. High organic content results in high biochemical oxygen demand (BOD) and oxygen depletion, which can cause fish kills (Greenway, 2000). Nutrients such as nitrogen and phosphorus are also often found in stormwater in high concentrations and play an important role in algal blooms in open water bodies (Kasper and Jenkins, 2007). Pathogenic organisms are washed into swimming areas and create health hazards, often making beach closures necessary. Debris, plastic bags, bottles, and cigarette butts washed into water bodies can choke, suffocate, or disable aquatic life such as ducks, fishes, turtles, and birds. Likewise, hazardous wastes such as insecticides, pesticides, paint, solvents, used motor oil, and other auto fluids present in stormwater can poison aquatic life (U.S. EPA, 2003).

Urban development results in an increase in the amount of impervious areas, such as pavements, roads, or roofs within any defined catchment. Consequently, the infiltration capacity of the catchment is reduced and the volume of stormwater runoff increases, which leads to higher flood peaks in urban creeks and rivers (Kasper and Jenkins, 2007). Structural drainage controls then bring the urban runoff to a point of discharge into the receiving water body. In this manner, the diffused pollution generated and accumulated over a wide area is transformed into a point source of pollution upon entry into the aquatic environment (Taebi and Droste, 2004). Unable to handle the increased water volume and flow, these water bodies often experience eroded banks, incised channels, loss of habitat and aquatic life, and increased flooding and property damage. The quality of the stormwater and the characteristics (physical, chemical, or microbial) of the pollutants are dependent on the types of surfaces the stormwater encounters (e.g., roads, parking lots, roofing materials, recreational areas). For example, lawns appear to have the highest runoff concentration of phosphorus and total Kjeldahl nitrogen (TKN) while streets have high runoff concentrations of total suspended solids (TSS) and many metals (Steuer et al., 1997). The major sources of stormwater runoff include releases from building materials and other constructions, traffic-related discharges (e.g., exhaust from vehicles, tires, brakes, road material, painting markers on roads, use of deicing agents), human activities (e.g., application of fertilizers and pesticides, leaching and corrosion of pollutants from exposed materials, washing of cars, disposal of paint, oil in the stormwater sewer), and wet and dry air depositions (Eriksson et al., 2007a).

Stormwater runoff scours contaminants mainly from impervious surfaces, where pollutants are deposited during dry periods, and transports them to receiving waters such as rivers, lakes, and estuaries (Hwang and Foster, 2006). Persistent pollutants present in stormwater runoff are particularly damaging when short duration intense summer storms follow a long dry period during which these pollutants have accumulated on the road surface and in the drainage system itself. A sudden flush of road drainage can harm receiving water ecology. More extensive rainfall would have less impact due to the greater dilution of pollutants in the runoff (Mungur et al., 1995). Though diluted, the pollutant impact and shock load associated with stormwater runoff is significantly higher than the secondary treated domestic sewage (Goonetilleke et al., 2005).

Various factors such as rainfall characteristics (intensity and depth), antecedent dry days before individual storm, and the specific activities in the catchment together determine the amount of pollutants being washed off during a particular rain/storm event (Chui, 1997; Davis et al., 2001; Granier et al., 1990). However, some elements such as As, Cr, and Cu did not follow the phenomenon, which could be due to the difference in solubility characteristics of these elements (Sansalone and Buchberger, 1997). In a study conducted by Brezonik and Stadelmann (2002), the event mean concentrations of dissolved phosphorus (DP), chemical oxygen demand (COD), TKN, nitrate plus nitrite–nitrogen (NN), and total nitrogen (TN) were found to be negatively correlated with precipitation amount. Further, all variables in their study except soluble reactive phosphorus (SRP) and Pb were negatively correlated with rainfall duration, which suggested that long storms generate more diluted runoff. Likewise, Maniquiz et al. (2010) observed that the rainfall variables did not display significant correlations to pollutant event mean concentrations (EMCs) and mostly were negatively correlated. The strongest relationships were observed for organics: biological oxygen demand (BOD; $r = -0.44$), COD ($r = -0.30$), and dissolved organic carbon (DOC; $r = -0.29$). The same observation was found in the case of rainfall duration

(RAINDUR) and average rainfall intensity (AVGINT), where most EMCs were either negatively correlated or the coefficients were low. Due to the minimal interparameter correlations between EMC and rainfall variables, it was hypothesized that big rainfalls with longer duration and high intensities produce more dilute runoff. However, the weak correlations also explain that not only rainfall variables contribute to runoff pollutant loads and EMCs, while other factors should also be considered. In another study, consistently negative response of DOC concentration to increased flow in wetland-dominated catchments suggested a dilution effect across peatlands (Eimers, 2008).

Stormwater runoff from urbanized watersheds has received increasing attention from the public and scientific community in recent years because it is perceived to be a large source of pollutants to water bodies (U.S. EPA, 1995). Throughout the United States, NPS pollution is a major contributor to the impairment of receiving waters (U.S. EPA, 2002). In Southern California, despite separate stormwater and sanitary sewer systems, the stormwater runoff from urbanized watersheds has contributed substantial loadings of a variety of constituents to receiving water environments (Schiff, 1997). For instance, the cumulative loads of Pb and Zn from all of the urbanized watersheds in the Southern California Bight to the coastal oceans were estimated to be 39 and 316 metric tons during the years of 1994 and 1995, respectively. These inputs represent over half of the combined mass emissions from all sources, which include traditional point sources such as publicly owned treatment works, industrial facilities, and power generating stations (Schiff et al., 2000).

The U.S. Environmental Protection Agency reported that 35% of assessed rivers and streams (in length) and 44% of assessed estuaries (in area) were impaired for one or more designated uses (e.g., aquatic life support, fish consumption, drinking water supply) mainly due to stormwater runoff, including storm sewer overflows, and agricultural runoff (U.S. EPA, 2000). Of the water bodies that have been assessed in the United States, urban runoff is responsible for about 38,114 miles of impaired rivers and streams, 948,420 acres of impaired lakes, 2,742 square miles of impaired bays and estuaries, and 79,582 acres of impaired wetlands. Urban stormwater is listed as the primary source of impairment for 13% of all rivers, 18% of all lakes, and 32% of all estuaries (NRC, 2008). Walker et al. (1999) reported that urban stormwater runoff alone ranks as the third most common source for rivers in the United States.

The Selected Stormwater Priority Pollutants (SSPP) list derived from the DayWater Project consists of 25 SSPPs that are shown in Table 1. It includes the following categories: basic water quality parameters (6), metals (7), PAHs (3), herbicides (4), and miscellaneous organic compounds (5; Eriksson et al., 2007b). More specifically, these parameters may be used to evaluate the potential for environmental problems related to eutrophication (elevated nutrients loads), oxygen depletion (high organic matter inputs), aesthetical problems and erosion (e.g., high concentrations of suspended solids resulting in

TABLE 1. List of selected stormwater priority pollutants (indicator parameters) (Eriksson et al., 2007b)

Type	Name
Basic parameters	Biochemical oxygen demand
	Chemical oxygen demand
	Suspended solids
	Phosphorus
	Nitrogen
	pH
Metals	Chromium as chromate
	Cadmium
	Platinum
	Copper
	Nickel
	Lead
	Zinc
PAH	Benzo[a]pyrene
	Naphthalene
	Pyrene
Herbicides	Phenmedipham
	Pendimethalin
	Terbutylazine
	Glyphosate
Miscellaneous	Nonylphenol ethoxylates and degradation products (e.g., nonylphenol)
	2,4,4′-Trichlorobiphenyl (Polychlorinated biphenyl 28)
	Di(2-ethylhexyl) phthalate
	Methyl *tert*-butyl ether
	Pentachlorophenol
Bacterial indicators	Fecal coliforms (*Escherichia coli*,[b] *Enterococci* sp.[b]) pathogens (*Pseudomonas aeruginosa*,[a] *Staphylococcus aureus*,[a] *Clostridium perfringens*[b])

[a]Oliveri et al. (1977). [b]Characklis (2005).

significant turbidity and damage to flora and fauna), direct toxicity problems (change of pH resulting in altered speciation and complexation of ammonia and metals), and long-term environmental risks (e.g., changes in the aquatic community due to elevated loads of organic matter; Eriksson et al., 2007b).

Heavy metals in the urban environment predominantly enter stormwater by the wash-off of atmospherically transported material, such as vehicle emissions, and by the dissolution of common building materials such as concrete, galvanized materials, bricks, and tiles (Davis et al., 2001). As all the heavy metals are inherently persistent, this characteristic is not enough for discrimination and hence, the seven metals were selected for inclusion following an evaluation of factors such as their level and mode of toxicity (i.e., highly acute or chronic) as well as being representative of a range of sources (Eriksson et al., 2007b). Walker et al. (1999) summarized the concentrations of heavy metals in urban runoff and reported them to be in the following concentration ranges: Cu (0.00006–1.41 mg L^{-1}), Pb

(0.00057–26.0 mg L^{-1}), and Zn (0.0007–22.0 mg L^{-1}). Platinum is included as a representative of the platinum group elements (PGE; i.e., platinum, palladium, and rhodium) because of its presence in vehicle exhaust catalysts. Lead was previously used as an antiknock agent in petrol but the primary sources are now lead fishing sinkers and ammunition (Sörme et al., 2001).

The group of PAHs identified as potentially present in stormwater runoff consisted of more than 50 compounds of which seven were found to be potentially hazardous with respect to the water phase and 35 in the solid phase (Eriksson et al., 2007b). Two PAH compounds that represented potential hazards in the solid phase were pyrene (4 aromatic rings) and benzo[a]pyrene (5 aromatic rings and a known carcinogen), and the 2-ring compound naphthalene constituted a potential hazard in the water and solid phases. Out of the large number of herbicides defined as potential stormwater pollutants, 23 were found to be used in large quantities in Europe, and 14 of these were identified as potential stormwater priority pollutants. Out of these 14 herbicides, three herbicides (pendimethalin, phenmedipham, and terbutylazine) were selected as SSPPs owing to there potential hazards in the water and solid phases with the additional selection of glyphosate due to its prevalence in monitoring programs (Eriksson et al., 2007b).

Another five compounds (pentachlorophenol [PCP], polychlorinated biphenyl-28 [PCB-28], di (2-ethylhexyl) phthalate [DEHP], nonylphenol ethoxylates [NPEO], and methyl *tert*-butyl ether [MTBE]) were selected from the group of miscellaneous xenobiotic organic compounds (XOCs) as representatives reflecting a range of sources, impacts, and political/public concerns in relation to XOCs. PCP is used as a wood preservative and is persistent to degradation, highly toxic to aquatic organisms and causes long-term adverse effects. PCBs are well known for their persistence and for their diffuse leakage from buildings and insulated electrical equipment (e.g., transformers). PCB-28 was selected as the representative PCB being the most water soluble of the seven PCBs confirmed to be present in stormwater runoff (Eriksson, 2002). DEHP is used as a plasticizer in tubing and is ubiquitous in the technosphere. It is one among 356 registered different phthalates and 28 high production volume chemicals (HPVC; European Chemical Substances Information System, 2005). Nonionic detergents such as NPEO are used as car wash detergents and have been shown to release the ethoxylate side chain on degradation producing compounds (e.g., nonylphenol), which have been confirmed to have endocrine disrupting effects. Likewise, Methyl *tert*-butyl ether was also included due to its presence in petrol, its high water solubility, and mobility in soils and because of concerns over its use in relation to odor (Eriksson et al., 2007b).

Even though, not specified in the SSPP list derived from the DayWater project, the presence and distribution of microorganisms in stormwater and wet weather discharges also need to be considered. This can be achieved by either using traditional parameters (total fecal coliforms, *E. coli*) or more recently identified indicators of fecal pollution (fecal sterols). Bacterial indicators such as fecal coliforms and pathogenic organisms are also specified as stormwater priority pollutants and are given in Table 1 (Characklis, 2005; Oliveri et al., 1977). Besides, recently, the European Commission supplemented the list with an additional eight compounds or compound groups. These eight new compounds are all chlorinated industrial products including pesticides (e.g., p,p′-DDT, aldrin, dieldrin) and solvents (e.g., tetrachloromethane, trichloroethylene; Eriksson et al., 2007b). On account of the presence of all the previously mentioned toxic contaminants, proper management practices are needed to bring down the concentration of pollutants in stormwater before their ultimate discharge into the receiving waters.

3. STRUCTURAL BEST MANAGEMENT PRACTICES FOR STORMWATER

Owing to the shift of the world's population to urban settings accompanied by continued landscape alteration to accommodate booming population, the magnitude of the stormwater problem is expected to aggravate further. As individual controls on stormwater discharges are inadequate as the sole solution to stormwater pollution in urban watersheds, **s**tormwater control measures (SCMs) or BMPs implementation needs to be designed as a system, integrating structural and nonstructural BMPs and incorporating watershed goals, site characteristics, land use development, erosion and sedimentation controls, aesthetics, monitoring, and maintenance. Stormwater cannot be adequately managed on a piecemeal basis due to the complexity of the hydrologic and pollutant processes and their effects on habitat and stream quality. Nonstructural SCMs or BMPs such as product substitution, better site design, conservation of natural areas, and watershed and land-use planning can dramatically reduce the volume of runoff and pollutant load from a new development. Such SCMs should be considered first before structural practices. For example, lead concentrations in stormwater have been reduced by at least a factor of four after the removal of Pb from gasoline. Likewise, not creating impervious surfaces or removing a contaminant from the runoff stream simplifies and reduces the reliance on structural SCMs (NRC, 2008).

For reduction of contaminants from stormwater runoff to acceptable level before being discharged into water bodies, numerous conventional and emerging methods (e.g., ion exchange, electrolyte or liquid extraction, electrodialysis, precipitation, reverse osmosis) could be applied. However, most of the available physicochemical technologies are either economically unfavorable or too technically complicated (Brown et al., 2000a; Brown et al., 2000b). Contrary to standard domestic or industrial wastewater treatment technologies, stormwater treatment systems have to be robust to highly variable flow rates and water quality variations (Scholz and Lee, 2005). To resolve these problems, and to reduce the urban runoff peak flows as well as the amount of stormwater based pollutants entering the receiving water environment, structural BMPs or sustainable urban drainage systems (SUDs) are widely used nowadays (Eriksson et al., 2007b).

Structural BMPs (Table 2) can be categorized into four main groups: filter strips and swales, infiltration systems (soak ways, infiltration trenches and infiltration basins), storage facilities (detention basins, retention ponds, lagoons, CWs, storage tanks, roof storage), and alternative road structures (porous paving, porous asphalt surfaces; Scholes et al., 2005). Depending on the nature of the stormwater pollutants being targeted and available space, a specific type of structural BMP is applied. In many cases, a sequence

TABLE 2. Descriptions of different structural BMPs (Scholes et al., 2005)

System type	Description
Filter strip	Grassed or vegetated strip of ground that stormwater flows across
Swales	Vegetated broad shallow channels for transporting stormwater
Soakaways	Underground chamber or rock-filled volume; stormwater soaks into the ground via the base and sides; unplanted but host to algal growth
Infiltration trench	A long thin soakaway; unplanted but host to algal growth
Infiltration basin	Detains stormwater above ground, which then soaks away into the ground through a vegetated or rock base
Detention basin	Dry most of the time and able to store rainwater during wet conditions; often possess a grassed surface
Retention pond	Contain some water at all times and retains incoming stormwater; frequently with vegetated margins
Sedimentation tank	Symmetrical concrete structure containing appropriate depth of water to assist the settling of suspended solids under quiescent conditions
Extended detention basin	Dry most of the time and able to store rainwater during wet conditions for up to 24 hr; grassed surface and may have a low basal marsh
Filter drains	Gravelled trench systems where stormwater can drain through the gravel to be collected in a pipe; unplanted but host to algal growth
Lagoons	Pond designed for the settlement of suspended solids; fringing vegetation can sometimes occur
Constructed wetlands	Vegetated system with extended retention time
(a) Subsurface flow	Typically contain a gravel substrate, planted with reeds, through which the water flows
(b) Surface flow	Typically contain a soil substrate, planted with reeds, over which the water flows
Porous asphalt	Open graded powdered/crushed stone with binder: high void ratio; no geotextile liner present
Porous paving	Continuous surface with high void content, porous blocks or solid blocks with adjoining infiltration spaces; an associated reservoir structure provides storage; no geotextile liner present; host to algal growth

(often called a treatment train) of measures may be used. For example, linear infiltration or biofiltration systems may be placed within the urban streetscape that may pass on water to a sedimentation basin and CW, before discharging to an ornamental pond (Lloyd et al., 2001).

Heightened concern during the 1980s about the deleterious effects of urbanization on watersheds of densely populated Mid-Atlantic States in the United States has resulted in adopting a variety of stormwater management regulations intended to mitigate the damage by these states. Many states such as Delaware, New Jersey, and Pennsylvania allow a range of practices and regulatory approaches from extended detention in Pennsylvania through the latest green technology methods promoted in Delaware to the recharge, quality, and peak reduction approaches of New Jersey (Balascio and Lucas, 2009).

Although, all the previously mentioned BMPs have been devised for attenuating hydrologic flows and removing contaminants from stormwater runoff, the major constraint continues to be its diffuse delivery, which necessitates extensive regional infrastructure in conjunction with a system that allows degradation as well as removal of pollutants present in stormwater runoff and thus, among all the structural BMPs, CWs hold a promising answer to these problems. In comparison with other technologies, a CW is a sustainable means of treating stormwater and also proves to be more economical (in terms of construction and maintenance) and energy efficient (Kadlec et al., 2000). Furthermore, wetlands enhance biodiversity and are less susceptible to variations of loading rates (Cooper et al., 1996).

4. CONSTRUCTED WETLANDS: AN ECO-TECHNOLOGICAL ADVANCEMENT

Using plants to purify wastewater has always fascinated researchers and holds instinctive appeal to the general public as well. Consequently, natural wetland systems, that use the ability of plants for uptaking or degrading the pollutants, were developed. Studies have shown that natural wetlands characteristically portrayed as kidneys of the landscape (Lai and Lam, 2009) are able to provide high levels of wastewater treatment. Physicochemical characteristics prevailing in natural wetlands provide many positive attributes for remediating contaminants. However, there has been concern over possible harmful effects of toxic materials and pathogens in wastewaters and long-term degradation of natural wetlands due to additional nutrient and hydraulic loadings from wastewater (Kivaisi, 2001). New regulations in the United States, aiming to protect natural wetlands, now restrict their use for stormwater runoff (Debusk et al., 1996). The remarkable ability of wetlands to remove contaminants from water, however, makes them a desirable choice of treatment method (Carleton et al., 2001; Haberl et al., 2003; Kivaisi, 2001). CWs that are designed to mimic natural wetland systems offer a compromise between preservation of existing natural systems and exploitation of the unique biological and physicochemical processes of wetlands to remove low levels of contamination from large volumes of stormwater runoff.

CWs are among the recently proven (Vymazal, 2011) efficient technologies for wastewater treatment. These are engineered treatment systems that encompass a plurality of treatment modules including biological, chemical, and physical processes, which are all akin to processes occurring in natural treatment wetlands (Babatunde et al., 2008). They are intended to take advantage of many of the same processes that occur in natural wetlands, but do so within a more controlled environment. They are created from a nonwetland ecosystem or a former terrestrial environment, mainly for the purpose of pollutant removal from wastewater.

Major advantages of CWs include:

- They operate on ambient solar energy and require low external energy input.
- They achieve high levels of treatment (Carleton et al., 2001) with little or no maintenance, making them especially appropriate in locations where no infrastructure support exists.
- They are relatively tolerant to shocks induced by hydraulic and pollutant loads that ensure the reliability of treated wastewater quality.
- Unlike the conventional treatment systems, no specific design and life period is generally prescribed for CWs and as such they tend to have increased treatment capacity over time, by setting up feedback loops that result in self-repairing systems.
- Wetland vegetation generates oxygen and consumes carbon dioxide, thereby helping in improving air quality and fight global warming. While the wetlands cover only 6% of the world's surface, they are estimated to hold 771 gigatons of greenhouse gases, or 10–20% of the globe's terrestrial carbon (Anonymous, 2008). No doubt, the emission of N_2O and CH_4 from CWs is high, however, their global influence is not significant, as Teiter and Mander (2005) established that even if all global domestic wastewater will be treated by wetlands, their share in the trace gas emission budget would be less than 1%. Moreover, greenhouse gas (GHG) fluxes are higher in unplanted and nonaerated treatments and, thus, the addition of artificial aeration reduces CH_4 fluxes (Maltais-Landry et al., 2009).
- Wetland vegetation provides indirect benefits such as green space, wildlife habitats, and recreational and educational areas (Sundaravadivel and Vigneswaran, 2001).

Although initially designed and used for domestic wastewater treatment, through the efforts of research and operation for over 50 years, CWs have now been successfully used for environmental pollution control by treating a wide variety of wastewaters including industrial effluents, urban and agricultural stormwater runoff, animal wastewaters, leachates, sludges, and mine drainage (Kadlec et al., 2000; Scholz and Lee, 2005). Significant advances have been made in the engineering knowledge of creating CWs that can closely imitate the specialized treatment functions that occur in the natural wetland ecosystems. Among different natural treatment systems, various advantages such as simplicity of design and lower costs of installation, operation, and maintenance offered by CWs make them an appropriate alternative for developed and developing countries (Sundaravadivel and Vigneswaran, 2001). However, these systems have not found widespread use, due to lack of awareness and expertise in developing the technology on a local basis.

4.1 Classification of Constructed Wetlands

According to the life form of the dominating macrophytes, CWs for wastewater treatment may be classified into three types (Brix, 1994): (a) free-floating macrophyte-based systems, (b) submerged macrophyte-based systems, and (c) rooted emergent macrophyte-based systems. Another classification of wetlands based upon the water flow regime of different rooted emergent systems distinguishes CWs into (a) surface flow systems, (b) horizontal subsurface flow systems, (c) vertical subsurface flow systems, and (d) hybrid systems.

Surface flow wetlands (SF) are densely vegetated by a variety of plant species and typically have water depths less than 0.4 m. Subsurface flow wetlands (SSF) use a bed of soil or gravel as a substrate for the growth of rooted emergent wetland plants. Mechanically pretreated wastewater flows by gravity, through the bed substrate, where it contacts a mixture of facultative microorganisms living in association with the substrate and plant roots (Haberl et al., 2003).

Depending on the direction of flow of the wastewater, SSF wetlands can be either horizontal flow type or vertical flow type. In horizontal SSF systems, the substrate is maintained water saturated through continuous application of wastewater. The bed depth of horizontal SSF wetlands is typically less than 0.6 m and the bottom of the bed is sloped to minimize flow above the surface. In vertical SSF wetlands, wastewater is applied through different arrangements of wastewater feeding and collection mechanisms to maintain a vertical direction of flow. This is achieved either by intermittent wastewater application or by burying inlet pipes into the bed at a depth of 60–100 cm. The total depth of bed is in the range of 2–3 m. Because the wastewater infiltrates through the substrate bed, this type of wetland is also called an infiltration wetland (Sundaravadivel and Vigneswaran, 2001). However, subsurface flow systems are susceptible to clogging; therefore, they are not recommended for wastewater with a high concentration of total solids (Hammer, 1994). Recently, the combinations of various types of CWs incorporated into a single system referred to as hybrid CW systems have been used to enhance the treatment effect, especially for nitrogen (Cui et al., 2009).

The use of surface water treatment wetlands (reed beds) to remove various pollutants from water began with the work of K. Seidel at the Max Planck Institute in Germany in 1960s (Seidel, 1961, 1964, 1965a, 1965b, 1966). The first full-scale free water surface (FWS; surface flow) CW was built in the Netherlands to treat wastewaters from a camping site during the period of 1967–1969. Within several years, there were about 20 FWS CWs built in the Netherlands. However, FWS CWs did not spread throughout the Europe and CWs with horizontal subsurface flow (HF CWs) became the dominant type of CWs in Europe. The first full-scale HF CW was built in 1974 in Othfresen in Germany. The early HF CWs in Germany and Denmark used predominantly heavy soils, often with high content of clay. These systems had a very high treatment effect but because of low hydraulic permeability, clogging occurred shortly and the systems resembled more or less FWS systems. In late 1980s in the United Kingdom, soil was replaced with coarse materials (washed gravel) and this setup has been successfully used since then (Vymazal, 2005).

In the 1980s, treatment technology of CWs rapidly spread around the world. In 1990s, increased demand of nitrogen removal from wastewaters led to more frequent use of vertical flow (VF) CWs, which provide higher degree of filtration bed oxygenation and consequent removal of ammonia via nitrification. In late 1990s, the inability to produce simultaneously nitrification and denitrification in a single HF or VF CW led to the use of hybrid systems that combined various types of CWs. Though the concept of combination of various types of filtration beds was suggested by Seidel in the 1960s but only few full-scale systems were built (e.g., Saint Bohaire in France or Oaklands Park in England) in the 1980s and early 1990s. At present, hybrid CWs are commonly used throughout Europe as well as other parts of the world. The VF–HF combination is the dominant setup but the HF–VF combination is also used along with FWS CWs in hybrid systems. In the 1970s and 1980s, CWs were nearly exclusively built to treat domestic or municipal sewage. However, since the 1990s, the CWs have also been used for all kinds of wastewaters including landfill leachate, runoff (e.g., urban, highway, airport, agricultural), food processing (e.g., winery, cheese and milk production), industrial (e.g., chemicals, paper mill, oil refineries), agriculture farms, and mine drainage (Vymazal, 2005).

4.2 Mechanisms Operating in Constructed Wetlands for Pollutant Removal

CW systems may be converted natural or constructed shallow ecosystems designed to capitalize on intrinsic physical, chemical, and biological processes for the primary purpose of water quality improvement (Imfeld et al., 2009). These are characterized by the presence of vegetation adapted to saturated conditions, sediments and soil, microbial biomass, and an aqueous phase loaded with the pollutants. Wetland treatment systems are effective in treating organic matter, nitrogen, and phosphorus, and additionally for decreasing the concentrations of heavy metals, organic chemicals, and pathogens (Haberl et al., 2003). In several aspects, CWs are complex bioreactors characterized by considerable fluxes of material and energy governing chemical reactions over spatial and temporal gradients. These fluxes are particularly pronounced in certain zones such as the rhizosphere and permit the maintenance of thermodynamic nonequilibrium conditions and enable various reactions with exergonic free energy changes to occur (Hanselmann, 1991). In CWs, the biogeochemical reactions affecting contaminant removal primarily depend on two types of processes simultaneously occurring at different scales: (a) the variety of coexisting redox processes at the wetland system scale and (b) the processes occurring at the rhizosphere scale (Imfeld et al., 2009). The relative importance of a particular process can vary significantly, depending on the contaminant being treated, the wetland type (e.g., SSF or SF, HF, VF) and operational design (e.g., retention time), the environmental conditions, the type of vegetation within the system, and the nature of soil matrix.

Several elimination pathways may occur in a complex CW system. Kadlec (1992) listed volatilization, photochemical oxidation, sedimentation, plant uptake, sorption, biological degradation, and metabolic transformation as the major processes affecting the contaminant loads in wetlands. The mere reduction of contaminant concentration within the aqueous phase via nondestructive partitioning processes, such as sorption and volatilization, may only relocate the contamination. Therefore, the mass transfer of contaminants from the aqueous phase to other compartments (soil and atmosphere) has to be considered carefully when evaluating potential environmental hazards (Imfeld et al., 2009). Different mechanisms operating in CWs for improvement of water quality are described subsequently.

4.2.1 Nondestructive Processes

4.2.1.1 Volatilization and Phytovolatilization. In addition to direct contaminant emission from the water phase to the atmosphere (volatilization), some wetland plants take up contaminants (e.g., benzene, toluene, ethylbenzene, xylene [BTEX]) and MTBE through the root system and transfer them to the atmosphere via their transpiration stream, in a process referred to as phytovolatilization (Hong et al., 2001; Ma and Burken, 2003). In the case of halophytes, this transfer may also occur via the aerenchymatous tissues (Pardue, 2002). If the atmospheric half-lives of VOCs are reasonably short such as the one for MTBE (three days at 25°C and the toxicological risk is assumed to be low), the water-to-atmosphere contaminant transfer occurring in wetlands may constitute a possible remediation option (Winnike-McMillan et al., 2003). However, volatilization of VOCs may also lead to dispersal of the contaminants in the environment resulting into air pollution. This fact and the lack of reliable risk assessment technology presently discourage the regulatory acceptance of phytoremediation as a strategy for VOCs removal (McCutcheon and Rock, 2001). Phytovolatilization may be of particular relevance in SSF systems, where direct volatilization is restrained due to slow diffusion rates of contaminants through the unsaturated zone as well as laminar flow in water saturated soil zones that may result in relatively low mass transfers. Direct contaminant volatilization is expected to be more pronounced in SF wetlands, as water remains in direct contact with the atmosphere (Kadlec and Wallace, 2008).

4.2.1.2 Plant Uptake and Phytoaccumulation. Phytoaccumulation occurs when the sequestered contaminants such as heavy metals are not degraded in the plant, resulting in its accumulation within the hyperaccumulator plant tissues (Susarla et al., 2002). The potential for plant uptake is highest in SSF CWs due to the increased contact between stormwater and the elaborate root systems of aquatic macrophytes (Scholes et al., 2005).

4.2.1.3 Sorption and Sedimentation. Sorption of a chemical to soil or sediment may result from the physical or chemical adhesion of molecules to the surfaces of solid bodies, or from partitioning of dissolved molecules between the aqueous phase and soil organic matter. During the early stages of CW operation, sorption onto soil substrate will naturally be higher due to the high adsorption capacity of previously unexposed material (Omari et al., 2003). As long as no sorption–desorption equilibrium is reached, the system acts as a sink for the contaminant. After reaching steady-state conditions, contaminants will still be retained by reversible sorption processes. This retention may increase contaminant residence time within the CW and support bioremediation by increasing exposure to degrading microorganisms (Pardue, 2002). Chemical reactions between substances, especially metals, can lead to their precipitation from the water column as insoluble compounds. However, such sorptive processes may also negatively affect the bioavailability of contaminants. The long-term P storage in the wetland can be achieved via peat accumulation and substrate fixation (Healy et al., 2007). During this long-term storage, phosphorus-containing particles settle to the substratum and are rapidly covered by a continuous accumulation of settled sediments. This continuous accumulation of sediments, leave some phosphorous too deep within the substratum to be reintroduced back into the water column later on. The efficiency of long-term peat storage is a function of the loading rate and also depends on the amount of native iron, calcium, aluminum, and organic matter in the substrate (Shatwell and Cordery, 1999). In point of fact, under aerobic conditions, insoluble phosphates are precipitated with ferric iron, calcium, and aluminum. These phosphates are adsorbed onto clay particles, organic peat, and ferric/aluminum hydroxides and oxides (Scholz and Lee, 2005). The capacity for phosphorus adsorption by a wetland, however, can be saturated in a few years if it has low amounts of aluminum and iron or calcium (Richardson, 1985). This is the reason why wetlands along rivers have a high capacity for phosphorus adsorption because as clay is deposited in the floodplain, aluminum and iron in the clay accumulate as well (Gambrell, 1994).

4.2.2 Destructive Processes

4.2.2.1 Phytodegradation. Phytodegradation occurs when a plant has taken up the contaminant into its tissues, and enzymes or other plant-based biomolecules within the plant, act to transform the organic contaminants by either mineralizing them completely to inorganic compounds (e.g., CO_2, H_2O, Cl_2) or degrading them partially to a stable intermediate that is stored in the plant or released in root exudates (McCutcheon and Schnoor, 2003). This enzymatic degradation of organics can happen in root and shoot tissues. Degradation within plant tissues is generally attributed to the plant, but may in some cases involve endophytic microorganisms (Barac et al., 2004).

4.2.2.2 Microbial Degradation. Microorganisms present in the wetland system, including bacteria as well as fungi, coagulate, stabilize, and remove dissolved and colloidal organic matter by converting them into various gases and new tissues (Sundaravadivel and Vigneswaran, 2001). The nature and extent of microbial degradation of contaminants within a CW is expected to be strongly dependent on the physicochemical properties of the contaminant. Additionally, microbial degradation is facilitated by the availability of attachment sites and nutrients within a CW and aerobic and anaerobic processes are enhanced by the occurrence of high contact ratios between stormwater and substrate material. Owing to this reason, microbial degradation is strongly encouraged within SSF CWs (Ellis et al., 2003)

The removal of nutrients and solids in wetlands is facilitated by shallow water (which maximizes the sediment to water interface), high primary productivity, presence of aerobic as well as anaerobic sediments, and accumulation of litter (Mitsch and Gosselink, 1993). In addition, slow water flow causes suspended solids to settle from the water column in wetlands. BOD is reduced by the settling of organic matter and through the decomposition of BOD-causing substances. Coliforms are reduced within wetlands for a number of reasons such as exposure to sunlight, predation, competition for resources, and toxins. In addition, they may be buried beneath sediment or adsorbed (Gersberg et al., 1989). Bactericidal excretions by plants also play a role in removal of pathogens in wetlands, but this is unlikely to be a significant removal process, as it may otherwise prevent biofilm development on the substrate surface. Likewise, exposure to atmospheric gases and sunlight can lead to breakdown of organic pesticides and destruction of pathogens (Sundaravadivel and Vigneswaran, 2001).

Nitrogen (N) removal in CWs is accomplished primarily by physical settlement, denitrification, and plant/microbial uptake. However, plant uptake does not represent permanent removal unless plants are routinely harvested. Phosphorus (P) is removed through short- or long-term storage. Uptake by bacteria, algae, duckweeds, and other macrophytes provides an initial removal mechanism (Kadlec, 1997). However, this is only a short-term P storage as 35–75% of P stored is eventually released back into the water upon dieback of algae and microorganisms (Richardson and Craft, 1993; White et al., 2000). Anaerobic conditions that exist at the soil/water interface may also cause release of P back into the water column.

4.3 Macrophytes as Purifiers in Constructed Wetlands

Wetland plants or aquatic macrophytes are central to wastewater treatment wetlands. Submerged macrophyte systems have plants species that are submerged in the water column and do protrude beyond the water surface. *Isoetes lacustris*, *Lobelia dortmanna*, *Egeria densa*, and *Elodea canadensis* are among the submerged aquatic plant species (Sundaravadivel and Vigneswaran, 2001).

Floating macrophytes are plant species that float on the surface of the water, and do not require a substrate for their growth. Duckweeds (*Lemna* sp., *Spirodela* sp.), water hyacinth (*Eicchornia crassipes*), *Myriophyllum aquaticum*, and *Salvania* sp. are some of the floating macrophytes adapted for wastewater treatment (Sundaravadivel and Vigneswaran, 2001). Free floating plants are superior to the submerged aquatic macrophytes as their removal requires neither extensive filtration equipment nor they produce significant disruption to the water body (Srivastava et al., 2008).

Rooted emergent macrophytes are plants that are generally attached to the substrate in the wetland with leaves extending above the water surface. Reeds (*Phragmites* sp.), cattails (*Typha* sp.), bulrushes (*Scirpus* sp.), and sedges (*Carex* sp.) are common among the emergent aquatic plant species used in treatment wetlands. Unless specifically mentioned otherwise, wetland treatment systems indicate CWs planted with rooted emergent macrophyte species (Sundaravadivel and Vigneswaran, 2001).

The principle in selecting a suitable plant species for use in CW systems depends on the type of wetland design (e.g., surface or subsurface, VF or HF), the mode of operation (e.g., continuous, batch, or intermittent flow), and the loading rate and characteristics of wastewaters (Cui et al., 2009). All the previously cited macrophytes are effective in removing pollutants from wastewaters, but the common reed (*Phragmites australis*) and the reedmace (*Typha latifolia*) are particularly efficient in this context. They have a large biomass both above (leaves) and below (underground rhizome system) the surface of the substrate. Additionally, the subsurface plant tissues grow horizontally and vertically and create an extensive matrix that binds the soil particles and crafts a large surface area for the uptake of nutrients and ions (Shutes, 2001).

Stormwater treatment areas (STAs) incorporate a variety of dominant plant communities ranging from emergent macrophytes to periphytic algae. Although emergent macrophytes show effective removal of contaminants (Shutes, 2001), in Florida it was observed that many shallow aquatic systems dominated by submerged aquatic vegetation (SAV) often show better water quality (clarity, TSS, pH, total phosphorous [TP], and total nitrogen [TN]) than other systems (Canfield and Hoyer, 1992; O'Dell et al., 1995). Submerged macrophytes have also been used in wastewater treatment for nutrient removal (Gu et al., 2001), as there are several advantages of using SAV as opposed to cattail dominated systems as a means for removing P.

First, SAV systems utilize nutrients from both the water column and sediments. Nonrooted submerged macrophytes such as *Ceratophyllum demersum* and *Chara* sp. appear to rely exclusively on dissolved nutrients from the water column (Barko and James, 1997). Moreover, the ability of SAV to occupy most of the water column allows it to remove nutrients from the water column effectively and to minimize effects of hydraulic currents. Second, the high rate of photosynthesis by SAV raises the water column's pH, which in turn may lead to coprecipitation of soluble reactive P (SRP) with $CaCO_3$ (Murphy et al., 1983). Third, a dense SAV community provides extensive surface area for periphyton growth that in turn helps in nutrient removal from the water column. Lastly, an SAV plant community may physically filter, detain, and cause sedimentation of suspended solids that contain organic P or adsorbed inorganic P. A treatment system dominated by SAV is one of several advanced treatment technologies being evaluated by the South Florida Water Management District and Florida Department of Environmental Protection (Knight et al., 2003).

SAV has also been associated with increased releases of P from the sediments of shallow aquatic systems. Two potential processes for increased P cycling in SAV-dominated ecosystems are uptake of sediment P through the roots with further nutrient release after plant tissue decomposition and the altered redox potential in the surface sediments due to high rates of SAV metabolism resulting into releases of iron-bound P under chemically reducing conditions (Knight et al., 2003).

Originally, the basis for employing CWs for wastewater treatment was the ability of water plants to translocate oxygen to their roots and the surrounding wastewater owing to the presence of hollow vessels in the plant tissues. Additionally, within the water column, the stems and leaves of the wetland plants significantly increase surface area for biofilm development. Plant tissues, moreover, are colonized by photosynthetic algae as well as by bacteria and protozoa (Brix, 1997). The extensive rhizosphere of wetland plants provides an enriched culture zone for the microorganisms involved in degradation. It has been reported that, vegetated wetlands are more effective in pathogen removal than those without plant growth, because the plants provide habitat for a variety of microorganisms, some of which (such as zooplanktons) are pathogen predators (Kadlec and Knight, 1996). Predation may be highly efficient in removing pathogens, especially in and around the root zones where zooplanktons are likely to shelter and feed.

Macrophytes can assimilate pollutants in their tissues and provide a surface and an environment for microorganisms to grow (Vymazal, 2002). Moreover, macrophytes create good conditions for the sedimentation of suspended solids and prevent erosion by reducing the velocity of the water in the wetland. The growth of roots within the aggregated sediments helps to decompose organic matter and prevents clogging by creating channels for the water to pass through the intermittently loaded vertical-flow system. The macrophytes transport approximately 90% of the oxygen available in the rhizosphere. This stimulates both aerobic decomposition of organic matter and the growth of nitrifying bacteria (Brix, 1997; Scholz, 2006). However, when compared with microorganisms, macrophytes only play a secondary role in the degradation of organic matters in wetland systems (Stottmeister et al., 2003). Organic matter accumulates in wetlands over time through the annual turnover of macrophytes in terms of leaves and shoots. This organic matter binds heavy metals directly, and provides a carbon and energy source for microbial metabolism. Thus, macrophytes are vital for the long-term functioning of wetlands (Batty, 2003; Scholz, 2006).

The selection of the aquatic plant species is one of the tricky tasks prior to the designing of a treatment facility. Among different macrophytic species, variation in pollutant removal occurs owing to various reasons, including variation among species in biomass, root architecture, and the area of absorptive surface (Skene, 2003), as well as in physiological uptake capacity, growth rate, and effects of roots on the physical and chemical characteristics of the soil medium and the microbial community (Read et al., 2008). Differences among species in their mycorrhizal associations may also account for some of the variations (Smith and Read, 1997). Species are not usually universally effective in removing different pollutants. Hence, consortia of species may be most suitable for CWs to maximize the spectrum of pollutant removal.

4.4 Constructed Wetlands: Management Tools for Wastewaters

CWs are used for treating various wastewater types (e.g., domestic wastewater (Schreijer et al., 1997), acid mine drainage (Howard et al., 1989), agricultural wastewaters (Rivera et al., 1997), and landfill leachate (Trautmann et al., 1989), and for polishing advanced treated wastewater effluents for return to fresh water resources (Gschlobl et al., 1998). To treat municipal and domestic (single house or cluster of houses) wastewaters, HF CWs are commonly used at the secondary and tertiary treatment stages (Vymazal, 2009). In general, HF CWs are not used to treat raw municipal wastewater as the amount of oxygen transported from the roots into zones under the water table is too little to facilitate aerobic processes. Therefore, anoxic and anaerobic processes play the most important role in HF CWs. Additionally, insufficient oxygen supplies results in incomplete nitrification (Langergraber and Haberl, 2001).

A variety of industrial wastewaters has been treated in CWs, such as wastewaters from petrochemical and chemical industries (Dias et al., 2006; Yang and Hu, 2005), the pulp and paper industry (Abira et al., 2005), the textile industry (Bulc et al., 2006; Mbuligwe, 2005), the tannery industry (Calheiros et al., 2007; Dotro et al., 2006), winery and distillery (Grismer et al., 2003; Sheridan et al., 2006), the food processing industry (Khalil et al., 2005; Mantovi et al., 2007), and the abattoir and meat processing industry (Gasiunas et al., 2005). Similarly, wastewaters from various feedlot operations such as pig farms effluents (Kantawanichkul and Somprasert, 2005), fish farm effluents (Schulz et al., 2003), and dairy effluents (Drizo et al., 2006), are commonly treated with free water surface CWs with series of lagoons as pretreatment step (Vymazal and Kröpfelová, 2008).

The removal of fecal indicator bacteria from wastewater by CWs is well documented (Gersberg et al., 1987; Perkins and Hunter, 1999). Reported removal efficiencies for coliforms generally exceed 90% (Kadlec and Knight, 1996) with significantly higher removal in extensively vegetated systems compared with unvegetated systems (Garcia and Becares, 1997; Gersberg et al., 1987). Similarly, removal efficiencies for fecal streptococci by wetlands generally go beyond 80% (Kadlec and Knight, 1996).

CWs are also employed for effective abatement of heavy metal pollution from acid mine drainage, landfill leachate, thermal power plant, municipal, agricultural, refinery, and chlor-alkali industry effluent (Rai, 2008). Concentration of different heavy metals such as Pb and Cd (DeBusk et al., 1996), Cd (Wang et al., 2002), Ni (Zurayk et al., 2002), Cr, Ni, Zn (Hadad et al., 2006), and Fe and Cu (Rai, 2007) were reduced by employing CWs for treatment of heavy metal–containing wastewater. Various other compounds commonly found in wastewaters such as endocrine-disrupting chemicals (EDCs; e.g., phthalates, pesticides, PCBs, dioxins, PAHs, alkylphenols, bis phenols and steroid estrogens; Vymazal, 2009), MTBE (Rubin and Ramaswami, 2001), trichloroethylene (TCE), BTEX, cyanide, herbicides and pesticides such as atrazine (Anderson et al., 2002), metaloxyl, simazine (Wilson et al., 2000), and DDT (Garrison et al., 2000) have been successfully treated by CWs. CWs have also shown promise for cleaning up explosives-related wastes such as TNT, hexahydro-1,3,5-trinitro-1,3,5-triazine (RDX; Best et al., 1997), and perchlorate (Susarla et al., 2000). In addition to these pollutants, urban stormwater is also a major source of aquatic pollution, leading to widespread environmental degradation (U.S. EPA, 2000). However, as compared with the previously discussed pollutants and wastewaters, stormwater management has received little attention. More recently, CWs have been used for the retention and treatment of all kinds of urban wet weather flows, including runoff from airports, parking lots, agricultural runoff, combined sewer overflows, and stormwater in general (Debo and Reese, 2003; Fink and Mitsch, 2005; Hathaway et al., 2010; Revitt et al., 2001; Wang et al., 2009).

5. CONSTRUCTED WETLANDS FOR STORMWATER TREATMENT

The use of CWs represents a relatively new approach for stormwater treatment (Dittmer et al., 2005; Line et al., 2008). The idea of employing wetlands, both natural and manmade, for capturing stormwater runoff and pollutants, has been materialized from an understanding of the role, wetlands naturally play in landscapes (Leibowitz et al., 2000). Specifically, wetland stormwater treatment areas (WSTAs) can provide the services of water storage and peak-flow attenuation (DeLaney, 1995), nutrient cycling and burial (Reddy et al., 1993), metal sequestration (Odum et al., 2000), sediment settling (Kadlec and Knight, 1996), and breakdown of organic compounds (Knight et al., 1999). CWs are particularly suited for treating urban stormwater runoff because they can operate under a wide range of hydraulic loads.

CWs offer a potentially cost-effective, practical option for treatment of these highly pulsed sources of diffuse pollutants before they reach streams, rivers, lakes, and estuaries. There is now a substantial database on the performance of wastewater treatment wetlands with relatively constant inflows (Kadlec et al., 2000), but this is hard to apply directly to systems receiving less organic-enriched drainage discharges with highly variable flow characteristics. In recent years, CW systems for managing highly pulsed urban stormwater flows are developing rapidly (Tanner et al., 2005). Historically, urban stormwater management was only concerned with collecting and distributing stormwater to minimize flooding (Chow et al., 1988). In the United States, amendments to the Clean Water Act addressing the impacts of nonpoint pollution sources on receiving water bodies (Federal Water Pollution Control Act, 1948) has forced stormwater management efforts of many municipalities to focus on providing treatment and modifying the discharge pattern of urban runoff so that it more closely resembles stormwater coming from undeveloped landscapes, as is provided in 40 C.F.R. Section 131b of US Code of Federal Regulations (US Code of Federal Regulations, 1995).

The use of CWs to treat stormwater has become widespread, although the pollutant removal efficiencies of wetlands are often highly variable and heavily dependent on the suitability of the design and implementation (Birch et al., 2004; CWP, 2007; Rushton, 2004; Somes et al., 2000). Studies suggest that wetland performance in treating stormwater is generally a function of inflow or hydraulic loading rate (HLR) and detention time (Dt), which are in turn functions of storm intensity, runoff volume, and wetland size (area and volume; Barten, 1987; Carleton et al., 2001; Meiorin, 1989). Inflow rate presumably influences pollutant retention by affecting the degree of bottom scouring and resuspension of settled solids, and therefore, the retention of solids and solids-associated pollutants. Wetland volume determines the fraction of a runoff event potentially captured, and hence made available for treatment, especially during quiescent periods between events (Woodward-Clyde, 1986). Thus, clear treatment goals and an evaluation of the occurrence and extent of putative removal processes are preliminary requirements for defining appropriate design and operation parameters for CWs (Imfeld et al., 2009).

Numerous workers have highlighted constraints, benefits, and design considerations for using wetlands to treat stormwater (Loucks, 1990; Rushton et al., 1997; Stockdale, 1991) and enhanced stormwater treatment wetland emerges where ecological and treatment objectives are simultaneously met (Otto et al., 2000). SFs (Greenway et al., 2006; Revitt et al., 2001) and horizontal SSFs (Shutes et al., 1999; Geary et al., 2006) have been used for the treatment of urban wet weather flows. Emerging research also focuses on the use of vertical flow CWs for the treatment of Combined Sewer Overflows (CSOs; Dittmer et al., 2005; Frechen et al., 2006; Welker, 2006). Several researchers have shown that wetlands are effective at reducing nutrient, sediment, organic carbon, and heavy metal loadings of urban stormwater runoff (Carr and Rushton, 1995; Reinelt and Horner, 1995; Rochfort et al., 1997; White and Myers, 1997). More recently, wetlands have been used for the retention and treatment of all kinds of urban wet weather flows, including runoff from airports, highways, parking lots, agricultural runoff, CSOs, and stormwater in general (Debo and Reese, 2003; Fink and Mitsch, 2005; Revitt et al., 2001).

While wetlands are effective at treating urban stormwater runoff, appropriate methods for determining stormwater wetland area requirements at a regional scale are also vital to enable feasibility studies and planning efforts. De Laney (1995) reviewed the approaches and suggested incorporating treatment wetlands into an agricultural landscape. While discussing how to select sites for wetlands created for intercepting agricultural runoff, van der Valk and Jolly (1992) suggested that placing small wetlands in the headwater subbasins or locating a large wetland in the downstream reach were practical alternatives. Likewise, Knight (1993) proposed that the location of treatment wetlands within the landscape should be driven by the goal chosen for the wetland; if water quality or attenuation of normal flooding is desired, then many small upstream sites are likely the best choice but if concern existed for controlling large episodic flow volumes or for creating wildlife habitat, then a large downstream wetland would prove as most useful.

As the intensity of land use activity increases within a watershed, so does the need for stormwater treatment wetlands. According to fitted estimates of the ratio of area of the neighborhood treatment wetlands to urban area, every 1% increase in urban area requires roughly 0.1% of the watershed area which should be used as wetland for treating stormwater runoff. The ratio of wetland treatment area to urban area for the basin scale should be in order of magnitude less (0.01% wetland area per 1% urban area), while at the subbasin scale the ratio of wetland treatment area to urban area lies in between the estimates for the neighborhood and basin treatment wetlands (0.05% per 1%). Perhaps, these ratios indicate general guidelines useful for planning stormwater management systems in urban development (Tilley and Brown, 1998).

However, it is necessary to realize that the need for stormwater treatment not only varies with the level of urbanization but that it also changes with the intensity of the urban use. If, for example, the classification of urban area is more heavily weighted toward industrial or commercial land, then the need for stormwater treatment would be higher than the land, more inclined toward residential uses. If subbasin treatment wetlands are sized to retain suspended sediments, then every one unit of land classified as commercial would require 0.12 units of stormwater wetland but each one unit of residential land would only require 0.045 units. Because each size class of wetland is designed for a particular purpose (e.g., nutrient sink, sediment trap, hydrologic pulse dampening), it is reasonable that all three size classes are necessary to achieve effective stormwater management. If the wetland areas for the three scales are simply summed, then for the most urbanized basins (urban area > 60% of total), the total wetland area needed for treatment is approximately 25% of the total basin, for basins of medium intensity (10% > urban area <60%) the wetland area is about 10% of basin area, while for the least urbanized (urban area <10%) the area required is less than 5% of the basin. However, if treatment wetlands are incorporated at each scale, then some synergism between the scales would likely emerge, leading to a smaller overall demand for land area. Therefore, the next step in evaluating the benefits of a network of stormwater wetlands, organized according to hierarchical principles, should be to investigate the cumulative or synergistic effects of including all three levels of the networks together (Tilley and Brown, 1998).

6. CONSTRUCTED WETLANDS FOR TREATMENT OF URBAN RUNOFF

Urban runoff is surface overflow of rainwater caused by urbanization due to construction of impervious surfaces such as roads, parking lots, and sidewalks. During rainstorms, these surfaces (built from materials such as asphalt, cement, and concrete), along with rooftops, carry polluted stormwater to storm drains, instead of allowing the water to percolate. Various studies regarding treatment of different SSPPs using CWs are arranged pollutant type wise as given in the SSPP list (Eriksson et al., 2007b), and discussed in subsections 6.1–6.5 (Table 3).

6.1 Basic Parameters

Basic parameters in the SSPP list include biochemical oxygen demand, chemical oxygen demand, suspended solids, phosphorus, nitrogen and pH. In a

TABLE 3. Examples of constructed wetlands used to treat various types of stormwater runoff containing SSPPs

Runoff waters	SSPPs	Location	Pollutant removal	Reference
(1) Basic parameters				
(i) Urban stormwater runoff	Phosphorus Suspended solids	Tampa, Florida, USA	TP 90% sediments 94%	Rushton et al. (1995)
(ii) Urban stormwater runoff	Phosphorus	Washington, USA	TP 82%	Reinelt and Horner (1995)
(iii) Urban stormwater runoff (residential)	Phosphorus, Nitrogen, COD, Suspended solids	Manassas, Virginia, USA	LTE 35.8% OP 48.1% TSP 45.9% TP 54.7% NH_3—N 19.8% SKN 25.5% TKN 39.4% OX-N 21.7% TN 57.9% TSS 21.9% COD	Carleton et al. (2000)
(iv) Nursery runoff	Phosphorus, Nitrogen	New South Wales, Australia	HRT 2–5 days TN removal > 84% TP removal > 65%	Headley et al. (2001)
(v) Urban stormwater runoff	Phosphorus	Henely Brook Perth, Western Australia	5% 1st year, 10% 2nd year	Lund et al. (2001)
(vii) Nursery runoff	Nitrogen	Nimes, France	nitrate reduction 70%	Merlin et al. (2002)
(viii) Urban stormwater runoff (residential)	Phosphorus, Nitrogen, Suspended solids,	Port Jackson, Sydney, Australia	MRE 12% TP, 16% TN, 9% TKN, 9–46% TSS	Birch et al. (2004)
(ix) Urban stormwater runoff	Nitrogen, Phosphorus	Crum Woods, Swarthmore College, Pennsylvania	nitrate-26% phosphate-59%	Bangs (2007)

(x) Urban stormwater runoff (gully pot) liquor)	Nitrogen, BOD	Experimental wetland, University of Edinburgh, Scotland	NH_3–N 78.7% 1st year 95.8%–2nd year NO_3–N 93.7% 1st year 95.5% 2nd year BOD 76.8% 1st year 96.5% 2nd year	Lee and Scholz (2007)
(xi) Combined stormwater Urban & agricultural runoff	Nitrogen, Phosphorus	Putrajaya wetlands, Malaysia	TN 82.1%, N-N 70.7% P 84.3%	Sim et al. (2008)
(xii) Highway runoff	COD, Suspended solids, Nitrogen, Phosphorus	Island of Crete, Greece	MRE-47% COD 89% TSS 49% TN 60% TP	Terzakis et al. (2008)
(xiii) Combined stormwater urban (14%) agricultural (40%), forested land (33%)	Nitrogen, Phosphorus	Seokmoon, Choongnam, Korea	MRE TN growing season 51.5% winter season 31.7% MRE TP growing season 50.6% winter season 53.0%	Ham et al. (2010)
(xiv) Combined wastewater runoff and urban stormwater runoff	BOD, Nitrogen Phosphorus	Danshui River Basin, Taipei	MRE-68% BOD 78% NH_4–N 49% TP	Ko et al. (2010)
		(2) Metals		
(i) Urban stormwater runoff (residential)	Cadmium Copper Lead Zinc	Manassas, Virginia	L.T.E. 29.2% TZn 35.5% SZn 30.8% TCd 28.0% SCd 65.5% TCu 47.7% SCu 74.7% TPb 33.2% SPb	Carleton et al. (2000)
(ii) Urban stormwater runoff	Zinc Lead Copper	Barker Inlet wetlands, Australia	Zn 57%, Pb 71% Cu 48%	Walker and Hurl (2002)

(Continued on next page)

TABLE 3. Examples of constructed wetlands used to treat various types of stormwater runoff containing SSPPs *(Continued)*

Runoff waters	SSPPs	Location	Pollutant removal	Reference
(iii) Urban stormwater runoff (residential)	Chromium Copper Lead Nickel Zinc	Port Jackson, Sydney, Australia	MRE 64% Cr 65% Cu 65% Pb 22% Ni 52% Zn	Birch et al. (2004)
(iv) Urban stormwater runoff (gully pot liquor)	Nickel Copper	Experimental wetland, University of Edinburgh, Scotland	Ni 81.5% 1st year 89.0% 2nd year Cu 97.3% 1st year 95.5% 2nd year	Lee and Scholz (2007)
(v) Highway runoff	Copper Nickel Lead Zinc	Island of Crete, Greece	MRE-23% Cu 61% Pb 59% Zn 33% Ni	Terzakis et al. (2008)
(3) PAH				
(i) Highway runoff	PAHs	Island of Crete, Greece	MRE 59% PAHs	Terzakis et al. (2008)
(4) Herbicides				
—				
(5) Miscellaneous				
—				
(6) Bacterial indicators				
(i) Urban stormwater runoff (residential)	Fecal coliforms	Port Jackson, Sydney, Australia	MRE 76%	Birch et al. (2004)
(ii) Urban stormwater runoff (residential)	Fecal coliforms	Plumpton Estate, New South Wales, Australia	MRE 79% for TTC 85% for ENT 87% for heterotrophic bacteria	Davies and Bavor (2000)

Note. Long-term efficiency (LTE) is based on the difference between the estimated total input (inlet + overland + wet fall + dryfall) and estimated total output (storms + base flow) loads over all monitored storms. TP = total phosphorus; P = phosphate; DRP = dissolved reactive phosphorus; TN = total nitrogen; N-N = nitrate-nitrogen; TKN = total Kjeldahl nitrogen; BOD = biochemical oxygen demand; COD = chemical oxygen demand; TSS = total suspended solids; PAHs = polycyclic hydrocarbons; HRT = hydraulic retention time; FRP = reactive phosphorus; OP = orthophosphate; SKN = soluble Kjeldahl nitrogen; TSP = total soluble phosphorus; OX-N = oxidized nitrogen; NH_3–N = ammonia nitrogen; NO_3–N = nitrate nitrogen; NH_4–N = ammonical nitrogen; TM = total metal; SM = soluble metal; TTC = thermotolerant coliforms; ENT = enterococci; MRE = mean removal efficiency; — = data not available.

study conducted by Tilley and Brown (1998), at each spatial scale in urbanized watersheds south of Miami, Florida, the wetland area needed to treat N, P, TSS, BOD, and the water quantity was calculated. Results indicated that at the neighborhood scale, P runoff generated by a five-year 24-hr design storm, required the largest wetland treatment area (i.e., between 2.3 and 10.8% of total basin area). Likewise, at the subbasin scale, the loading of TSS, derived from land use specific criteria, required the largest treatment area, ranging from 0.2 to 4.5% of basin area. The basin scale treatment, based on retaining drainage canal discharge for at least 72 hr, required between 0.1 and 2.5% of basin area.

Kadlec and Hey (1994) reported that CWs receiving diverted water from a river that drained a watershed with 80% agricultural and 20% urban land use, reduced sediment loads by an average of approximately 90% and TP loads by about 80% over a two-year period. Rushton et al. (1995) tested the effect of hydraulic residence time on the removal rate of various pollutants in a CW receiving urban runoff in Tampa, Florida. They found that the sediment load was reduced by 94% and TP by 90% for the wetland with the 14-day residence time but sediment was only reduced by 67% and TP by 57% when residence time was five days. Likewise, Reinelt and Horner (1995) observed that the percent removal of the TP load to a wetland was 8% when the residence time was only 3.3 hr. On the other hand, wetland with a 20-hr residence time removed 82%.

In a study conducted by Nairn et al. (2000), water quality changes and biogeochemical development were evaluated for over two years in two newly created freshwater riparian wetland ponds (1 ha each) in an agricultural and urban watershed. One wetland was planted with 13 species of vegetation while the other received no planted vegetation. Both wetlands significantly decreased turbidity (from 62 to 27 NTU) and increased DO (9–11 mg L^{-1}). Inflow dissolved reactive phosphorus (DRP) and TP concentrations (1793 and 169911 mg P L^{-1}, respectively) were significantly higher than outflow concentrations (DRP: 591 and 691 mg P L^{-1}; TP: 6998 and 7499 mg P L^{-1}) for planted and unplanted wetlands, respectively. The higher phosphorus removal was related to decrease in turbidity and the high level of biological activity. Extensive and highly productive algal growth in wetlands and the subsequent deposition and decomposition of the algal mat influenced P retention through biological uptake as well as by chemical sorption and coprecipitation. In another experiment, nitrate removal rates were studied by using three free-surface marsh vegetation treatments (bulrush or *Scirpus* sp., cattail or *Typha* sp., and a mixed stand of macrophytes and grasses) in replicated macrocosms (Bachand and Horne, 2000). Average nitrate removal rates differed significantly between vegetation treatments with the mixed treatment removing over three times more nitrate than the bulrush treatment. Mass balance computations demonstrated that bacterial denitrification rather than plant uptake was the main mechanism for nitrate removal. Both water temperature and organic carbon availability affected denitrification rates whereas DO and nitrogen concentrations showed no impact on the denitrification rates.

In another study, a CW design, consisting of 16 repeating cells was proposed for Henley Brook (Perth, Western Australia) to optimize the removal of filterable reactive phosphorus (FRP) from urban stormwater (Lund et al., 2001). Three replicate experimental ponds (15 × 5 m) were constructed and three 5-m zones of each pond were sampled such as shallow (0.3 m) inflow and outflow zones vegetated with *Schoenoplectus validus* and a deeper (1 m), V-shaped central zone. A removal efficiency of 5% (first year) and 10% (second year) was obtained for FRP and initial uptake was reported to be mainly in plant biomass, although the sediment became an increasingly important sink. Occurrence of very low biofilm biomass (<1 g m^{-2}) during the treatment duration was attributed to the dark color of urban stormwater. Further, benthic flux experiments showed that anoxic conditions did not cause release of P from sediments, indicating that most of the P was bound as apatite rather than associated with Fe or Mn.

Sim et al. (2008) evaluated the performance of Putrajaya Wetlands in Malaysia, a 200-ha CW system consisting of 24 cells and created in 1997–1998 to treat surface runoff caused by development and agricultural activities from an upstream catchment before entering Putrajaya Lake (400 ha). The nutrient removal performance was found to be 82.11% for total nitrogen, 70.73% for nitrate–nitrogen, and 84.32% for phosphate along six wetland cells. Additionally, nutrient removal studies in pilot-scale tank systems (Figure 1), simulating a CW and planted with the common reed (*Phragmites karka*) and tube sedge (*Lepironia articulate*) revealed 42.1% TKN; 28.9% P and 17.4% TKN; and 26.1% P uptake by the common reed and tube sedge, respectively.

In a different study, Ko et al. (2010) assessed loss in the pollutant removal efficiencies of FWS CW located in the Danshui River Basin of metropolitan Taipei, having total open surface water area of 3.29 ha and treating 4000 m^3 day^{-1} of combined domestic wastewater and rainfall runoff from the urban drainage watershed, with a hydraulic retention time of seven days. They reported that the mean influent BOD values of the CW before and after the typhoon Krosa that hit in 2007 were 28 and 26 mg L^{-1}, respectively, while the effluent BOD values were 8 and 13 mg L^{-1}, respectively. Likewise, the mean influent NH_4–N values before and after the typhoon were 16.5 and 12.3 mg L^{-1}, respectively, and the effluent NH_4–N values before and after the typhoon were 1.7 and 6.9 mg L^{-1}, with removal efficiencies of 78% and 46%, respectively. This 33% reduction in the removal rate of NH_4–N after the typhoon was attributed to increased hydraulic loading, damage to the wetland plantation, and a change in the bacterial population.

Han et al. (2010) monitored the performance of experimental field-scale wetland systems (four sets, 0.88 ha each) of Seokmoon watershed on the west coast of the Korean peninsula, where water depth was maintained

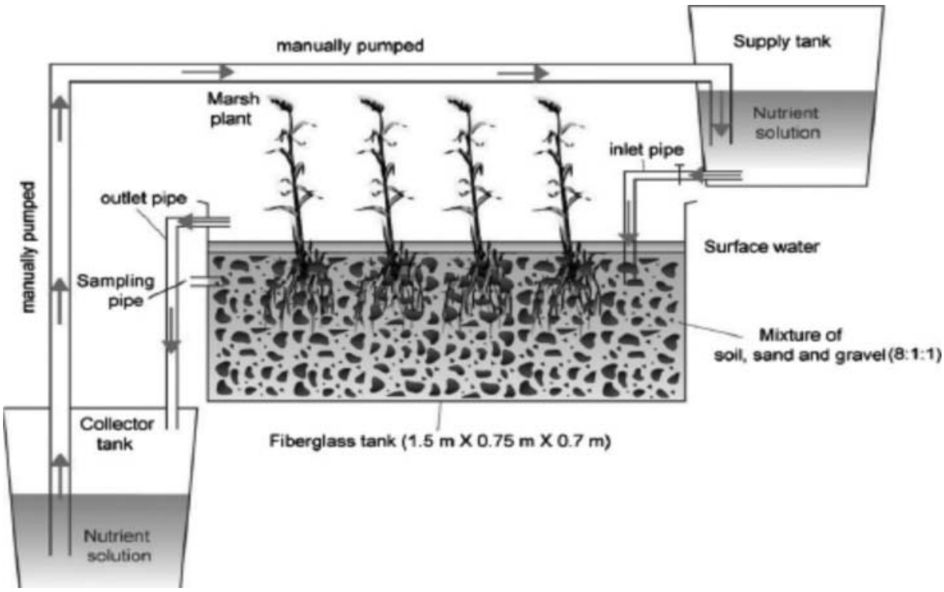

FIGURE 1. Schematic diagram of a surface-flow wetland mesocosm at University Putra Malaysia (Sim et al., 2008).

at 30–50 cm and hydraulic loading rate was at 6.3–18.8 cm day^{-1} and reported that average removal rate for TN in winter (about 30%; *M* effluent TN = 3.7 mg L^{-1}) was lower than that in the growing season (about 50%; *M* effluent TN = 1.5 mg L^{-1}). This was attributed to temperature dependence of various processes responsible for TN reduction in wetlands such as ammonification, nitrification, and denitrification (Werker et al., 2002). However, no difference was observed in the removal rate of TP between the growing season and winter with average effluent TP concentration of 0.14 mg L^{-1} in both seasons. This was explained as phosphorus removal, being largely a physical (sedimentation) and chemical (adsorption) process, was less likely sensitive to temperature (Wittgren and Mæhlum, 1997). The calibrated integrated modeling system estimated that constructing wetlands on 0.5% (about 114 ha) of the watershed area at the mouth of reservoir could reduce 11.61% and 13.49% of total external nitrogen and phosphorus loads, respectively.

In another study, a retention pond receiving stormwater runoff from a portion of a 273-ha Midwestern rail yard including fuel storage tanks was redesigned to have a 6.25 million L storage capacity and configured into a CW to control a 50-year storm event and increase its ability to treat stormwater runoff (Schaad et al., 2008). A network of riparian plants (5,700) was placed within the stormwater wetland to treat runoff prior to discharge off site. Significant improvements in the retention and treatment ability were observed during postconstruction flow analysis with 45% reduction in mean TSS and oil and grease concentrations.

The nursery industry is one of the fastest growing segments and it uses insecticides, fungicides, growth regulators, and fertilizers to aid production. In addition, high-intensity irrigation practices may create large volumes of wastewater. Traditional overhead, hand, or sprinkler irrigation may generate 18,000–90,000 L of wastewater per hectare daily (Aldrich and Bartok, 1994). Runoff from nurseries and greenhouses typically contains high concentrations of nitrogen (mostly as nitrate) and very low concentrations of organics. Prystay and Lo (1996) tested the potential use of an HF CW with a surface area of 254 m^2 for the treatment of low organic carbon, high-nutrient wastewaters (TOC: 21 mg L^{-1}, TP: 126 mg L^{-1}, NH_4–N: 38 mg L^{-1}, NOx–N: 240 mg L^{-1}) generated during the greenhouse operations in Canada. They suggested that the treatment efficiency appeared to be related to the organic carbon concentration in the system, which indicated that increased treatment efficiencies can be achieved as the wetland matures and larger litter layer accumulates.

Merlin et al. (2002) tested in Nimes, France, HF constructed experimental units to treat tomato greenhouse drainage solutions with the mean nitrate–N concentration of 329 mg L^{-1}. Up to 70% of nitrate was reduced in *Phragmites*-planted units. Likewise, Headley et al. (2001) noted that in New South Wales, Australia, the introduction of legislation to charge for the water used in agricultural production and to control runoff, has encouraged commercial plant nurseries to collect and recycle their irrigation drainage. They tested HF pilot-scale units filled with 10 mm basaltic gravel and planted with *Phragmites australis*. TN and TP load removal was found to be >84% and >65%, respectively, at HRTs between 2 and 5 days.

To assess the role of the macrophyte *Phragmites australis* (Cav.) Trin. ex Steud. in treatment of urban stormwater, Lee and Scholz (2007) used different experimental temporarily flooded vertical-flow wetland filters (four planted and four unplanted filters). Out of the four typical reed bed filters containing gravel and *P. australis*, two of them contained Filtralite (3% calcium oxide having diameters between 1.5 and 2.5 mm) as well as Frogmat (a natural product based on raw barley straw) as adsorption media. The results revealed that planted filters had a negative impact on the five-day N-allylthiourea BOD removal processes, although they provided good filtration conditions by preventing the filters from clogging. The possible reason for the relatively low effluent BOD concentration in the unplanted filters was assigned to the lack of vegetation cover, which resulted in extra aeration and oxidation of the organic load (Thomas et al., 1995). Furthermore, plant debris decayed within the wetland filters, and, hence, subsequently increased the organic loading in planted filters during the winter season. In contrast, the ammonia-nitrogen removal performance of planted filters was more efficient and stable throughout the year (particularly after the filters have matured) as compared with the unplanted filters. This was explained as the mature root system enhances the capacity of transporting oxygen to the substrate and provides a large surface area for microorganisms to grow on (Cooper et al., 1996; Scholz, 2006). Furthermore, a substantial amount of total nitrogen (80–140 mg per filter) was removed by harvesting *P. australis*. In contrast to this study, Vymazal (2002) reported that the removal of total nitrogen from wastewater was lower for vegetated beds compared with nonvegetated beds. However, he had only assessed horizontal-flow systems for wastewater treatment. With regard to pH change in the wetland filters, it was explained that the conversion of ammonium to nitrite results in the formation of H^+ ions (Eq. 1) and, subsequently, the H^+ are neutralized by $HCO_3{}^-$ ions during the nitrification process (Eq. 2). A decrease in bicarbonate alkalinity and an increase in the CO_2 lower the pH. Thus, a low pH significantly reduces the rate of nitrification, particularly at a pH of 7.2, when the rate falls rapidly, approaching zero at a pH of 6 (Rich, 2005; Tchobanoglous et al., 2003).

$$NH_4^+ + 1.5O_2 \rightarrow 2H^+ + H_2O + NO_2^- \tag{1}$$

$$H^+ + HCO_3^- \rightarrow CO_2 + H_2O^- \tag{2}$$

A lower pH level in a planted filter as compared with an unplanted filter was most likely attributed to the more active nitrification process. In addition, plants utilized nitrogen and thus contributed to the lowering of pH through respiration and litter decomposition processes (Collins et al., 2004). A similar relationship between pH change and nitrogen removal was observed by Sun et al. (2003) during treatment of agricultural wastewater.

6.2 Metals

Metals in the SSPP list comprise chromium as chromate, cadmium, platinum, copper, nickel, lead, and zinc. To study Cr and As removal by CWs, five sediment trap stations were set up in a long and narrow stormwater wetland (Barker Inlet wetlands, Australia) and it was found that the concentration of Cr remained relatively constant and that of As actually increased by 150% (Walker and Hurl, 2002). The performance of the different metals could be explained in terms of their chemical behavior and the role of organic matter in the wetland. In another study, the treatment efficiencies of vertical flow CWs containing *P. australis* were assessed for two years by Scholz (2004). In his trial, hydrated nickel and copper nitrate were added to sieved gully pot liquor to simulate contaminated primary treated stormwater runoff. For the CW filters receiving heavy metals, an obvious breakthrough of dissolved Ni was recorded after road salting during the first winter. However, after the first year of operation, breakthrough of Ni was not observed because the inflow pH was raised to 8 as high pH facilitated the formation of particulate metal compounds such as nickel hydroxide. During the second year, reduction efficiencies for heavy metals, BOD and SS improved considerably.

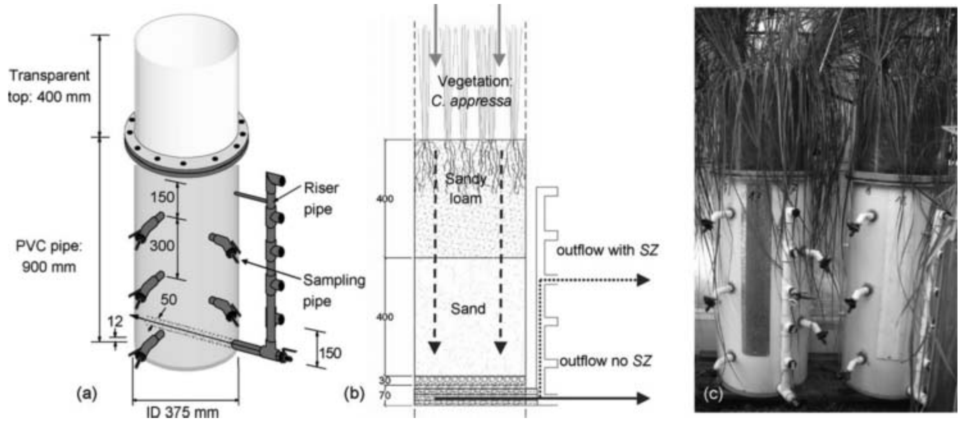

FIGURE 2. Biofilter column: (a) configuration, (b) filter media set-up and vegetation, (c) in greenhouse (Blecken et al., 2009).

Blecken et al. (2009) reported that the presence of a submerged zone (SZ) with added cellulose carbon (C) did significantly affect heavy metal removal, generally enhancing its uptake by the biofilter. A combination of SZ and C helps to meet target concentrations (ANZECC, 2000; Swedish EPA, 2000) which are usually not met without their incorporation. This enhancement was explained by the introduction of C that supports removal of Cu by forming Cu–OM complexes. The most efficient treatment was achieved for a SZ of 450 mm. Thus, in stormwater biofilters of 900 mm height, introduction of a 450 mm submerged zone (created by lining the whole system and raising the outlet to create a permanent pool at the base of the system) and a cellulose based carbon source was recommended (Figure 2). The authors further suggested verifying the observations in a field study of stormwater biofiltration system under real operating conditions.

In a different study, moderate to high but highly variable removal of trace metals from urban stormwater (draining a residential urban catchment of about 480,000 m^2) by a wetland constructed in the Sydney catchment was reported (Birch et al., 2004). In this study, 67–84% removal of Cr was reported during four of the six high-flow events while removal efficiency of Cu was reported to be 56–86% for all events except during the largest event (21%). Pb and Zn also displayed substantial removal by the wetland during five of the six events monitored (Pb 44–89%; Zn 33–87%), but efficiencies were again markedly reduced during the highest flow event (Pb 27% and Zn –5%). Further, the removal of Ni was highly variable and ranged from −76% to 72%. However, the concentrations of Ni in stormwater were generally below 0.005 mg L^{-1} and do not represent a threat to aquatic biota. A surprising outcome of the present work was the identification of the wetland as a source of Fe and Mn with the mean removal efficiencies of −84% for

Fe and −294% for Mn. Further, weighted average concentrations (WACs) of Mn were up to fivefold greater than influent concentrations, indicating that the wetland contributed rather than removing this element from stormwater. This increase in concentrations of Mn, and to a lesser extent Fe, was explained by the occurrence of medium to coarse-grained Fe- and Mn-oxide coated grains. They explained that these grains settle and accumulate in the wetland during low-flow conditions and are removed during periods of high flow when the resuspension threshold velocity for coarse grains is exceeded.

During the highest rainfall event, the removal efficiency of trace metals and TSS was substantially lowered than other events due to the intensity of rainfall (6.50 mm during the first hour of the event and peak intensity of 4.00 mm during a 10-min period). The peak rainfall intensity was greater during this event than during other events monitored, and a maximum flow of 140 L s^{-1} was observed at the inflow point to the wetland. The rainfall intensity and the resulting large flow volumes through the wetland during this high-flow event had contributed to the resuspension of fine-grained sediment and the transport of suspended particulates out of the wetland. The authors suggested maintenance dredging to minimize the release of acid-volatile, sulfide-bound trace metals from the wetland during the periods of high flow and sediment resuspension (Birch et al., 2004).

While assessing the role of the macrophyte *P. australis* in experimental temporarily flooded VF wetland filters for treatment of urban stormwater, reduction rates of Cu were found to be high (>95%) regardless of the planting regime during the two years of operation while the removal of Ni by the planted filter (0.057 g m^{-2} day^{-1}) was found to be lower than that by the unplanted filter (0.062 g m^{-2} day^{-1}; Lee and Scholz, 2007). The higher concentration of Ni in the effluent of planted filters was attributed to their lower pH values (6.72–6.93), compared with those of unplanted filters (7.19–7.31) as explained in section 6.1. These findings indicated that metal retention in the wetland filters is more susceptible to the change of environmental conditions such as pH and redox potential in the long term. Furthermore, metal loads removed by harvesting (0.3% and 0.1% of Ni and Cu loads) were negligible compared with those retained in the filters. Therefore, it was concluded that most of the heavy metal loads were accumulated in the sediment rather than taken up by *P. australis* in the filters. This confirms earlier findings of Scholz and Xu (2002) and Du Laing et al. (2006).

6.3 PAHs

In the SSPP list, PAHs comprise benzo[a]pyrene, naphthalene, and pyrene. In a study conducted by Terzakis et al. (2008), two FWS and two SSF pilot-size CWs treating highway drainage or runoff waters (HRO) were monitored over a period of two years. One FWS and one SSF were designed with a HRT of 12 hr, with each one capable of treating a maximum HRO of 12.6 m^3

day^{-1}. The other pair was designed with a HRT of 24 hr, with each receiving a maximum HRO of 6.3 m^3 day^{-1}. The influent flowed from a highway section with a total surface 2752 m^2 on the island of Crete, Greece, in the heart of the South-Central Mediterranean region. The outcome of the study showed that the performance of the SSF wetlands for PAHs removal was better than the two FWS wetlands, which may be attributed to the nature of the pollutants (attached to the surfaces of solids or low organic matter concentration) and the way these pollutants could be removed (mostly physically through filtration and sedimentation with little biological action).

In another study, Weiss et al. (2006) studied uptake of contaminants from the nutrient solution simulating HRO by three wetland species, soft-stem bulrush (*Scirpus validus*), prairie cordgrass (*Spartina pectinata*), and reed manna grass (*Glyceria grandis*). Consistent with other studies (Qian et al., 1999; Zhu et al., 1999), all three species accumulated a higher concentration of the target contaminants in their belowground tissues than the aboveground tissues. The data demonstrated that plants grown hydroponically had a higher concentration of contaminants than those grown in sand and also the plants grown in flow reactors removed a greater mass of contaminants than those grown in nonflow reactors. In another study, the hydrology and water quality of an urban wetland receiving stormwater runoff from a municipal maintenance garage were measured to evaluate the wetland's water quality enhancement function. Hydrological and analytical data together suggested that sedimentation was the primary mechanism for actively reducing concentrations of lead and petroleum hydrocarbons from the water column, which were introduced to the wetland via stormwater runoff (Thurston et al., 1999).

6.4 Miscellaneous

Miscellaneous category in the SSPP list of pollutants consists nonylphenol ethoxylates and degradation products, 2,4,4′-trichlorobiphenyl (PCB-28), DEHP, MTBE, and PCP. Airport runoff contains deicing and anti-icing compounds applied to the aircraft, runways, and taxiways. The principal materials involved are ethylene, di-ethylene, and propylene glycols (Worrall et al., 2002). Probably the first full-scale HF CW for airport runoff was a 5500-m^2 system, built in 1994 to treat deicing runoff water at Zürich-Kloten Airport, Switzerland (Röthlisberger, 1996). After the reed bed experiment in 1994 (Revitt et al., 2001), a full-scale system at London Heathrow International Airport was completed in 2002 with the primary aim to treat deicing compounds contaminated runoff from an extensive catchment of some 600 ha of runways, taxiways, cargo areas, and terminal buildings. The system comprised of a series of aerated balancing ponds combined with 2.08 ha of gravel-based HF CWs together with a kilometer of rafted reed beds (Richter et al., 2004). Karrh et al. (2002) reported on the use of an HF CW for the treatment of anti- and deicing runoff built at Westover Air Reserve base in western Massachusetts, USA. Deicing runoff was also treated in HF CWs at Edmonton International Airport, Canada (Higgins and Dechaine, 2006), and Berlin-Schönefeld, Germany (Vymazal, 2009). The catchment area at Edmonton is very large, and this, coupled with the airport's tight clay soil, results in very large amount of stormwater runoff. The HF CW used by Higgins and Dechaine (2006) consisted of 12 square gravel-filled cells with sides of 47.5 m, each arranged in six trains of two cells each. Wetland surface area was 2.7 ha and design conditions for the wetland were for the treatment of stormwater runoff contaminated with up to 1350 mg L^{-1} of ethylene glycol at flows up to 1500 m^3 day^{-1}.

6.5 Bacterial Indicators

Bacterial indicators in stormwater include fecal coliforms (*Escherichia coli*, *Enterococci* sp.) and pathogens (*Pseudomonas aeruginosa*, *Staphylococcus aureus*, *Clostridium perfringens*). In a study conducted by Davies and Bavor (2000) on stormwater from residential catchment of Plumpton Estate, New South Wales, Australia, mean removal efficiencies for the CW over a six-month period were reported to be 79%, 85%, and 87% for thermotolerant coliforms (TTC), enterococci (ENT), and heterotrophic bacteria, respectively. The concentrations of bacteria in sediments were reported to be higher than the water column concentrations, often by several orders of magnitude. This difference was most pronounced for *Clostridium perfringens* spores, the concentrations of which ranged from <1 to 40 spores per 100 ml in the water column and from 104 to 107 spores per 100 g d.w. of the sediment. Additionally, the bacterial concentrations in the top 10 cm remained relatively constant with time. This suggests that the bacteria were almost exclusively associated with the smaller particles (<2 mm) that remained suspended throughout the duration of the settling experiment, and not attached to the larger particles that settled out within the experimental duration. However, it has been shown that the process of bacterial adsorption to particles increases bacterial persistence in aquatic environments by protecting them from environmental stresses that may otherwise be responsible for their mortality (e.g., solar radiation, starvation and attack by bacteriophages (Gerba and McLeod, 1976; Roper and Marshall, 1974). Further, the greater effect of predation on TTC compared with ENT concentrations may be related to the hydrophobic properties of streptococci, which enable them to bind more efficiently than coliforms to clay particles (Huysman and Verstraete, 1993). Consequently, ENT may be protected from predators to a greater degree in CWs. Additionally, it is reported that the protozoa preferentially prey on coliform bacteria as compared with ENT (Gonzalez et al., 1990).

In addition, several workers have found a significant relationship between bacterial mortality rates and sediment particle size. TTC mortality rates were shown to be significantly lower in sediment with predominantly clay-sized particles than in coarser sediments (Howell et al., 1996). Burton et al. (1987) found that particle size was the only sediment characteristic that was related to the survival of *Escherichia coli* and *Salmonella newport*, both of which survived significantly longer in sediments containing at least 25% clay. The persistence of microorganisms in wetland sediments suggest that the sediments may act as reservoirs of viable bacteria. It has been shown that sediment-bound bacteria may be resuspended back into the water column by storm activity, thereby resulting in deterioration in the quality of the overlying water (Crabill et al., 1999). The TTC removal efficiencies for the wetland in the study of Davies and Bavor (2000) were somewhat lower than values previously reported which usually exceeded 90%. However, most of the previous microbiological studies have focused on the assessment of wetlands for the treatment of municipal and industrial wastewater rather than for the treatment of stormwater (Kadlec and Knight, 1996). In point of fact, poor performance of CWs for stormwater treatment was attributed to presence of higher proportions of fine particles (<2 mm) in comparison with municipal wastewater treatments.

Birch et al. (2004) in their study on treatment of urban stormwater runoff by CW reported that the number of fecal coliform colonies (FC) in stormwater was very high (1,10,000 cfu 100 ml^{-1}), representing a level that was 110 times above the recommended number for human health safety for secondary contact (e.g., boating). Although fecal coliform counts in effluent water during high-flow events were below 5500 cfu 100 ml^{-1} for two events, substantially greater FC contents of up to 2,20,000 cfu 100 ml^{-1} were recorded during the largest of the high-flow events monitored. This indicated a high removal efficiency of FC during moderately intense high-flow events ($\sim$1.0 mm of rain $hour^{-1}$), but efficiency was substantially reduced during periods of intense rainfall (>4.0 mm of rain $hour^{-1}$). The mean removal efficiency of fecal coliforms was reported to be 98%, 83%, and 99% during three high flow events but decreased to 26% during the largest high-flow event sampled. During the latter, settling and removal of suspended particulates was substantially reduced and resuspended particulates likely contributed to the elevated TSS and fecal coliforms contents in outflowing stormwater during this event.

7. STORMWATER TREATMENT WETLANDS: DESIGN CRITERIA AND ECONOMIC PERSPECTIVE

Stormwater treatment wetlands are CW systems designed to maximize the removal of pollutants from stormwater runoff via several mechanisms such as microbial breakdown of pollutants, plant uptake, retention, settling, and adsorption. They also promote the growth of microbial populations that can utilize soluble carbon and nutrients and potentially reduce BOD and fecal coliform levels concentrations. Stormwater wetlands temporarily store runoff in shallow pools that support conditions suitable for the growth of wetland plants (Schueler, 1992). In addition, organic soils found in mature wetland act like a sponge to retain water and allow infiltration of surface water into the groundwater. This decreases not only runoff volume, but also peak discharges, which may otherwise cause flooding or erosion downstream. As channelized flow enters a wetland, the velocity is reduced as the water spreads out over the wetland. Velocity is further reduced by the frictional resistance of aquatic vegetation. It is this reduction in velocity that is most responsible for sediment and nutrient retention in CWs (Jones, 1996).

The Metropolitan Council of Governments has developed guidelines for constructing wetland stormwater basins (Schueler, 1992). These guidelines recommend a wetland surface area of 1–2% of the watershed area, depending on the nature of the watershed and the design of the facility. Effective wetland design displays complex microtopography. In other words, wetlands should have both very shallow (<6 inches) and moderately shallow (<18 inches) zones created by underwater earth shelves (California Stormwater Quality Association, 2003). However, the use of stormwater wetlands is limited by a number of site constraints, including soil types, depth to groundwater, contributing drainage area, and available land area. A pondscaping plan should be developed for each stormwater wetland. This plan should include hydrological calculations (or water budget), a wetland design and configuration, elevations and grades, a site/soil analysis, and estimated depth zones. This plan should also contain the location, quantity, and propagation methods for the stormwater wetland plants. A site appropriate for a stormwater wetland must have adequate water flow and appropriate underlying soils. Base flows from the drainage area or groundwater must be sufficient to maintain a shallow pool in the wetland and support the vegetation, including species susceptible to damage during dry periods.

Underlying soils that are Natural Resources Conservation Service (NRCS) types B (sandy loam), C (clay loams), or D (heavy plastic clays) will have only small infiltration losses. Sites with type A (sandy) soils have high infiltration rates and may require a geotextile liner or a 15-cm (6-inch) layer of clay. Medium-fine texture soils (e.g., loams and silt loams) are best to establish vegetation, retain surface water, permit groundwater discharge, and capture pollutants. At sites where infiltration is too rapid to sustain permanent soil saturation, an impermeable liner may be required. Even, where the potential for groundwater contamination is high, such as runoff from sites with a high potential pollutant load, the use of liners should be required. After excavation and grading of a basin, at least 10 cm of soil should be applied to the site. This material, which may be the previously excavated soil or other suitable material, is needed to provide a substrate in which vegetation can become established (Schueler, 1992).

For planning of stormwater wetlands, sites must be carefully evaluated. Soils, depth to bedrock, and depth to water table must be investigated before designing and siting stormwater wetlands. Site preparation requirements and a maintenance schedule are also necessary components of the plan. The water budget should demonstrate that there will be a continuous supply of water to sustain the stormwater wetland. The water budget should be developed during site selection and checked after preliminary site design. Drying periods of longer than two months have been shown to adversely affect plant community richness, so the water balance should confirm that drying will not exceed two months. After excavation and grading, the wetland should be kept flooded until planting. Six to nine months after being flooded and two weeks before planting, the wetland is typically drained and surveyed to ensure that depth zones are appropriate for plant growth. Revision may be necessary to account for any changes in depth (Schueler, 1992).

The wet basin should be configured as a two-stage facility with a sediment forebay and a main pool. Sediment forebays are recommended to decrease the velocity and sediment loading to the wetland. The forebays provide the additional benefits of creating sheet flow, extending the flow path, and preventing short circuiting. They should contain at least 10% of the wetland's treatment volume and should be 4–6-ft deep to prevent vegetation from encroaching on the pond open water surface. The forebay is typically separated from the wetland by gravel or by an earthen shelf (Schueler, 1992). The bottom of the forebay may be hardened (concrete) to make sediment removal easier. A fixed vertical sediment depth marker should be installed in the forebay to measure sediment accumulation (California Stormwater Quality Association, 2003). The wetland design should include a buffer to separate the wetland from surrounding land. Buffers may alleviate some potential wetland nuisances, such as accumulated floatables, odors, and or geese. A buffer of 25 ft is recommended, plus an additional 25 ft when wildlife habitat is of concern. Leaving trees undisturbed in the buffer zone will minimize the disruption to wildlife and reduce the chance for invasion of nuisance such as cattails and primrose willow (Schueler, 1992).

Flow from the wetland should be conveyed through an outlet structure that is located within the deeper areas of the wetland. Before the outlet, a 4–6-ft deep micropool (having a capacity of at least 10% of the total treatment volume), should be included in the design to prevent the outlet from clogging. The outlet from the micropool should be located at least one foot below the normal pool surface. Because of the abundance of vegetation in the wetland, a trash guard should be used to protect the outlet. A trash guard large enough, so that velocities through it are less than 2 fps (feet per second) will reduce clogging problems (Schueler, 1992). High velocities can wash out rooted vegetation and clean deposited sediments. Ideally, flow velocities should be less than 0.6 fps. Water depths less than 40 inches result in greater resistance to flow and favor aquatic vegetation. The preferred depth ranges are 0–1 ft of water for emergent plants, 1–2 ft for rooted surface plants, and 1.5–6.5 ft for rooted submerged plants. Pools deeper than 40 inches should be included in the wetland design to maximize sediment deposition and provide winter fish habitat. The terrestrial-aquatic boundary should have a very gradual slope. This allows for the establishment of a continuum of emergent species and reduces the erosive effects of waves hitting a sharp shoreline boundary (Jones, 1996).

Wetland vegetation can be established by three methods: allowing volunteer vegetation to become established (not recommended), planting nursery vegetation and seeding. A higher diversity wetland can be established when nursery plants are used. Vegetation from a nursery should be planted during the growing season (not during late summer or fall) to allow vegetation to store food reserves for their dormant period. Species adaptable to the broadest ranges of depth, frequency and duration of inundation (hydroperiod) should be selected (Schueler, 1992). Among different aquatic vegetation, persistent emergent vegetation have stems that persist even after the growing season. This provides year-round resistance to water flow. These plants include cattail (*Typha* sp.), iris (*Iris pseudacorus* or *I. versicolor*), rush (*Juncus* sp.), cordgrass (*Spartina* sp.), reedgrass (*Calamagrostis* sp.), sawgrass (*Cladium jamaicense*), and switchgrass (*Panicum virgatum*). Woody plants, such as alder (*Alnus* sp.), buttonbush (*Cephalanthus occidentalis*), and black willow (*Salix nigra*), are useful edge species with persistent stems. On the other hand, submerged vegetation removes nutrients seasonally, but does not offer significant frictional resistance to suspended sediments (Jones, 1996).

A generalized layout of urban stormwater treatment wetland (California Stormwater Quality Association, 2003) is given in Figure 3. Schueler (1992) has described four types of basic stormwater wetland designs that are explained as the following.

Design 1: Shallow Marsh System

Shallow marsh systems are configured with different low marsh and high marsh areas, which are referred to as cells. They also include a forebay for coarse particulate settlement before the wetland cell and a micropool at the outlet. Shallow marshes are designed with sinuous pathways to increase retention time and contact area. Most shallow marsh systems consist of pools ranging from 6 to 18 inches during normal conditions. Shallow marshes may require larger contributing drainage areas than other systems, as runoff volumes are stored primarily within the marshes, not in deeper pools where flow may be regulated and controlled over longer periods of time.

Design 2: Pond/Wetland Systems

Multiple cell systems, such as pond/wetland systems, utilize at least one pond component in conjunction with a shallow marsh component. The first cell is typically the wetpond that provides for particulate pollutant removal. The wetpond is also used to reduce the velocity of the runoff entering the system. The shallow marsh provides additional treatment of

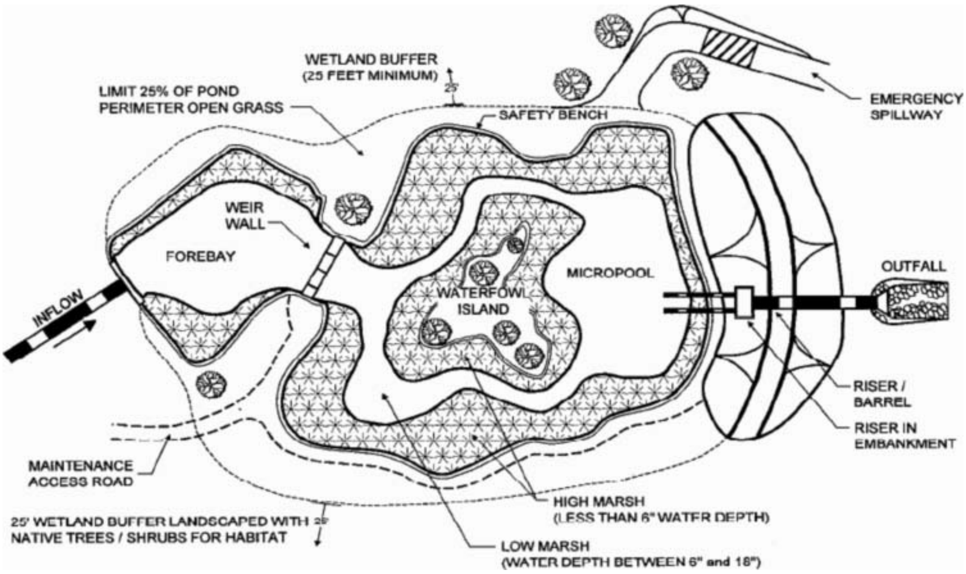

FIGURE 3. A generalized layout of urban stormwater treatment wetland (California Stormwater Quality Association, 2003).

the runoff, particularly for soluble pollutants. These systems require less space than the shallow marsh systems and generally achieve a higher pollutant removal rate than other stormwater wetland systems.

Design 3: Extended Detention Wetlands

Extended detention wetlands provide a greater degree of downstream channel protection. These systems require less space than the shallow marsh systems since temporary vertical storage is substituted for shallow marsh storage. The additional vertical storage area also provides extra runoff detention above the normal elevations. Water levels in the extended detention wetlands may increase by as much as three feet after a storm event and return gradually to normal within 24 hr of the rain event. The vegetated area in extended detention wetlands expands from the normal pool elevation to the maximum surface water elevation. Wetland plants that tolerate intermittent flooding and dry periods should be selected for the extended detention area.

Design 4: Pocket Wetlands

These systems may be utilized for smaller sites of 1–10 acres. To maintain adequate water levels, pocket wetlands are generally excavated down to the groundwater table. Pocket wetlands that are supported exclusively by stormwater runoff generally have difficulty in maintaining marsh vegetation due to extended periods of drought.

Stormwater wetlands require routine maintenance, for instance, the small forebay should be dredged every year to protect the wetland from excessive sediment buildup. In the first three years after construction, twice-yearly inspections are needed during both the growing and nongrowing season. Regulating the sediment input to the wetland is the priority maintenance activity. The majority of sediments should be trapped and removed before they reach the wetlands either in the forebay or in a pond component. Gradual sediment accumulation in the wetland results in reduced water depths and changes in the growing conditions for the emergent plants and its removal within the wetland can destroy the wetland plant community. Thus, shallow marsh and extended detention wetland designs include forebays to trap sediment before reaching the wetland. These forebays should be cleaned out every year. Harvesting of wetland vegetation can also be considered to remove nutrients from the wetland system and to minimize nutrient release when vegetation dies in the autumn. This is generally not recommended, but in special cases it will remove the nutrients contained in the vegetation from the system (Schueler, 1992). CWs should be stocked regularly with mosquito fish (*Gambusia* sp.) to enhance natural mosquito and midge control. An annual vegetation harvest in summer appears to be optimum, because it is after the bird breeding season and mosquito fish can provide the needed control until vegetation reaches late summer density, and there is time for re-growth for runoff treatment purposes before the wet season (California Stormwater Quality Association, 2003).

Wetland based stormwater treatments are relatively inexpensive. Construction cost data for wetlands are rare, but one simplifying assumption is that they are about 25% more expensive than stormwater ponds of an equivalent volume. Using this assumption, Brown and Schueler (1997) developed following equation to estimate the cost of stormwater wetlands: $C = 30.6^{V0.705}$. Where C is construction, design, and permitting cost, and V is wetland volume needed to control the 10-year storm (ft^3).

Using this equation, the following typical construction costs are obtained: \$57,100 for a 1 acre-foot facility, \$289,000 for a 10 acre-foot facility, and \$1,470,000 for a 100 acre-foot facility.

Wetlands consume about 3–5% of the land that drains to them, which is relatively high compared with other stormwater management practices. In areas where land value is high, this may make wetlands an impracticable option. For ponds, the annual cost of routine maintenance has typically been estimated at about 3–5% of the construction cost; however, the published literature is almost totally devoid of actual maintenance costs. Because CWs are long-lived facilities (typically longer than 20 years), major maintenance activities are unlikely to occur during a relatively short study (California Stormwater Quality Association, 2003).

Highway runoff is responsible for serious environmental impacts, especially in the long term. Roads represent approximately 20% of urban catchment areas, but the HRO can contribute as much as 50% of the TSS, 16% of the total hydrocarbons, and between 35 and 75% of the total metal pollutant inputs to receiving watercourses (Shutes et al., 1997). Traffic characteristics (e.g., mean vehicle speed, traffic load), climate, long dry/wet periods, and rainfall event intensity and duration are regarded as important factors in generating pollutants in HRO (Crabtree et al., 2006). CWs have been employed successfully as a viable solution for the treatment of HRO. There are several studies regarding performance of wetlands in treating HRO (e.g., Bulc and Slak, 2003; Mitchell et al., 2002; Shutes et al., 1999; Shutes et al., 1997).

The variable quality and quantity of HRO requires a more complex design for a CW treatment system. Figure 4 shows the use of pretreatment

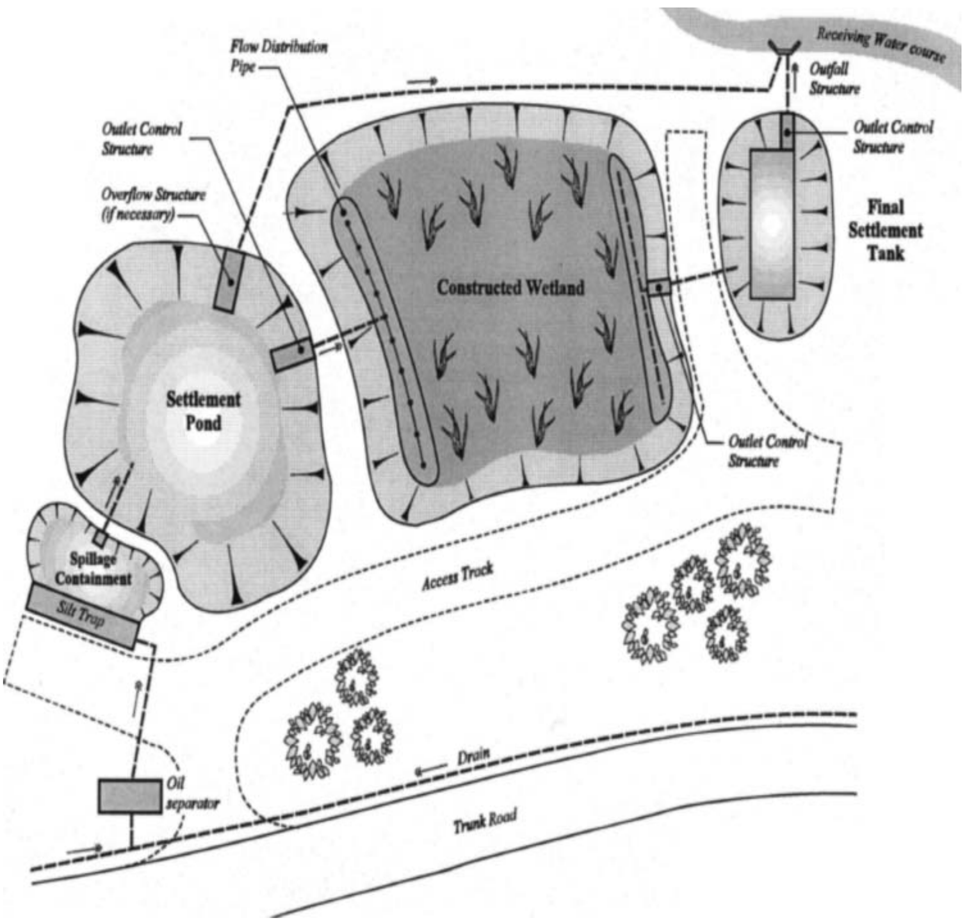

FIGURE 4. An idealized design of constructed wetland for HRO treatment (Shutes et al., 1999).

structures including an oil separator, silt trap, spillage containment basin, and settlement pond in an idealized HRO treatment system incorporating a horizontal subsurface flow CW. A final settlement tank provides post treatment followed by discharge to a receiving water body. An overflow structure from the settlement basin prevents excessive flows passing through the wetland and damaging the plants (Shutes, 2001). Bresciani et al. (2007) reported on the HRO treatment project for the highway connection Villesse-Gorizia in Italy. The project included a total of 60 CWs along 17 km of highway. Each system consisted of a first flush sedimentation tank, HF CW, wet pond, and a final vegetated retention area.

As CWs require 1–3 years to mature and become capable of efficient wastewater treatment, bypass oil separators, silt traps, and spillage containment facilities must be installed prior to the discharge of HRO into the CW owing to the phytotoxic nature of HRO (Shutes et al., 1999). If sufficient land is available, a final settlement tank with a minimum capacity of 50 m^3 extending across the width of the wetland can be installed. It will help to prevent fine sediment from the wetland being transferred into the receiving water body especially during the highway construction period. A SSF system provides very little flood storage and thus, where significant flood storage of runoff is required, it would be necessary to combine the treatment facility with separate storage (Shutes et al., 1999).

The CW should be large enough to retain the first flush of the longer storms to achieve partial treatment and delay transfers to the watercourse until normal flows have returned. They should ideally have a minimum retention time of 30 min. An ideal design should retain the average annual storm volume for a minimum of 3–5 hr and preferably for 10–15 hr to achieve good removal efficiency. For the CW design, the following criteria are recommended: a retention time of 24 hr, an aspect ratio (width:length) of 1:1 to 1:2, a slope of wetland bed of 1% maximum, a minimum substrate bed depth of 0.6 m, a substrate of 0.15 m of soil over 0.45 m pea gravel, a hydraulic conductivity of substrate of 10^{-3} m s^{-1} to 10^{-2} m s^{-1} (Shutes et al., 1999). During storm events, high rates of HRO may discharge over CWs, but optimal hydraulic loading should not exceed 1 m^3 m^{-2} day^{-1} in order to achieve a satisfactory treatment. Flow velocity should not exceed 0.3–0.5 m s^{-1} at the inlet zone if effective sedimentation has to be achieved. At velocities greater than 0.7 m s^{-1}, high flow may damage the plants physically and cause a decline in system efficiency (Shutes et al., 1999).

The inlet pipe should be constructed in such a way that influent flow is evenly distributed across the width of the bed. This may be achieved using slotted inlet pipes where the slots are sufficiently large to prevent clogging by algae. A stilling structure under the inlet, usually a 1-m-wide stone trench is necessary to either dissipate high water flows or to contain the inlet distributor pipe. The lowest level in the wetland should be 300 mm below the substrate surface and during dry periods an additional source of water may be needed to supply the reed beds. A grid of slotted plastic pipes with diameter of about 100 mm should be installed vertically in the substrate at 5-m intervals to aid aeration of the root zone. Nonmetallic items should be incorporated into the construction of the wetland so that metals in the wetland only arrive from road runoff. The location of CWs depends on low points in the road drainage system, location of the receiving waters, and local topography. It is not necessary to site CWs beside the road. However, the centers of roundabouts and areas between the arms of slip roads are spaces that may be utilized in wetland construction. Gravel provides the most suitable substrate for emergent plants growing in CWs, supporting adequate root growth and superior permeability (Shutes et al., 1999). Natural boulder or bentonite clay or geotexile liners may be used as reed bed bases, in instances where prevention of leakage to groundwater is imperative. An impermeable layer is also necessary to retain water in the wetland during dry periods.

Similar to general stormwater treatment wetlands, the operation and maintenance procedures connected with a CW treating HRO include removal of sediments, maintenance of the substrate and plants, harvesting, maintenance of water levels, maintenance of nutrient levels, general structure maintenance, and control of weed growth. Finally, different water quality parameters (pH, DO, TSS, BOD, COD, nitrates, phosphates, heavy metals, and hydrocarbons) should be regularly monitored to assess the performance of wetland (Shutes et al., 1999).

Scholes et al. (1998) compared the performance of two wetlands, one in Brentwood (north of London) including SSF (main system) and FWS wetlands (storm event system), and one FWS wetland in Dagenham (a large suburb in northeast London). Even though the authors compared the two systems, they indicated that there are limitations, as the physicochemical characteristics of the HRO vary considerably and it is very difficult to estimate the amount for HRO due to its direct relation to the amount of rainfall. Thus, wetlands operating in England have provision for balancing tank or pond for regulating the incoming HRO. In the event of a storm, this reduces the threat of flushing retained pollutants from the wetlands and provides some retention time for the wetland system to treat the influent (Pontier et al., 2004).

There is very little published information regarding economics of CWs treating HRO, with Weiss et al. (2007) providing the most interesting data. They conducted a comparative study between various HRO treatment options and concluded that wetlands are the most cost-effective systems in North America, as long as the land cost is not included in the relevant estimations. However, none of the publication had a cost breakup, which would have made the data more useful to potential end users, especially local authorities and highway operators. Only Soderqvist (2002), in a cost analysis for wetlands in Sweden, presented a cost breakup of the different tasks involved in the project. One of the main observations was that excavation activities form the major cost of such constructions, emphasizing the importance of detailed excavation planning for more accurate budget estimate predictions.

In the study conducted by Manios et al. (2009), a good understanding of construction and operation costs, as well as the resources required for the establishment and implementation of CWs systems, was presented. The construction of wetland in this study represented 25% of the total construction cost, while 5% was spent on the influent automated (and sun-powered) control and distribution system from the storage tank to the wetlands. The respective total cost allocated to the two SSF systems (€14,676) was approximately 10% higher than that of the FWS (€13,596), mainly due to the three different-sized gravel layers used in the SSF substrate compared with the topsoil used in the FWS, which tripled the cost and placement time. The total annual economic cost (TAEC) was €1,799 year^{-1} and €1,847 year^{-1} for the FWS and SSF pair, respectively.

They also suggested that the excavation cost for the construction of wetlands should be taken into consideration very carefully. It is a parameter often ignored by designers of wetland systems, as they do not know what lies 50 cm below the surface of the site. Excavation constitutes at least 20% of the construction cost. The construction of a storage tank was recommended as a control device, smoothing the flow into the wetlands. Its cost represents at least 25% of the construction. If all the tasks involving the use of reinforced concrete are added, this cost represents almost 35% of the budget allocated. Tasks directly related to the wetland, such as membrane liner installation, substrate addition, and planting, represent about 30% of the total cost. Together with excavation they make up 50% of the total cost, further supporting the idea that wetlands are a low-cost system. It should be noted that most competing systems (such as packed bioreactors) also require pretreatment, resulting in a substantially higher construction cost. SSF systems are generally far more difficult to construct, especially with the new design demand to layer the gravel in various sizes. Additionally, they also cost almost 10% more than FWS. The economical and construction superiority of the FWS is further supported by analyses of the TAEC, where the performance of the systems was taken into account. Financially and construction-wise, FWS systems are more suitable than SSF systems for stormwater such as highway runoff (Manios et al., 2009).

8. CONCLUSION

CWs are increasingly being installed to polish urban drainage and stormwater by reducing contaminants before disposal into receiving waters. Given proper considerations, they can greatly enhance water quality in a variety of urban settings such as municipal, industrial, and agricultural. The CW systems reduce or remove contaminants including organic matter, inorganic matter, trace organics and pathogens from the stormwater. As already discussed, reduction is said to be accomplished by diverse treatment mechanisms including sedimentation, filtration, chemical precipitation, adsorption, microbial interactions, and uptake by vegetation. However, these mechanisms are complex and not yet entirely understood.

In context of stormwater treatment, CWs have been looked at as a black-box, as only influent and effluent concentrations are measured without any further investigations. There is a strong need to enlighten the black-box to understand the basic elimination and transformation processes for specific contaminants and to design CWs for optimized treatment of stormwater priority pollutants. For most of the contaminants, data on removal rates are not available such as cadmium and platinum among heavy metals; benzo[a]pyrene, naphthalene, and pyrene among PAHs; phenmedipham, pendimethalin, terbutylazine, and glyphosate among herbicides; and nonylphenol, 2,4,4′-trichlorobiphenyl, DEHP, MTBE, and PCP among miscellaneous contaminants. Particular attention should also be given to the improvement of the effectiveness of microbial and molecular biology methods to measure microbial reaction rates (Truu et al., 2009). Therefore, there is a great need of research to determine process rates for proper designing of CWs.

As is evident, CWs have been commissioned as stormwater treatment facilities in many parts of the world (Vymazal, 2009), but to date, the technology has been largely ignored in developing countries where effective, low-cost wastewater treatment strategies are critically required. Even though, the potential for application of wetland technology in the developing world is enormous, as tropical and subtropical regions are known to sustain a rich diversity of biota that may be used in wetlands (Kivaisi, 2001). For the temperate and subtropical climates found in the Northern Hemisphere, basic principles related to design, application, operation, and maintenance of CWs have been established, and comprehensive guidelines and recommendations are available. However, these established road maps may not be directly transferable to tropical environments.

Additionally, developed world advisors may be entrenched in appropriate technologies for their countries and are unable to transfer their conceptual thinking to the realities and cultures of the third world. Thus, rather than assisting developing countries to develop their own CW technologies, the tendency has been to translocate northern designs to tropical environments (Kivaisi, 2001). The developing countries need to create in-house expertise through establishment of research units in the regions where CWs will be established, as they better understand the technology and economic scenario of their countries. Exchange of knowledge with external institutions should be encouraged. Well-qualified scientists, engineers, and managers are needed to formulate and implement strategies for overall environmental protection.

To determine whether CW treatment technology is cost-effective, environmentally sensitive, and technically reliable option for a given project location, its careful and exhaustive analysis must be conducted. It should be remembered that CWs might not always be the best alternative for low-cost and effective stormwater treatment. Different factors such as cost of development, suitable free-land availability, a relatively flat topography, nature of soils, and operating and maintenance costs including harvesting of vegetation and nuisance control must be cautiously considered in determining the suitability of CW technology as an appropriate option for stormwater treatment, especially in developing nations. This is because economic and technological constraints and lack of effective environmental pollution control laws or law enforcement in developing nations discourage them to adopt CWs and even other BMPs for stormwater runoff.

Although there are well-established design criteria of CWs for municipal wastewater treatment, criteria for urban and highway runoff treatment systems have yet to be fully agreed upon because urban storm runoff is much more difficult to control due to the random nature of rainfall and uncertainty of the pollution source. Particularly, the presence of heavy metals in urban runoff is of great concern, as they are most toxic due to enhanced bioavailability and nonbiodegradability in the environment. Further, the partitioning of heavy metals into different particle size classes in terms of their adsorption to particulates has major implications for urban water quality management (Herngren et al., 2005). Therefore, further approaches adopted to mitigate the impacts of urban runoff on receiving waters should include the use of structural or regulatory measures such as extended detention basins, gross pollutant traps, and restrictive zoning. However, for these measures to be effective, an in-depth understanding of the different processes involved in interaction of urban runoff contaminants is highly essential.

Further, variability in contaminant concentrations in urban stormwater runoff from one location to another has been observed, that is attributed to differences in land use, climatic influences, traffic density, atmospheric deposition, maintenance, road drainage designs, and vehicular traffic density (Brezonik and Stadelmann, 2002). Thus, stormwater management requires collection of local data that ultimately determines the local designing of CWs.

Additionally, stormwater treatment wetlands may pose a risk to the surrounding environment as heavy rainfall events may cause a breakthrough of contaminants and pathogens. Further, the first flush phenomenon generates very high loads of contaminants that are very difficult to be predicted and needs additional amendments in structural designing of CWs. Thus, the long-term efficiency and sustainability of these systems is critically dependent on an integrated understanding of their biological, chemical and hydrological processes as well as hydrometeorological effects of rainfall on them. Furthermore, management plans and budgets need to be prepared at the design stage and provision should be made for resolving unforeseen operational problems.

Natural treatment systems are too often considered to be a build-and-forget solution not requiring further attention at all. However, when denied the minimal amount of maintenance that even natural systems need, failing treatment systems are often reported. Proper functioning of CWs for stormwater treatment also presents various tribulations, such as clogging, toxicity effects of heavy metals and organic contaminants, disposal of biomass enriched with heavy metals, mosquito problems, seasonal growth of aquatic macrophytes, and requirement of a large area to accommodate huge runoff. Therefore, CWs could not be considered as panacea for stormwater pollution; however, in combination with other structural and nonstructural BMPs, effective performance could surely be obtained.

On the whole, for effective and sustainable urban wastewater management, further research is required to define optimal plant species and microorganisms prevalent in wetlands and to quantify rates of individual processes in the field. Laboratory studies are best suited to acquire such data on treatment wetlands. Additionally, genetically modified plants should be considered to enhance the treatment effectiveness of CWs for specific compounds, though biosafety apprehensions are always there. Performance data are also required for microbial activity and contribution of the plants to the overall removal process. The application of approaches and techniques recently developed in related fields, such as contaminant hydrology, environmental microbiology and biotechnology, environmental chemistry, phytoremediation, statistics, and environmental modeling, will tremendously enhance the research on stormwater treatment wetlands and will open new possibilities for process characterization and understanding of treatment wetlands. Future challenges will surely consist of optimizing CWs for more sustainable and reliable treatment of stormwater priority pollutants.

ACKNOWLEDGEMENTS

The first author gratefully acknowledges the financial support provided by Department of Biotechnology, Government of India in the form of research project (BT/PR11033/GBD/27/138/2008).

REFERENCES

Abira, M.A., van Bruggen, J.J.A., and Denny, P. (2005). Potential of a tropical subsurface constructed wetland to remove phenol from pre-treated pulp and papermill wastewater. *Water Sci. Technol.* 51, 173–175.

Aldrich, R.A., and Bartok, J.W. (1994). *Greenhouse engineering,* 3rd rev. ed. NRAES-33. Ithaca, NY: Northeast Regional Agricultural Engineering Service.

Anderson, K.L., Wheeler, K.A., Robinson, J.B., and Tuovinen, O.H. (2002). Atrazine mineralization potential in two wetlands. *Water Res.* 36, 4785–4794.

Anonymous. (2008, July 20). *Destruction of wetlands worsens global warming*. Retrieved from http://news.mongabay.com/2008/0720-wetlands.html

ANZECC. (2000). *Australian and New Zealand Guidelines for Fresh and Marine Water Quality*. Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand. Canberra, ACT, Australia.

Babatunde, A.O., Zhao, Y.Q., O'Neill, M., and O'Sullivan, B. (2008). Constructed wetlands for environmental pollution control: A review of developments, research and practice in Ireland. *Environ. Int.* 34, 116–126.

Bachand, P.A.M., and Horne, A.J. (2000). Denitrification in constructed free-water surface wetlands: II. Effects of vegetation and temperature. *Ecol. Eng.* 14, 17–32.

Balascio, C.C., and Lucas, W.C. (2009). A survey of storm-water management water quality regulations in four Mid-Atlantic states. *J. Environ. Manage.* 90, 1–7.

Bangs, C. (2007). *Urban stormwater treatment by experimental constructed wetlands*. Senior Design Project, Department of Engineering, Swarthmore College.

Barac, T., Taghavi, S., Borremans, B., Provoost, A., Oeyen, L., Colpaert, J.V., Vangronsveld, J., Van der Lelie, D. (2004). Engineered endophytic bacteria improve phytoremediation of water-soluble, volatile, organic pollutants. *Nature Biotechnol.* 22, 583–588.

Barko, J.W., and James, W.F. (1997). Effects of submerged aquatic macrophytes on nutrient dynamics, sedimentation, and resuspension. In Jeppesen, E., Sondergaard, M., Sondergaard, M., and Christoffersen, K. (Eds.), *The structuring role of submerged macrophytes in lakes* (pp. 197–216). New York, NY: Springer-Verlag.

Barten, J.M. (1987). Stormwater runoff treatment in a wetland filter: Effects on the water quality of Clear Lake. In *Lake and reservoir management: Vol. III* (pp. 297–305). Washington, DC: North American Lake Management Society.

Batty, L.C. (2003). Wetland plants—more than just a pretty face? *Land Contam. Reclam.* 112, 173–180.

Best, E.P., Zappi, M.E., Fredrickson, H.L., Sprecher, S.L., Larson, S.L., and Ochman, M. (1997). Screening of aquatic and wetland plant species for phytoremediation of explosives- contaminated groundwater from the Iowa Army Ammunition Plant. *Annal. NY Acad. Sci.* 829, 179–194.

Birch, G.F., Matthai, C., Fazeli, M.S., and Suh, J.Y. (2004). Efficiency of a constructed wetland in removing contaminants from stormwater. *Wetlands* 24, 459–466.

Blecken, G.T., Zinger, Y., Deletic, A., Fletcher, T.D., and Viklander, M. (2009). Impact of a submerged zone and a carbon source on heavy metal removal in stormwater biofilters. *Ecol. Eng.* 35, 769–778.

Brezonik, P.L., and Stadelmann, T.H. (2002). Analysis and predictive models of stormwater runoff volumes, loads, and pollutant concentrations from watersheds in the Twin Cities metropolitan area, Minnesota, USA. *Water Res.* 36, 1743–1757.

Brix, H. (1994). Functions of macrophytes in constructed wetlands. *Water Sci. Technol.* 29, 71–78.

Brix, H. (1997). Do macrophytes play a role in constructed treatment wetlands. *Water Sci. Technol.* 35, 11–17.

Brown, P.A., Gill, S.A., and Allen, S.J. (2000a). Metal removal from wastewater using peat. *Water Res.* 16, 3907–3916.

Brown, P.A., Jefcoat, I.A., Parrish, D., Gill, S., and Graham, E. (2000b). Evaluation of the adsorptive capacity of peanut hull pellets for heavy metals in solution. *Adv. Environ. Res.* 4, 19–29.

Brown, W., and Schueler, T. (1997). *The economics of stormwater BMPs in the Mid-Atlantic region*. Ellicott City, MD: Center for Watershed Protection.

Bulc, T.G., Ojstrsek, A., and Vrhovsek, D. (2006). The use of constructed wetland for textile wastewater treatment. *Proceedings of the of the 10th International Conference on Wetland Systems for Water Pollution Control* 1667–1675.

Bulc, T., and Slak, A.S. (2003). Performance of constructed wetland for highway runoff treatment. *Water Sci. Technol.* 48, 315–322.

Burton, G.A. Jr., Gunnison, D., and Lanza, G.R. (1987). Survival of pathogenic bacteria in various freshwater sediments. *Appl. Environ. Microbiol.* 53, 633–638.

Calheiros, C.S.C., Rangel, A.O.S.S., and Castro, P.K.L. (2007). Constructed wetland systems vegetated with different plants applied to the treatment of tannery wastewater. *Water Res.* 41, 1790–1798.

California Stormwater Quality Association. (2003). *California stormwater BMP handbook: New development and redevelopment*. Menlo Park, CA: CASQA.

Canfield, D.E., and Hoyer, M.V. (1992). *Aquatic macrophytes and their relation to the limnology of Florida Lakes*. Final Report submitted to the Bureau of Aquatic Plant Management, Florida Department of Natural Resources, Tallahassee, Florida.

Carleton, J.N., Grizzard, T.J., Godrej, A.N., and Post, H.E. (2001). Factors affecting the performance of stormwater treatment wetlands. *Water Res.* 35, 1552–1562.

Carr, D.W., and Rushton, B.T. (1995). *Integrating a native herbacious wetland into stormwater management*. Stormwater Research Program, Southwest Florida Water Management District, Brooksville, Florida.

Characklis, G.W., Dilts, M.J., Simmons, O.D. III, Likirdopulos, C.A., Krometis, L.A.H., and Sobsey, M.D. (2005). Microbial partitioning to settleable particles in stormwater. *Water Res.* 39, 1773–1782.

Chevreuil, G.L., Carru, A.M., and Letolle, R. (1990). Urban runoff pollution by organochlorines (polychlorinated-biphenyls and lindane) and heavy-metals (lead, zinc and chromium). *Chemosphere* 21, 1101–1107.

Chow, V.T., Maidment, D.R., and Mays, L.W. (1988). *Applied hydrology*. New York, NY: McGraw-Hill.

Chui, P.C. (1997). Characteristics of stormwater quality from two watersheds in Singapore. *Environ. Monit. Assess.* 44, 173–181.

Collins, B., McArthur, J.V., and Sharitz, R.R. (2004). Plant effects on microbial assemblages and remediation of acidic coal pile runoff in mesocosm treatment wetlands. *Ecol. Eng.* 23, 107–115.

Cooper, P.F., Job, G.D., Green, M.B., and Shutes, R.B.E. (1996). *Reed beds and constructed wetlands for wastewater treatment*. Swindon, England: WRc.

Crabill, C., Donald, R., Snelling, J., Foust, R., and Southam, G. (1999). The impact of sediment fecal coliform reservoirs on seasonal water quality in Oak Creek, Arizona. *Water Res.* 33, 2163–2171.

Crabtree, B., Moy, F., Whitehead, M., and Roe, A. (2006). Monitoring pollutants in highway runoff. *Water Environ. J.* 20, 287–294.

Cui, L.H., Ouyang, Y., Chen, Y., Zhu, X.Z., and Zhu, W.L. (2009). Removal of total nitrogen by *Cyperus alternifolius* from wastewaters in simulated vertical-flow constructed wetlands *Ecol. Eng.* 35, 1271–1274.

CWP. (2007). *National pollutant removal performance*, Database version 3. Center for Watershed Protection, USA.

Davies, C.M., and Bavor, H.J. (2000). The fate of stormwater-associated bacteria in constructed wetland and water pollution control pond systems. *J. Appl. Microbiol.* 89, 349–360

Davis, A.P. (2005). Green engineering principles promote low-impact development. *Environ. Sci. Technol.* 39, 338A–344A.

Davis, A.P., Shokouhian, M., and Ni, S. (2001). Loading estimates of lead, copper, cadmium, and zinc in urban runoff from specific sources. *Chemosphere* 44, 997–1009.

Debo, T.N., and Reese, A.J. (2003). *Municipal stormwater management*. Boca Raton, FL: CRC Press LLC.

DeBusk, T.A., Laughlin, R.B., and Schwartz, L.N. (1996). Retention and compartmentalization of lead and cadmium in wetland microcosms. *Water Res.* 30, 2707–2716.

DeLaney, T.A. (1995). Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. *J. Soil Water Conserv. Sci.* 50, 620–626.

Dias, V.N., Canseiro, C., Gomes, A.R., Correia, B., and Bicho, C. (2006). Constructed wetlands for wastewater treatment in Portugal: A global overview. *Proceedings of the 10th International Conference on Wetland Systems for Water Pollution Control* 91–101.

Dittmer, U., Meyer, D., and Langergraber, G. (2005). Simulation of a subsurface vertical flow constructed wetland for CSO treatment. *Water Sci. Technol.* 51, 225–232.

Dotro, G., Fitch, M., Larsen, D., and Palazolo, P. (2006). Treatment of chromium-bearing wastewaters from tannery operations with constructed wetlands. *Proceedings of the 10th International Conference on Wetland Systems for Water Pollution Control* 1725–1733.

Drizo, A., Twohig, E., Weber, D., Bird, S., and Ross, D. (2006). Constructed wetlands for dairy effluent treatment in Vermont: Two years of operation. *Proceedings of the 10th International Conference on Wetland Systems for Water Pollution Control* 1611–1621.

Du Laing, G., Van Ryckegem, G., Tack, F.M.G., and Verloo, M.G. (2006). Metal accumulation in intertidal litter through decomposing leaf blades, sheaths and stems of *Phragmites australis*. *Chemosphere* 63, 1815–1823.

Eimers, M.C., Buttle, J., and Watmough, S.A. (2008). Influence of seasonal changes in runoff and extreme events on dissolved organic carbon trends in wetland- and upland-draining streams. Impact of flood damage on pollutant removal efficiencies of a subtropical urban constructed wetland. *Can. J. Fish. Aquat. Sci.* 65, 796–808.

Ellis, J.B., Shutes, R.B.E., and Revitt, D.M. (2003). *Constructed wetlands and links with sustainable drainage systems.* Technical Report P2-159/ TR1, Environment Agency.

Eriksson, E. (2002). *Potential and problems related to reuse of water in households.* PhD thesis, Institute of Environment & Resources, Technical University of Denmark.

Eriksson, E., Baun, A., Mikkelsen, P.S., and Ledin, A. (2007a). Risk assessment of xenobiotics in stormwater discharged to Harrestrup Å, Denmark. *Desalination* 215, 187–197.

Eriksson, E., Baun, A., Scholes, L., Ledin, A., Ahlman, S., Revitt, M., Noutsopoulos, C., and Mikkelsen, P.S. (2007b). Selected stormwater priority pollutants: A European perspective. *Sci. Total Environ.* 383, 41–51.

Federal Water Pollution Control Act (Clean Water Act), 62 Stat. 1155, 33 U.S.C. 1251–1376; Pub. Law 845 (1948).

Fink, D.F., and Mitsch, W.J. (2005). Seasonal and storm event nutrient removal by a created wetland in an agricultural watershed. *Ecol. Eng.* 23, 313–325.

Frechen, F.B., Schier, W., and Felmeden, J. (2006). The plant-covered retention soil filter (RSF): The mechanical and biological combined sewer overflow (CSO) treatment plant. *Eng. Life Sci.* 6, 74–79.

Gambrell, R.P. (1994). Trace and toxic metals in wetlands-a review. *J. Environ. Qual.* 23, 883–891.

Garcia, M., and Beâcares, E. (1997). Bacterial removal in three pilot-scale wastewater treatment systems for rural areas. *Water Sci. Technol.* 35, 197–200.

Garrison, A.W., Nzengung, V.A., Avants, J.K., Ellingtion, J.J., Jones, W.J., and Rennels, D. (2000). Phytoremediation of *p,p′*-DDT and the enantiomers of *o,p′*-DDT. *Environ. Sci. Technol.* 3, 13–40.

Gasiunas, V., Strusevicius, Z., and Struseviciéne, M.S. (2005). Pollutant removal by horizontal subsurface flow constructed wetlands in Lithuania. *J. Environ. Sci. Health* 40A, 1467–1478.

Geary, P.M., Mendez, H., and Dunstan, R.H. (2006). *Design considerations in the performance of stormwater devices incorporating constructed wetlands (1833–1842).* Paper presented at the 10th International Conference on Wetland Systems for Water Pollution Control, Lisbon, Portugal, September 23–29.

Gerba, C.P., and McLeod, J.S. (1976). Effect of sediments on the survival of *Escherichia coli* in marine waters. *Appl. Environ. Microbiol.* 32, 114–120.

Gersberg, R.M., Brenner, R., Lyon, S.R., and Elkins, B.V. (1987). Survival of bacteria and viruses in municipal wastewater applied to artificial wetlands. In Reddy, K.R., and Smith, W.H. (Eds.), *Aquatic plants for water treatment and resource recovery* (pp. 237–245). Orlando, FL: Magnolia.

Gersberg, R.M., Gearhart, R.A. and Ives, M. (1989). Pathogen removal in constructed wetlands. In Hammer, D.A. (Ed.), *Constructed wetlands for wastewater treatment* (pp. 431–445). Chelsea, MI: Lewis.

Gonzalez, J., Iriberri, J.M., Egea, J., and Barcina, I. (1990). Differential rates of digestion of bacteria by freshwater and marine phagotrophic protozoa. *Appl. Environ. Microbiol.* 56, 1851–1857.

Goonetilleke, A., Thomas, E., Ginn, S., and Gilbert, D. (2005). Understanding the role of land use in urban stormwater quality management. *J. Environ. Manage.* 74, 31–42.

Greenway, M. (2000). *Role of constructed wetlands for stormwater management*. Paper presented at the Third Queensland Environmental Conference, Brisbane, Australia, May 25–26.

Greenway, M., Jenkins, G., and Polson, C. (2006). Macrophyte zonation in stormwater wetlands-getting it right! (1807–1819). Paper presented at the *10th International Conference on Wetland Systems for Water Pollution Control*, Lisbon, Portugal, September 23–29.

Grismer, M., Carr, M., and Shepherd, H. (2003). Evaluation of constructed wetland treatment performance for winery wastewater. *Water Environ. Res.* 75, 412–421.

Gschlobl, T., Steinmann, C., Schleypen, P., and Melzer, A. (1998). Constructed wetlands for effluent polishing of lagoons. *Water Res.* 32, 2639–2645.

Gu, B., DeBusk, T., Dierberg, F.E., Chimney, M., Pietro, K., and Aziz, T. (2001). Phosphorus removal from Everglades agricultural area runoff by submerged aquatic vegetation/limerock treatment technology: An overview of research. *Water Sci. Technol.* 44, 101–108.

Haberl, R., Grego, S., Langergraber, G., Kadlec, R.H., Cicalini, A.R., Dias, S.M., Novais, J.M., Aubert, S., Gerth, A., Thomas, H., and Hebner, A. (2003). Constructed wetlands for the treatment of organic pollutants. *J. Soils Sediments* 3, 109–124.

Hadad, H.R., Maine, M., and Bonetto, C.A. (2006). Macrophyte growth in a pilot-scale constructed wetland for industrial wastewater treatment. *Chemosphere* 63, 1744–1753.

Ham, J., Yoon, C.G., Kim, H.-J., and Kim, H.-C. (2010). Modeling the effects of constructed wetland on nonpoint source pollution control and reservoir water quality improvement. *J. Environ. Sci.* 22, 834–839.

Hammer, D.A., and Knight, R.L. (1994). Designing constructed wetlands for nitrogen removal. *Water Sci. Technol.* 29, 15–27.

Hanselmann, K.W. (1991). Microbial energetics applied to waste repositories. *Experientia* 47, 645–687.

Hathaway, J.M., and Hunt, W.F. (2010). An evaluation of stormwater wetlands in series in Piedmont, North Carolina. *J. Environ. Eng.* 136, 140–146.

Headley, T.R., Huett, D.O., and Davison, L. (2001). The removal of nutrients from plant nursery irrigation runoff in subsurface horizontal-flow wetlands. *Water Sci. Technol.* 44, 77–84.

Healy, M.G., Rodgers, M., and Mulqueen, J. (2007). Treatment of dairy wastewater using constructed wetlands and intermittent sand filters. *Bioresour. Technol.* 98, 2268–2281.

Herngren, L., Goonetilleke, A., and Ayoko, G.A. (2005). Understanding heavy metal and suspended soilds relationships in urban stormwater using simulated rainfall. *J. Environ. Manage.* 78, 149–158

Higgins, J., and Dechaine, L. (2006). The use of large sub-surface flow wetlands to treat glycol-contaminated stormwater from aircraft deicing operations. *Proceedings of the 10th International Conference on Wetland Systems for Water Pollution Control* 1879–1889.

Hong, M.S., Farmayan, W.F., Dortch, I.J., and Chiang, C.Y. (2001). Phytoremediation of MTBE from a groundwater plume. *Environ. Sci. Technol.* 35, 1231–1239.

Howard, E.A., Emerick, L.C., and Wildeman, T.R. (1989). Design and construction of a research site for passive mine drainage treatment in Idaho Springs, Colorado. In Hammer, D.A. (Ed.), *Constructed wetlands for wastewater treatment* (pp. 761–764). Chelsea, MI: Lewis.

Howell, J.M., Coyne, M.S., and Cornelius, P.L. (1996). Effect of sediment particle size and temperature on fecal bacteria mortality rates and fecal coliform/fecal streptococci ratio. *J. Environ. Qual.* 25, 1216–1220.

Huysman, F., and Verstraete, W. (1993). Water-facilitated transport of bacteria in unsaturated soil columns: Influence of inoculation and irrigation methods. *Soil Biol. Biochem.* 25, 91–97.

Hwang, H.M., and Foster, G.D. (2006). Characterization of polycyclic aromatic hydrocarbons in urban stormwater runoff flowing into the tidal Anacostia River, Washington, DC, USA. *Environ. Pollut.* 140, 416–426.

Imfeld, G., Braeckevelt, M., Kuschk, P., and Richnow, H.H. (2009). Monitoring and assessing processes of organic chemicals removal in constructed wetlands. *Chemosphere* 74, 349–362.

Jones, W.W. (1996). Design features of constructed wetlands for nonpoint source treatment. *Lakeline, North American Lake Management Society* 16, 14–15, 52–53.

Kadlec, R.H. (1992). Hydrological factors in wetland water treatment. In Hammer, D.A. (Ed.), *Constructed wetlands for wastewater treatment: Municipal, industrial and agricultural* (pp. 25–29). Chelsea, MI: Lewis.

Kadlec, R.H. (1997). An autobiotic wetland phosphorus model. *Ecol. Eng.* 8, 145–172.

Kadlec, R.H., and Hey, D.L. (1994). Constructed wetlands for river water quality improvement. *Water Sci. Technol.* 29, 159–168.

Kadlec, R.H., and Knight, R.L. (1996). *Treatment wetlands*. Boca Raton, FL: CRC Press.

Kadlec, R.H., Knight, R.L., Vymazal, J., Brix, H., Cooper, P., and Haberl, R. (Eds.). (2000). *Constructed wetlands for pollution control: Processes, performance, design and operation*. London, England: IWA.

Kadlec, R.H., and Wallace, S.D. (2008). *Treatment wetlands,* 2nd ed. Boca Raton, FL: CRC Press.

Kantawanichkul, S., and Somprasert, S. (2005). Using a compact combined constructed wetland system to treat agricultural wastewater with high nitrogen. *Water Sci. Technol.* 51, 47–53.

Karrh, J.D., Moriarty, J., Kornue, J.J., and Knight, R.L. (2002). Sustainable management of aircraft anti/de-icing process effluents using a subsurface-flow treatment wetland. In Nehring, K.W., and Brauning, S.E. (Eds.), *Wetlands and remediation II* (pp. 187–195). Columbus, OH: Battelle Press.

Kasper, T.M., and Jenkins, G.A. (2007). Measuring the background concentration in a constructed stormwater treatment wetland. *Urban Water J.* 4, 79–91.

Khalil, A., Prudent, P., Bettaieb, M.M., and Domeizel, M. (2005). Pilot treatment plant: Constructed soil reed bed for a cheese dairy farm effluent. In *Book of abstracts of the International Symposium on Wetland Pollutant Dynamics and Control* (pp. 77–78). Ghent, Belgium: Ghent University.

Kivaisi, A.K. (2001). The potential for constructed wetlands for wastewater treatment and reuse in developing countries: A review. *Ecol. Eng.* 16, 545–560.

Knight, R.L. (1993). Ancillary benefits and potential problems with the use of wetlands for nonpoint source pollution control. In Olson, R.K. (Ed.), *Created and natural wetlands for controlling nonpoint source pollution*. Corvallis, Oregon: ManTech Environmental Technologies, Inc., USEPA Environmental Research Laboratory.

Knight, R.L., Gu, B., Clarke, R.A., and Newman, J.M. (2003). Long-term phosphorus removal in Florida aquatic systems dominated by submerged aquatic vegetation. *Ecol. Eng.* 20, 45–63.

Knight, R.L., Kadlec, R.H., and Ohlendorf, H.M. (1999). The use of treatment wetlands for petroleum industry effluent. *Environ. Sci. Technol.* 33, 973–980.

Ko, C.-H., Chang, F.-C.C., Lee, T.-M., Chen, P.-Y., Chen, H.-H., Hsieh, H.-L., Guan, C.-Y. (2010). Impact of flood damage on pollutant removal efficiencies of a subtropical urban constructed wetland. *Sci. Total Environ.* 408, 4328–4333.

Lai, D.Y.F., and Lam, K.C. (2009). Phosphorus sorption by sediments in a subtropical constructed wetland receiving stormwater runoff. *Ecol. Eng.* 35, 735–743.

Landry, G.M., Maranger, R., Brisson, J., and Chazarenc, F. (2009). Greenhouse gas production and efficiency of planted and artificially aerated constructed wetlands. *Environ. Pollut.* 157, 748–754.

Langergraber, G., Haberl, R. (2001). Constructed wetlands for water treatment. *Minerva Biotecnol.* 13, 123–134.

Langergraber, G., Leroch, K., Pressl, A., Rohrhofer, R., and Haberl, R. (2008). A two-stage subsurface vertical flow constructed wetland for high-rate nitrogen removal. *Water Sci. Technol.* 57, 1881–1887.

Lee, B.-H., and Scholz, M. (2007). What is the role of *Phragmites australis* in experimental constructed wetland filters treating urban runoff? *Ecol. Eng.* 29, 87–95.

Leibowitz, S.G., Loehle, C., Li, B.L., and Preston, E.M. (2000). Modeling landscape functions and effects: A network approach. *Ecol. Model.* 132, 77–94.

Line, D.E., Jennings, G.D., Shaffer, M.B., Calabria, J., and Hunt, W.F. (2008). Evaluating the effectiveness of two stormwater wetlands in North Carolina. *Trans. ASABE* 51, 521–528.

Lloyd, S.D., Wong, T.H.F., and Porter, B. (2001). Implementing an ecologically sustainable stormwater drainage system in a residential development. *Water Sci. Technol.* 45, 1–7.

Loucks, O.L. (1990). Restoration of the pulse control function of wetlands and its relationship to water quality objectives. In Kusler, J.A., and Kentula, M.E. (Eds.), *Wetland creation and restoration: The status of the science*. Washington, DC: Island Press.

Lund, M.A., Lavery, P.S., and Froend, R.F. (2001). Removing filterable reactive phosphorus from highly coloured stormwater using constructed wetlands. *Water Sci. Technol.* 44(11–12), 85–92.

Ma, X., and Burken, J.G. (2003). TCE diffusion to the atmosphere in phytoremediation applications. *Environ. Sci. Technol.* 37, 2534–2539.

Maniquiz, M.C., Lee, S., and Kim, L.-H. (2010). Multiple linear regression models of urban runoff pollutant load and event mean concentration considering rainfall variables. *J. Environ. Sci.* 22, 946–952.

Mantovi, P., Piccinni, S., Lina, F., Marmiroli, M., and Marmiroli, N. (2007). Treating wastewaters from cheese productions in H-SSF constructed wetlands. In Borin, M., and Bacelle, S. (Eds.), *Proceedings of the International Conference on Multi Functions of Wetland Systems* (pp. 72–73). Padova, Italy: P.A.N. s.r.l.

Manios, T., Fountoulakis, M.S., and Karathanasis, A.D. (2009). Construction simplicity and cost as selection criteria between two types of constructed wetlands treating highway runoff. *Environ. Manage.* 43, 908–920.

Mbuligwe, S.E. (2005). Comparative treatment of dye-rich wastewater in engineered wetland system (EWSs) vegetated with different plants. *Water Res.* 39, 271–280.

McCutcheon, S.C., and Rock, S. (2001). Phytoremediation. State of the science conference and other developments. *Int. J. Phytorem.* 3, 1–11.

McCutcheon, S.C., and Schnoor, J.L. (2003). *Phytoremediation: Transformation and control of contaminants*. Hoboken, NJ: Wiley.

Meiorin, E.C. (1989). Urban runoff treatment in a fresh/brackish water marsh in Fremont, California. In Hammer, D.A. (Ed.), *Constructed wetlands for wastewater treatment: Municipal, industrial, and agricultural* (pp. 677–685). Chelsea, MI: Lewis.

Merlin, G., Sedilot, C., Herbert, N., and Grasselly, D. (2002). Treatment of tomato greenhouses drainage solutions by experimental constructed wetlands. *Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control* 526–533.

Mitchell, G.F., Hunt, C.L., and Su, Y.M. (2002). Mitigating highway runoff constituents via a wetland. *Transport. Res. Rec.* 1808, 127–133.

Mitsch, W.J. and Gosselink, J.G. (1993). *Wetlands,* 2nd ed. New York, NY: Van Nostrand Reinhold.

Moore, M.T., Bennett, E.R., Cooper, C.M., Smith, S. Jr., Farris, J.L., Drouillard, K.G., and Schulz, R. (2006). Influence of vegetation in mitigation of methyl parathion runoff. *Environ. Pollut.* 142, 288–294.

Mungur, A.S., Shutes, R.B.E., Revitt, D.M., and House, M.A. (1995). An assessment of metal removal from highway runoff by a natural wetland. *Water Sci. Technol.* 32, 169–175.

Murphy, T., Hall, K., and Yasaki, I. (1983). Co-precipitation of phosphate and calcite in a naturally eutrophic lake. *Limnol. Oceanogr.* 28, 28–67.

Nairn, R.W., and Mitsch, W.J. (2000). Phosphorus removal in created wetland ponds receiving river overflow. *Ecol. Eng.* 14, 107–126.

National Research Council. (2008). *Urban stormwater management in the United States*. Washington, DC: National Academy Press.

O'Dell, K.M., VanArman, J., Welch, B.H., and Hill, S.D. (1995). Changes in water chemistry in a macrophyte-dominated lake before and after herbicide treatment. *Lake Res. Manage.* 11, 311–316.

Odum, H.T., Woucik, W., and Pritchard, L. (2000). *Heavy metals in the environment*. Boca Raton, FL: CRC Press.

Oliveri, V.P., Kruse, C.W., Kawata, K., and Smith, J.E. (1977). Microorganisms in urban stormwater. USEPA Report no. EPA-600/2-77–087.

Omari, K., Revitt, M., Shutes, B., and Garelick, H. (2003). Hydrocarbon removal in an experimental gravel bed constructed wetland. *Water Sci. Technol.* 48, 275–281.

Otto, G.M., Clark, M.W., Walker, T.J., and Crisman, T.L. (2000). Reintroduction of wetland functions to the urban landscape: The Stormwater Ecological Enhancement Project. *Verh. Internat. Vereln. Limnol.* 27, 1–6.

Pardue, J.H. (2002). Remediating chlorinated solvents in wetlands: Natural processes or an active approach? In Nehring, K.W., and Brauning, S.E. (Eds.), *Wetlands and Remediation II. Proceedings of the Second International Conference on Wetlands & Remediation* (pp. 1–8). Columbus, OH: Batelle Press.

Perkins, J., and Hunter, C. (1999). An investigation of sanitary indicator bacteria in a macrophyte wastewater-treatment system. *J. Chartered Inst. Water Environ. Manage.* 13, 141–145.

Pontier, H., Williams, J.B., and May, E. (2004). Progressive changes in water and sediment quality in a wetland system for control of highway runoff. *Sci. Total Environ.* 319, 215–224.

Prystay, W., and Lo, K.V. (1996). *An assessment of constructed wetlands for treatment of greenhouse/glasshouse effluents*. Poster presented at the 5th International Conference on Wetland Systems for Water Pollution Control, Universität für Bodenkultur, Vienna, Austria, Poster 26.

Qian, J.H., Zayed, A., Zhu, Y.L., Yu, M., and Terry, N. 1999. Phytoaccumulation of trace elements by wetland plants: III. Uptake and accumulation of ten trace elements by twelve plant species. *J. Environ. Qual.* 28, 1448–1455.

Rai, P.K. (2007). Wastewater management through biomass of *Azolla pinnata*: An ecosustainable approach. *Ambio.* 36, 426–428.

Rai, P.K. (2008). Heavy metal pollution in aquatic ecosystems and its phytoremediation using wetland plants: An ecosustainable approach. *Int. J. Phytorem.* 10:133–160.

Read, J., Wevill, T., Fletcher, T., and Deletic, A. (2008). Variation among plant species in pollutant removal from stormwater in biofiltration systems. *Water Res.* 42, 893–902.

Reddy, K.R., DeLaune, R.D., DeBusk, W.F., and Koch, M.S. (1993). Long-term nutrient accumulation rates in the everglades. *Soil Sci. Soc. Am. J.* 57, 1147–1155.

Reinelt, L.E., and Horner, R.R. (1995). Pollutant removal from stormwater runoff by palustrine wetlands based on comprehensive budgets. *Ecol. Eng.* 4, 77–97.

Revitt, D.M., Worall, P., and Brewer, D. (2001). The integration of constructed wetlands into a treatment system for airport runoff. *Water Sci. Technol.* 44, 469–476.

Richardson, C.J. (1985). Mechanisms controlling phosphorus retention capacity in freshwater wetlands. *Science* 228, 1424–1427.

Richardson, C.J., and Craft, C.B. (1993). Effective phosphorus retention in wetlands: Fact or fiction. In Morshiri, G.A. (Ed.), *Constructed wetlands for water quality improvement* (pp. 271–282). London, England: Lewis.

Rich, L.G. (2005). *Nitrification in aerated lagoons and with intermittent sand filters*. Technical Note 6. Retrieved from http://www.ces.clemson.edu/ees/rich/technotes

Richter, K.M., Guymer, I., Worrall, P., and Jones, C. (2004). Treatment performance of Heathrow Constructed Wetlands. In *Wetland systems and waste stabilization ponds communications of common interest* (pp. 125–131). Lyon, France: ASTEE.

Rivera, R., Warren, A., Curds, C.R., Robles, E., Gutierrez, A., Gallegos, E., and Caldeffin, A. (1997). The application of the root zone method for the treatment and reuse of high strength abattoir waste in Mexico. *Water Sci. Technol.* 35, 271–278.

Rochfort, Q.J., Anderson, B.C., and Crowder, A.A. (1997). Field-scale studies of subsurface flow constructed wetlands for stormwater quality enhancement. *Water Qual. Res. J. Can.* 32, 101–117.

Roper, M.M., and Marshall, K.C. (1974). Modification of the interaction between *Escherichia coli* and bacteriophage in saline sediment. *Microbial Ecol.* 1, 1–13.

Röthlisberger, F. (1996). *Kickuth reed bed technology—the situation in Switzerland with a comparison between technical wastewater treatment and Kickuth reed bed technology*. Paper presented at the 5th International Conference on Wetland Systems for Water Pollution Control, Universität für Bodenkultur, Vienna, Austria.

Rubin, E., and Ramaswami, A. (2001). The potential for phytoremediation of MTBE. *Water Res.* 35, 1348–1653.

Rushton, B., Miller, C., and Hull, C. (1995). Residence time as a pollutant removal mechanism in stormwater detention ponds. In *Proceedings of the 4th Biennial Stormwater Research Conference, Clearwater, FL* (pp. 210–221). Brooksville, FL: Southwest Florida Water Management District.

Rushton, B.T. (1997). *Processes that affect stormwater pollution*. Paper presented at the Fifth Biennial Stormwater Research Conference, Southwest Florida Water Management District, Brooksville, Florida.

Sansalone, J.J. (1996). *Adsorptive infiltration by oxide coated sand media for immobilizing metal elements in runoff*. Ph.D. thesis, University of Cincinnati, Ohio.

Sansalone, J.J., and Buchberger, S.G. (1997). Partitioning and first flush of metals in urban roadway stormwater. *J. Environ. Eng.* 123, 134–143.

Schaad, D.E., Chambers, W.B., Halley, J.M., and Denson, C. (2008). Design and performance of multipurpose constructed wetland and flow equalization basin. *J. Environ. Eng.* 134, 118–125

Schiff, K. (1997). Review of existing stormwater monitoring programs for estimating bight-wide mass emissions from urban runoff. In Weisberg, S., Francisco, C., and Hallock, D. (Eds.), *Southern California coastal water research project annual report 1995–96* (pp. 44–55). Westminster, CA.

Schiff, K., Bay, S.M., Allen, M.J., and Zeng, E. (2000). Southern California. *Mar. Pollut. Bull.* 41, 76–93.

Scholes, L., Revitt, D.M., and Ellis, J.B. (2005). *The fate of stormwater priority pollutants in BMPs*. Public report in the DayWater project.

Scholes, L., Shutes, R.B.E., Revitt, D.M., Forshaw, M., and Purchase, D. (1998). The treatment of metals in urban runoff by constructed wetlands. *Sci. Total Environ.* 214, 211–219.

Scholz, M. (2004). Treatment of gully pot effluent containing nickel and copper with constructed wetlands in a cold climate. *J. Chem. Tech. Biotechnol.* 79, 153–162.

Scholz, M. (2006). *Wetland systems to control urban runoff*. Amsterdam, the Netherlands: Elsevier.

Scholz, M., and Lee, B. (2005). Constructed wetlands: A review. *Int. J. Environ. Stud.* 62, 421–447.

Scholz, M., and Xu, J. (2002). Comparison of constructed reed beds with different filter media and macrophytes treating urban stream water contaminated with lead and copper. *Ecol. Eng.* 18, 385–390.

Schreijer, M., Xampf, R., Toet, S., and Verhoeven, J. (1997). The use of constructed wetlands to upgrade treated sewage effluents before discharge to natural surface water in Texel island, the Netherlands: Pilot study. *Water Sci. Technol.* 35, 231–237.

Schueler, T.R. (1992). *Design of stormwater wetland system: Guidelines for creating diverse and effective stormwater wetlands in the mid-Atlantic region*. Washington, DC: Metropolitan Washington Council of Governments.

Schulz, C., Gelbrecht, J., and Rennert, B. (2003). Treatment of rainbow trout farm effluents in constructed wetland with emergent plants and subsurface horizontal water flow. *Aquaculture* 217, 207–221.

Seidel, K. (1961). Zur Problematik der Keim- und Pflanzengewasserr. *Verh. Internat. Verein. Limnol.*; 14, 1035–1039.

Seidel, K. (1964). Abau von Bacterium coli durch höhere Pflanzen. *Naturwissenschaften* 51, 395.

Seidel, K. (1965a). Phenol-Abbau in Wasser durch *Scirpus lacustris* L. wehrend einer versuchsdauer von 31 Monaten. *Naturwissenschaften* 52, 398–406.

Seidel, K. (1965b). Neue Wege zur Grundwasseranreicherung in Krefeld, Vol. II. Hydrobotanische Reinigungsmethodee. *GWF Wasser/Abwasser*; 30, 831–833.

Seidel, K. (1966). Reinigung von Gewässern durch höhere Pflanzenn. *Naturwissenschaften*; 53, 289–297.

Shatwell, T., and Cordery, I. (1999). Nutrient storage in urban wetlands. In Ellis, B. (Ed.), *Impacts of urban growth on surface water and ground water quality* (Publ. 259, pp. 339–347). Wallingford, England: IAHS.

Sheridan, C., Peterson, J., Rohwer, J., and Burton, S. (2006). Engineering design of subsurface flow constructed wetlands for the primary treatment of winery effluent. *Proceedings of the 10th International Conference on Wetland Systems for Water Pollution Control* 1623–1632.

Shutes, R.B.E. (2001). Artificial wetlands and water quality improvement. *Environ. Int.* 26, 441–447.

Shutes, R.B.E., Revitt, D.M., Lagerberg, I.M., and Barraud, V.C.E. (1999). The design of vegetative constructed wetlands for the treatment of highway runoff. *Sci. Total Environ.* 235, 189–197.

Shutes, R.B.E., Revitt, D.M., Mungur, A.S., and Scholes, L.N.L. (1997). The design of wetland systems for the treatment of urban runoff. *Water Sci. Technol.* 35, 19–25.

Sim, C.H., Yusoff, M.K., Shutes, B., Ho, S.C., and Mansor, M. (2008). Nutrient removal in a pilot and full scale constructed wetland, Putrajaya city, Malaysia. *J. Environ. Manage.* 88, 307–317.

Skene, K.R. (2003). The evolution of physiology and development in the cluster root: Teaching an old dog new tricks? *Plant Soil* 248, 21–30.

Smith, S.E., and Read, D.J. (1997). *Mycorrhizal symbiosis,* 2nd ed. Cambridge, England: Academic Press.

Soderqvist, T. (2002). Constructed wetlands as nitrogen sinks in southern Sweden: An empirical analysis of cost determinants. *Ecol. Eng.* 19, 161–173.

Sörme, L., Bergbäck, B., and Lohm, U. (2001). Goods in the anthroposphere as a metal emission source: A case study of Stockholm, Sweden. *Water Air Soil Pollut. Focus* 1, 213–27.

Srivastava, J., Gupta, A., and Chandra, H. (2008). Managing water quality with aquatic macrophytes. *Rev. Environ. Sci. Biotechnol.* 7, 255–266.

Steuer, J., Selbig, H.W.N., and Prey, J. (1997). *Sources of contamination in an urban basin in Marquette, Michigan and an analysis of concentrations, loads, and data quality*. WRI Report 97-4242. Middleton, WI: U.S. Geological Survey.

Stockdale, E.C. (1991). *Freshwater wetlands, urban stormwater and non-point source pollution: A literature review and annotated bibliography*. Olympia, WA: Washington State Department of Ecology.

Stottmeister, U., Wiesner, A., Kuschk, P., Kappelmeyer, M., and Kaster, M. (2003). Effects of plants and microorganisms in constructed wetlands for wastewater treatment. *Biotech. Adv.* 22, 93–117.

Strecker, E.W., Kersnar, J.M., Driscoll, E.D., and Horner, R.R. (1992). *The use of wetlands for controlling stormwater pollution*. U.S. EPA, Region V. The Terrene Institute.

Sun, G., Gray, K.R., Biddlestone, A.J., Allen, S.J., and Cooper, D.J. (2003). Effect of effluent recirculation on the performance of a reed bed system treating agricultural wastewater. *Proc. Biochem.* 39, 351–357.

Sundaravadivel, M., and Vigneswaran, S. (2001). Constructed wetlands for wastewater treatment. *Crit. Rev. Environ. Sci. Technol.* 31, 351–409.

Susarla, S., Bacchus, S.T., Harvey, G., and McCutcheon, S.C. (2000). Phytotransformations of perchlorate contaminated waters. *Environ. Technol.* 21, 1055–1065.

Swedish EPA. (2000). Environmental Quality Criteria - Lakes and Watercourses. Report 5050.

Taebi, A., and Droste, R. (2004). Pollution loads in urban runoff and sanitary wastewater. *Sci. Total Environ.* 327, 175–184.

Tanner, C.C., Nguyen, M.L., and Sukias, J.P.S. (2005). Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. *Agric. Ecosys. Environ.* 105, 145–162.

Tchobanoglous, G., Burton, F.L., and Stensel, H.D. (2003). *Wastewater engineering: Treatment and reuse* (4th rev. ed.). New York, NY: Metcalf & Eddy.

Teiter, S., and Mander, U. (2005). Emission of N_2O, N_2, CH_4, and CO_2 from constructed wetlands for wastewater treatment and from riparian buffer zones. *Ecol. Eng.* 25, 528–541.

Terzakis, S., Fountoulakis, M.S., Georgaki, I., Albantakis, D., Sabathianakis, I., Karathanasis, A.D., Kalogerakis, N., and Manios, T. (2008). Constructed wetlands treating highway runoff in the central Mediterranean region. *Chemosphere* 72, 141–149.

Tilley, D.R., and Brown, M.T. (1998). Wetland networks for stormwater management in subtropical urban watersheds. *Ecol. Eng.* 10, 131–158.

Thomas, P.R., Glover, P., and Kalaroopan, T. (1995). An evaluation of pollutant removal from secondary treated sewage effluent using a constructed wetland system. *Water Sci. Technol.* 32, 87–93.

Thurston, K.A. (1999). Lead and petroleum hydrocarbon changes in an urban wetland receiving stormwater runoff. *Ecol. Eng.* 12, 387–399.

Trautmann, N.M., Martin, J.H. Jr., Porter, K.S., and Hawk, K.C. Jr. (1989). Use of artificial wetlands for treatment of municipal solid waste landfill leachate. In

Hammer, D.A. (Ed.), *Constructed wetlands for wastewater treatment* (pp. 245–251). Chelsea, MI: Lewis.

Truu, M., Juhanson, J., and Truu, J. (2009). Microbial biomass, activity and community composition in constructed wetlands. *Sci. Total Environ.* 407, 3958–3971.

US Code of Federal Regulations. (1995). *Protection of Environment 40*. Washington, DC.

U.S. Environmental Protection Agency. (1995). *National Water Quality Inventory: 1994 Report to Congress*. EPA/841/R-95/005. Washington, DC: U.S. EPA.

U.S. Environmental Protection Agency. (2000). *National Water Quality Inventory* (EPA-841-R-00-001). Washington, DC: U.S. EPA.

U.S. Environmental Protection Agency. (2002). *National management measures to control nonpoint source pollution from urban areas-draft*. EPA 8421-B-02–003. Washington, DC: U.S. Office of Water.

U.S. Environmental Protection Agency. (2003). *Water: After the storm—weather* (833-B-03-002). Washington, DC: U.S. EPA.

Van der Valk, A.J., and Jolly, R.W. (1992). Recommendations for research to develop guidelines for the use of wetlands to control rural nonpoint source pollution. *Ecol. Eng.* 1, 115–134.

Vymazal, J. (2002). The use of sub-surface constructed wetlands for wastewater treatment in the Czech Republic: 10 years experience. *Ecol. Eng.* 18, 633–646.

Vymazal, J. (2005). Constructed wetlands for wastewater treatment. *Ecol. Eng.* 25, 475–477.

Vymazal, J. (2009). The use of constructed wetlands with horizontal sub-surface flow for various types of wastewater. *Ecol. Eng.* 35, 1–17.

Vymazal, J. (2011). Constructed wetlands for wastewater treatment: Five decades of experience. *Environ. Sci. Technol.* 45, 61–69.

Vymazal, J., and Kröpfelová, L. (2008). *Wastewater treatment in constructed wetlands with horizontal sub-surface flow*. Dordrecht, the Netherlands: Springer.

Walker, D.J., and Hurl, S. (2002). The reduction of heavy metals in a stormwater wetland. *Ecol. Eng.* 18, 407–414.

Walker, W.J., McNutt, R.P., and Maslanka, C.A.K. (1999). The potential contribution of urban runoff to surface sediments of the Passaic river: Sources and chemical characteristics. *Chemosphere* 38, 363–377.

Wang, Q., Cui, Y., and Dong, Y. (2002). Phytoremediation of polluted waters: Potentials and prospects of wetland plants. *Acta Biotechnol.* 22, 199–208.

Wang, X., Han, B.-P., Shi, Y.-Z., and Pang, Z.Q. (2009). Advanced wastewater treatment by integrated vertical flow constructed wetland with *Vetiveria zizanioides* in north China. *Procedia Earth and Planetary Science* 1, 1258–1262.

Weiss, J., Hondzo, M., Biesboer, D., and Semmens, M. (2006). A laboratory study of heavy metal phytoremediation by three wetland macrophytes. *Int. J. Phytorem.* 8, 245–259.

Weiss, P.T., Gulliver, J.S., and Erickson, A.J. (2007). Cost and pollutant removal of storm-water treatment practices. *J. Water Resour. Plann. Manag.* 133, 218–229.

Welker, A. (2006). *Vertical flow constructed wetlands for enhanced CSO treatment-an alternative for elimination of organic pollutants?* Paper presented at the

10th International Conference on Wetland Systems for Water Pollution Control, Lisbon, Portugla, September 23–29.

Werker, A.G., Dougherty, J.M., McHenry, J.L., and Van Loon, W.A. (2002). Treatment variability for wetland wastewater treatment designin cold climates. *Ecol. Eng.* 19, 1–11.

Winnike-McMillan, S.K., Zhang, Q., Davis, L.C., Erickson, L.E., and Schnoor, J.L. (2003). Phytoremediation of methyl tertiary-butyl ether. In McCutcheon, S.C., and Schnoor, J.L. (Eds.), *Phytoremediation-transformation and control of contaminants* (pp. 805–828). Hoboken, NJ: Wiley.

Wittgren, H.B., and Mæhlum, T. (1997). Wastewater treatment wetlands in cold climates. *Water Sci. Technol.* 35(5), 45–53.

White, J.S., Bayley, S.E., and Curtis, P.J. (2000). Sediment storage of phosphorus in a northern prairie wetland receiving municipal and agro-industrial wastewater. *Ecol. Eng.* 14, 127–138.

White, K.D., and Myers, A.L. (1997). Above par storm-water management. *Civil Engineering-ASCE* 67, 50–51.

Wilson, P.C., Whitwell, T., and Klaine, S.J. (2000). Metalaxyl and simazine toxicity to and uptake by *Typha latifolia*. *Arch. Environ. Contam. Toxicol.* 39, 282–288.

Woodward-Clyde Consultants. (1986). *Methodology for analysis of detention basins for control of urban runoff quality.* U.S. Environmental Protection Agency, Office of Water, NTIS #PB93-223576.

Worrall, P., Revitt, D.M., Prickett, G., and Brewer, D. (2002). Constructed wetlands for airport runoff: The London Heathrow experience. In Nehring, K.W., and Brauning, S.E. (Eds.), *Wetlands and remediation II* (pp. 177–186). Columbus, OH: Battelle Press.

Yang, L., and Hu, C.C. (2005). Treatments of oil-refinery and steel-mill wastewaters by mesocosm constructed wetland systems. *Water Sci. Technol.* 51, 157–164.

Zhu, Y.L., Zayed, A.M., Qian, J.H., De Souza, M., and Terry, N. (1999). Phytoremediation of trace elements by wetland plants: II. Water hyacinth. *J. Environ. Qual.* 28, 339–344.

Zurayk, R., Sukkariyah, B., Baalbaki, R., and AbiGhanem, D. (2002). Ni phytoaccumulation in *Mentha aquatica* L. and *Mentha sylvestris* L. *Water Air Soil Pollut.* 139, 355–364.