International Research Journal of Engineering Science, Technology and Innovation (IRJESTI) Vol. 1(2) pp. 57-65, May 2012
Available online http://www.interesjournals.org/IRJESTI
Copyright © 2012 International Research Journals

Full Length Research Paper

Dynamic performance of a constructed wetland to treat lindane-contaminated water

L. A. Ortega-Clemente[a*], V.M. Luna-Pabello[b]

[a]Boca del Río Institute of Technology (ITBoca), Division of Graduate Studies and Research, Carr. Veracruz-Cordoba km. 12, Boca del Río, Veracruz, Mexico. P.O Box 94290.
[b]Facultad de Química, Departamento de Biología, Universidad Nacional Autónoma de México, Ciudad Universitaria, P.O Box 04510, México, D.F.

Accepted 5 June, 2012

This work evaluates the removal/degradation of lindane in water contaminated by agricultural activity. A test to treat lindane-contaminated water was carried out using four SF-CW systems operating simultaneously: two fed with lindane-enriched water (10 mg/L), and the other two fed with lindane-free water as control units. The monitored parameters were: average lindane-removal efficiency (η, %), COD, and BOD_5, which were determined as 63, 43, and 52%, respectively. A correlation between the increase in dissolved oxygen (DO) concentration, and the increase in lindane degradation (10.58-0.91 mg/L) was observed. The average phosphate concentration (2.79 ± 1.07 mg/L) in the effluent was higher than the phosphate concentration plants can assimilate (0.5 mg/L), this lead us to believe there is an inhibition of the removal activity due to phosphate saturation in the system.

Keywords: Constructed wetland, lindane, bioaccumulation, treatment.

INTRODUCTION

Water pollution has been causing hazards to human health, and ecosystems both in large cities and rural areas. The major contaminants in rural area water are pesticides, which are carried down to the rivers by rain and soil erosion, causing the detriment of field fertility due to the abuse of agricultural techniques.

Wetlands, commonly known as biological filters, are an alternative to solve a wide range of environmental and water quality issues (Greenway and Simpson, 1996; Greenway, 1997). There are reports about the use of wetlands to treat a variety of wastewater effluents to remove pollutants, and reduce their possible impact on humans and other ecosystems (Sheoran and Sheoran, 2006).

A constructed wetland system (CW) seeks to mimic the characteristics of natural wetlands in an environment that can be manipulated and controlled (Sheoran and Sheoran, 2006). A CW is a waste water treatment system characterized by being shallow (pond or stream), i.e. no more than 0.60 m depth, which in most cases contain aquatic plants to treat wastewater. These systems have the advantage of requiring little or no energy to operate, compared to alternative systems.

Pesticides are used worldwide to control a variety of pests. The intensive use of pesticides began with the massive development of agriculture and agribusiness, especially after World War II (Liu et al., 2012; Poissant et al., 2008; Leu et al., 2004). Like other xenobiotics, pesticides often have mutagenic, teratogenic, and carcinogenic effects on animals and humans. Different pesticides may distribute into water or accumulate mainly in sediments, with a varying degree of persistence (De la Vega-Salazar et al., 1997; Boone and James, 2003; Peterson et al., 1994; Zaga et al., 1998). Substantial amounts (0.1-5%) of pesticides might be lost from the application fields to the surface waters via runoff and drainage (Liu et al., 2012; Poissant et al., 2008; Leu et al., 2004), as well as volatilization (Poissant and Koprivnjak, 1996; Garmounma and Poissant, 2004; Aulagnier and Poissant, 2005). Of all the pesticides in the market, between 55 and 60% are herbicides, with about 400 organic compounds developed and reported so far.

*Corresponding Author E-mail: alfclemen2002@yahoo.com.mx; Tel./Fax: + 52 22 99860189.

The use of pesticide agrochemicals requires a rational selection and management to prevent toxicities to non-target organisms (Rose et al., 2008; Reichenberger et al., 2007). Of the environmental compartments into which pesticides can partition, contamination of fresh water sources often presents the highest risk because of its ubiquitous use by organisms, and their high potential to direct exposure (Liu et al., 2012; Rose et al., 2008). Organochlorine pesticides (OCPs) have been widely used, globally, since 1940s because of their high effectiveness in controlling pests and diseases (Kalyoncu et al., 2009; Chen et al., 2007). OCPs were the first synthetic organic pesticides to be used in agriculture. DDT and gamma-hexachlorocyclohexane (γ-HCH, Lindane) are probably the best known. The production and intensive agricultural and industrial use of OCPs have caused the widespread contamination of the environment (Kalyoncu et al., 2009; Ayas et al., 2007). Organochlorine pesticides and polychlorinated biphenyls (PCBs) are very persistent compounds, and their persistence in the environment is measured in years. As a result of their persistence and high capacity for accumulation in living organisms (Liu et al., 2012; Kalyoncu et al., 2009; Zaranko et al., 1997; Andersson et al., 2001; Stefanelli et al., 2004; Kalyoncu et al., 2009), the use of the majority of OCP has been banned in developed countries.

The contamination of water bodies with agricultural pesticides can pose a significant threat to aquatic ecosystems, and drinking water resources (Dabrowski et al., 2003; Reichenberger et al., 2007). However, the risk to the aquatic community or the human health can often be substantially reduced by appropriate measures. Mitigation of pesticide inputs into water bodies includes both reduction of diffuse-source (runoff and erosion, tile drainage, spray drift, leaching to groundwater), and of point-source inputs (mainly farmyard runoff), which in some regions of Europe (e.g. Western Germany, Sweden) have been shown to make a highly significant contribution to the observed pesticide loads in rivers (Reichenberger et al., 2007; Kreuger and Nilsson, 2001; Jaeken and Debaer, 2005; Rose et al., 2006, 2008).

Natural and artificial water bodies, such as CWs, are highly likely to receive contaminated treated and non-treated water resulting in detriment of quality, and thus the decrease in native biota. In most studies with CWs, a distinction between bioattenuation and physical/chemical removal of contaminants cannot be established, mainly because said removal is commonly determined based on differences in input and output, considering wetlands as a black box. In particular, the hydraulic retention time (HRT) of pesticides in CWs is the main parameter to be considered in the performance of a biological treatment. Contrary to most plant treatments where flow rates are in a narrow range of values, in the case of CWs, flow rates are often close to zero within a few hours after a storm event, while flow rates are very high during storm events, showing lower hydraulic retention times than the time needed for biological treatment (Braeckevelt et al., 2011; Holvoet et al., 2007; Warren et al., 2003, 2002). Therefore, both the pesticide holding time in CWs and the close contact between pesticides and plants (and/or microorganisms) should be improved while designing biological removal processes. Macrophytes play a role in biological removal processes due to their ability to extract metals and/or organic compounds (Gregoire et al., 2009). An appropriate hydraulic design (hydraulic retention time), and the use of adsorbing materials may contribute to increase the pesticides holding time, and the contact between pesticides and biocatalysts. Pesticide fluxes can be reduced by 50–80% when increasing the retention time by ten times. This, in turn, leads to CW lengths that are much longer than those for municipal wastewater treatment plants (Gregoire et al., 2009). Thus, the objective of this work was to evaluate the removal/degradation of lindane present in water contaminated by agricultural activity, using a constructed wetland of the free surface flow type as the treatment system.

MATERIALS AND METHODS

Field of Study

The field of study for this research was the hydrological area of the Papaloapan Basin. This is the most important watershed in Mexico having an extension of 51,025.52 km², and it goes beyond three (3) Mexican states (Oaxaca, Puebla, and Veracruz). It has fertile soil, and a hot humid climate, which are favorable factors for the development of agriculture, cattle raising, fishing, and the sugar industry (sugar cane). The Papaloapan river basin is second ranked in terms of volume in Mexico. This river flows into the Gulf of Mexico, mainly coming out from the cities of Tuxtepec (Oaxaca), Alvarado, Tlacotalpan, and Cosamaloapan (in Veracruz).

Bioaccumulation Test Plant

The three herbaceous plant species selected for the lindane bioaccumulation test were native to natural wetlands in the field of study. *Typha domingensis*, *Echinochloa pyramidalis*, and *Sagittaria lancifolia* were collected in the region of Alvarado and Boca del Rio, in Veracruz State. These three species are monocots of the homorrizia type, which have a secondary root system developing adventitious roots that provide the ability to reproduce asexually by rhizome extension, and generation of new outbreaks. These species are common in freshwater wetlands.

Transplants were acclimatized to green house cond-

itions (18-34 °C) for at least one month, in a semi-continuous regime of humidity (flooded wetland model). For the bioaccumulation and lindane-tolerance studies, the plant species were exposed to lindane.

The plant species were kept in contact with lindane in contaminated soil at a concentration of 17 mg lindane/kg dry soil. The contaminated soil (5 kg) was placed in pots, plants were planted and subsequently transplanted (when smaller than 25 cm), and finally flooded with water (3 cm depth). The exposure time was 18 days. For the analysis of bioaccumulated lindane, plant samples were removed every third day (destructive test), separating root, stem, and leaves for further analysis.

Dynamic Performance of a Constructed Wetland

The study system comprised: a feeding tank; the experimental surface-flow constructed wetland (SF-CW) unit; a treated water tank; and a continuous aeration system (which starts operating as of Day 7). This system utilized the plant species selected from the bioaccumulation test. The study model was comprised of four SF-CWs operating simultaneously: two experimental units (model and replicate) fed with lindane-enriched water (10 mg/L); and two control units fed with lindane-free water.

The SF-CWs were comprised of acrylic cells of 0.50 x 0.30 x 0.20 m (LxWxD) simulating a laboratory scale SF-CW system. Each SF-CW was fed with influent pumped through PVC pipes running horizontally through the SF-CWs simulating a plug flow. The contaminated and control SF-CWs systems were respectively packed with the following: a primary layer (bottom) of fine-sized silica sand (2.00 mm, # 10), 0.04 m depth; a secondary layer of natural soil from the field of study, free of lindane, 0.03 m depth; and a tertiary layer of intermediate sized silica sand (4.76 mm, # 4), 0.03 m depth. The SF-CWs also included the plant species selected to be at least 0.25 m high (2 per SF-CW), and spaced apart by 0.15 m. The experimental units were flooded (0.06 m flood level) with 9 L (0.009 m3) of influent as the system operating volume, and were acclimated for 20 days, and finally started up and operated. The feeding flow rate was 3.47 E^{-5} L/s to achieve a hydraulic retention time (HRT) of 3 days.

The sampling of water samples (500 mL) was made every 3 days during the 62 days of operation; the plant samples (washed and dried whole root, stem, and leaves) were taken at the end of the 62-day period. The following-up and analysis parameters of the SF-CWs were: lindane, $N\text{-}NO_2$, $N\text{-}NO_3$, $N\text{-}NH_4$, PO_4^{-3}, COD, BOD_5, DO, reported in mg/L, respectively, temperature (°C), and pH for the water samples; and for the plant samples, mg lindane/kg plant.

Analysis

The lindane concentration in the plant (root, stem, and leaves) was determined as per the method described by Waliszewski et al., (1985), modified for microwave oven. The plant samples (200 g) were placed in Ziploc bags, and frozen at -40 °C in a deep freezer. The frozen samples were dehydrated in a Thermo Savant Modulyo D-114 Freeze-dryer for 24 hours at -49 °C, and 36x10-3 mbar. The dried samples were milled, and placed back in the Ziploc bag for storage. The extraction was carried out on 10 g of dry sample by placing it in a microwavable Teflon vessel, and adding 20 mL of acetone and 20 mL of hexane. Then microwaved in a CEM model MARSX Microwave oven for 10 min. Then, 20 mL of hexane and 20 mL of acetone were added again to the sample in the Teflon vessel, and filtered through a glass funnel using filter paper No. 4 (Whatman), and collecting the filtrate in a 250-mL flask, to separate solids. 50 ml of a 5% sodium sulfate aqueous solution was added to the filtered extract, and the mixture was transferred to a separatory funnel. The mixture was extracted three times with 30 mL of hexane. The hexane layer was filtered into a 250-mL flat-bottom balloon flask through a 5 cm layer of sodium sulfate (10 g activated sodium sulfate at 660 °C for 48 hours), then concentrated in a rotary evaporator (Buchi Model R-114) to 5 mL. The extract was purified in a separatory funnel by adding 1 mL of concentrated sulfuric acid, and vigorously shaking for 1 minute. It was let stand for 3 minutes until the phases separated, and the organic phase was filtered again through a layer of sodium sulfate. The sodium sulfate was then rinsed twice with 50 mL of hexane, and the rinses were collected into a balloon flask, and concentrated to 1 mL. The extract was transferred to a volumetric vial, and diluted to the final volume of 1.5 mL with hexane. Sulfuric acid was used in purifying the extract because it causes organic substances to precipitate, and hydrolyzes the endogenous organic compounds/pesticide complex. The extract was used to quantify the pesticide by gas chromatography.

Lindane was extracted from the effluent by following the method described by EPA method 508 and Standard Methods (1985), which was adapted. A 100 mL effluent sample was measured in a graduated cylinder, and transferred into a 250-mL separatory funnel, then 20 mL of hexane and 5 mL of sodium sulfate saturated solution were added, and the mixture was vigorously shaken for 3 min, and left stand until phase separation. The aqueous phase was drained into the same graduated cylinder used to measure the sample, and the hexane phase was transferred into a flat-bottom balloon flask (250 mL), the aqueous phase was transferred back to the separatory funnel, and 10mL of hexane and 5 mL of sodium sulphate saturated solution were added (stirring the mixture). This procedure was repeated twice, and the hexane extracts were collected in the flask. The organic phase was

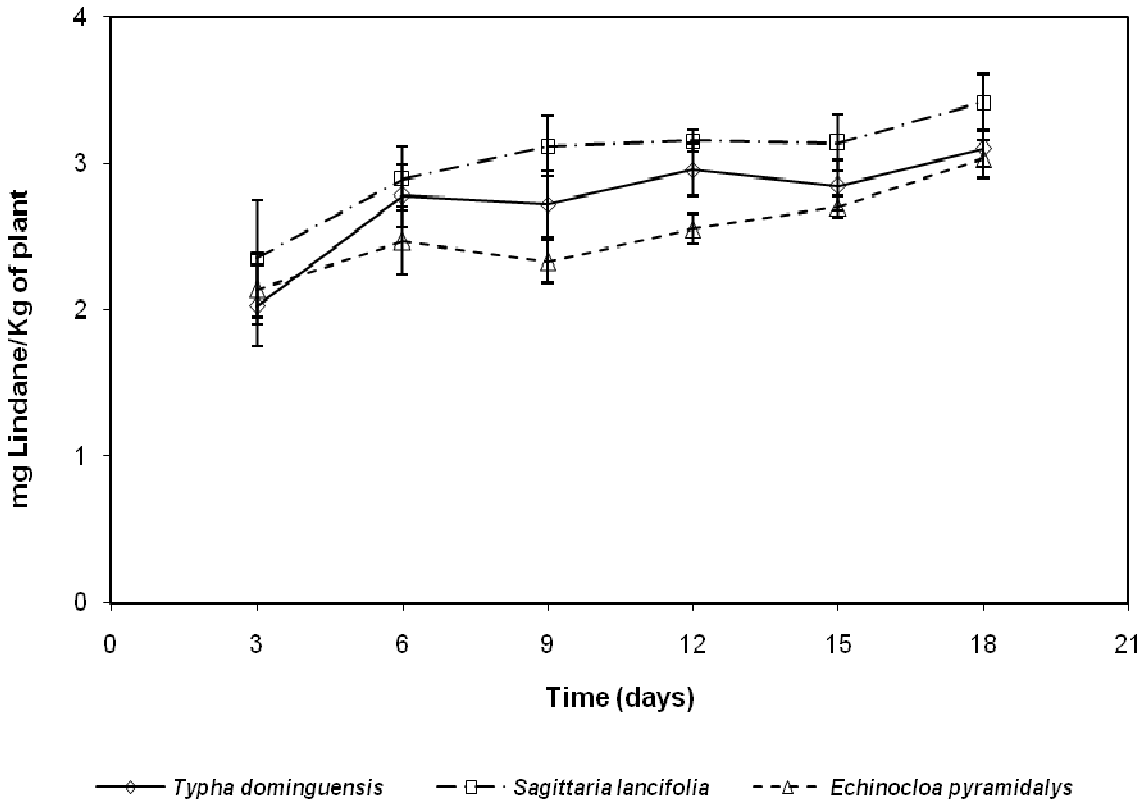

Figure 1. Kinetics of lindane bioaccumulation in: ◊ *Typha domingensis,* □ *Sagittaria lancifolia,* Δ *Echinochloa pyramidalis*

passed through a sodium sulfate column of 5 cm (10 g activated sodium sulfate at 660 °C for 48 hours). The organic phase was collected into a balloon flask, and then concentrated to 5 mL in a rotating evaporator. The concentrated extract was purified by adding 1 mL of concentrated sulfuric acid, and vigorously stirred for 1 minute. It was left stand for 3 minutes to separate phases, and then the organic phase was filtered through a sodium sulfate layer. The sodium sulfate was rinsed with 50 mL of hexane, and the rinses were collected in a balloon flask, and concentrated again to 1 mL. The concentrated extract was transferred to a graduated vial, and diluted to a final volume of 1.5 mL with hexane. The pesticide content of the extract was quantified by gas chromatography.

The lindane concentration was determined using a Trace GC Ultra, Thermo Scientific gas chromatograph with an electron capture detector (ECD), and a capillary column "Thermo TR-1701 (30 m x 0.32 mm ID X 0.25 µm, Thermo Scientific)". Nitrogen was used as carrier gas (99.99%) at a rate of 1.2 mL/min. The oven temperature was set for the range of 60 °C (1 min) to 280 °C, at a rate of 15 °C/min, 280 °C (1 min). The temperature of the injector operating in splitless mode (injected volume 1 µL) was 300 °C, and the ECD detector temperature was 350 °C.

Calculation Formulae for the Removal Efficiencies

The conventional removal efficiency (η, %) of Lindane, Nitrites ($N\text{-}NO_2$), Nitrates ($N\text{-}NO_3$), Ammonium ($N\text{-}NH_4$), Phosphate (PO_4^{-3}), Chemical Oxygen Demand (COD), and Biochemical Oxygen Demand (BOD_5) were determined according to Eq. (1):

$$\eta = \left[\frac{(P_i - P_e)}{P_i}\right] * 100 \qquad (1)$$

Where Pi = baseline contamination value in the constructed wetland (influent), Pe = final contamination value in the constructed wetland (effluent).

RESULTS AND DISCUSSION

The results of lindane bioaccumulation per plant species are shown in Figure 1, whereas the average results are shown in Table 1.

Sagittaria lancifolia presents greater lindane bioaccumulation after 18 days of exposure, compared with the other plant species. Furthermore, the bioaccumulation in *Sagittaria lancifolia* and *Typha domingensis* is 6 and 12, respectively, which does notseems to be a significant difference. *Echinochloa pyramidalis*

Table 1. Lindane bioaccumulation in root, stem, and leaf per plant species.

Plant species/ (mg lindane/Kg plant)	Root	Stem	Leaf
Typha domingensis	1.99 ± 0.37	-------	0.75 ± 0.09
Sagittaria lancifolia	2.22 ± 0.29	0.74 ± 0.12	0.07 ± 0.01
Echinochloa pyramidalis	1.59 ± 0.35	0.89 ± 0.16	0.09 ± 0.01

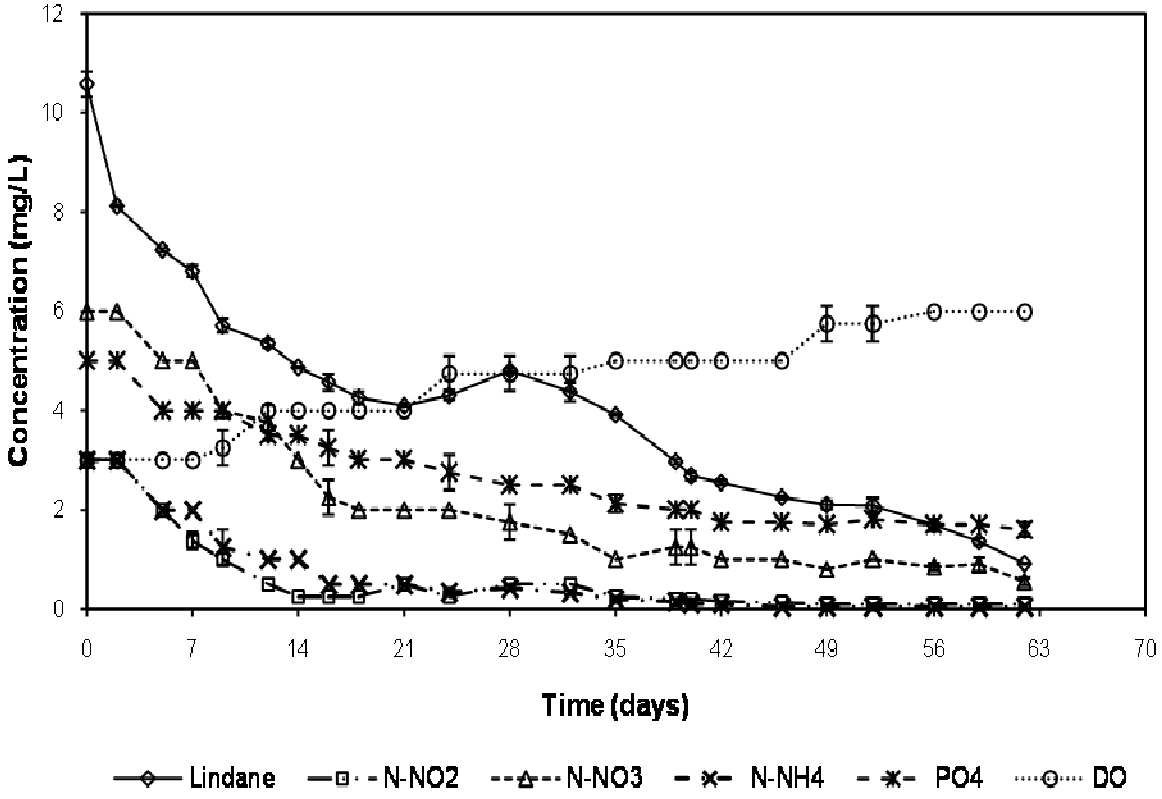

Figure 2. Dynamic performance of removal of Lindane, Nitrites ($N\text{-}NO_2$), Nitrates ($N\text{-}NO_3$), Ammonium ($N\text{-}NH_4$), Phosphate (PO_4^{-3}), and Dissolved Oxygen (DO) in contaminated constructed wetlands to 62 days of operation and a HRT of 3 days.

shows a linear tendency to bioaccumulate; however, it performs below *Sagittaria lancifolia* and *Typha domingensis.*

The root showed higher concentration of lindane as detected in the three plant species. The concentration of lindane bioaccumulation in the root was 2.22 ± 0.29, 1.99 ± 0.37, and 1.59 ± 0.35 mg/kg plant (dry basis) in *Sagittaria lancifolia* , *Typha domingensis,* and *Echinochloa pyramidalis*, respectively. Lindane bioaccumulation in the stem was slightly higher for *Echinochloa pyramidalis* than for *Sagittaria lancifolia* . Moreover, lindane bioaccumulation in leaves was significantly higher for *Typha domingensis* (Table 1).

Figure 2 shows the dynamic performance analysis of the removal (mg/L) of lindane, nutrients ($N\text{-}NO_2$, $N\text{-}NO_3$, $N\text{-}NH_4$, PO_4^{-3}), and dissolved oxygen, for 62 days of operation of the constructed wetlands with a HRT of 3 days.

Most biochemical reactions that convert pollutants take place in the solid-liquid interphase: such as in the granular medium, the liquid phase, the rhizosphere, and particularly, the biofilm. On the other hand, the set of physical/chemical (abiotic) or microbiological (biotic) processes that occur in a constructed wetland, show a reduction in the number of pathogenic microorganisms, and therefore, the constructed wetland systems show a very good efficiency to eliminate pathogens to public health and aquatic life, by improving the sanitary quality of treated effluents. (Gregoire et al., 2009; Dabrowski et al., 2003; Reichenberger et al., 2007).

The results of the dynamic performance of the parameters under study show that the concentration

Table 2. Average results for the study parametersto 62 days of operation, the dynamic performance of the constructed wetlands (contaminated and control), and comparison with standards for the protection of aquatic life and for water for human use and consumption (minimum and maximum measured).

Parameter (mg/L)	Influent CW[a] Input Contaminated	Influent CW Input Control	Effluent CW output Contaminated	Effluent CW output Control	NOM-001- SEMARNAT-1996. Protection of aquatic life. Monthly average. (Mexico)	NOM-127-SSA1-1994. Water for human use and consumption. Permissible limits of quality and treatments to be applied to water for drinking (Mexico)	Directive 78/659/EEC concerning the quality of fresh waters needing protection to be suitable for fish life (European Economic Community)	Current Recommendation on National Water Quality Criteria. Download in lakes for the protection of aquatic life. Region XIII (USA)
Lindane	10.58 ± 0.25 (10.75-10.40)	0.05 ± 0.001 (0.06-0.05)	4.24 ± 2.35 (10.58-0.91)	0.05 ± 0.001 (0.06-0.05)	Unspecified	0.2	Unspecified	Unspecified
$N\text{-}NH_4$	3.00 ± 0.00	3.00 ± 0.00	0.73 ± 0.92 (3.00-0.05)	0.52 ± 0.72 (3.00-0.02)	Unspecified[b]	0.5	<1	<0.5
$N\text{-}NO_2$	3.00 ± 0.00	3.00 ± 0.00	0.64 ± 0.87 (3.00-0.10)	1.02 ± 1.04 (3.00-0.10)	Unspecified	1	<0.09	<0.05
$N\text{-}NO_3$	6.00 ± 0.00	6.00 ± 0.00	2.34 ± 1.74 (6.00-0.55)	2.36 ± 1.77 (6.00-0.50)	Unspecified	10	<0.3	<0.5
PO_4^{-3}	5.00 ± 0.00	5.00 ± 000	2.79 ± 1.07 (5.00-1.60)	2.78 ± 0.99 (5.00-1.5)	Unspecified[c]	Unspecified	<1.5	<0.5
COD	295 ± 0.00	295 ± 0.00	173.48 ± 82.51 (295-55)	158.04±80.15 (295-53)	Unspecified	Unspecified	<125	40
BOD_5	140 ± 0.00	140 ± 0.00	70.28 ± 29.20 (140-28)	71.20 ± 28.59 (140-25)	30	Unspecified	Unspecified	Unspecified
DO	3.00 ± 0.00	3.00 ± 0.00	4.52 ± 1.03 (3-6)	4.78 ± 1.05 (3-6)	Unspecified	Unspecified	1<	Unspecified
Temperature (°C)	36.00 ± 0.00	36.00 ± 0.00	29.17± 3.04 (36-27.5)	29.17± 3.04 (36-27.5)	< 40	Unspecified	10-28	6-30
pH (unit)	7.80 ± 0.00	7.80 ± 0.00	7.29 ± 0.29 (7.8-7.6)	7.27 ± 0.24 (7.8-7.3)	Unspecified	6-9	6-9	6.5-9

[a]Constructed wetlands, [b]The Mexican Standard specifies values for nitrogen and total nitrogen (TN), and establishes a maximum permissible limit value of 15 mg/L, [c]Mexican Standard specifies, as total phosphorus (TP), and establishes a maximum permissible limit of 5 mg/L. In domestic wastewater treated biologically, the PO_4^{-3} is approximately 80% of TP.

Table 3. Average results for the removal efficiency (η, %) of the study parameters for constructed wetlands (contaminated and control), for 62 days of operation and a HRT of 3 days.

Removal efficiencies (η, %)	Contaminated	Control
Lindane	62.59 ± 18.38	0.00 ± 0.00
$N\text{-}NH_4$	79.26 ± 26.53	86.47 ± 15.93
$N\text{-}NO_2$	82.11 ± 24.17	69.17 ± 32.33
$N\text{-}NO_3$	63.79 ± 26.51	63.48± 27.05
PO_4^{-3}	46.34 ± 19.79	46.39 ± 17.86
COD	43.07 ± 27.11	48.54 ± 25.81
BOD_5	52.14 ± 18.36	51.38 ± 17.80

(mg/L) in the effluent decreased as the time of operation increased, and in general, all the parameters follow the same pattern. It can also be observed that an increase in the concentration of dissolved oxygen improves the degradation of lindane (10.58-0.91 mg/L), and the consumption of nutrients ($N\text{-}NO_2$, $N\text{-}NO_3$, $N\text{-}NH_4$, PO_4^{-3}) is accelerated. It is noteworthy that the degree of removal of nutrients can still be improved in these systems, especially for nitrogen, if there were more oxygen supply in the site of the rhizomes, thereby causing a greater transformation of $N\text{-}NH_4$ into $N\text{-}NO_3$, and allowing a better nitrogen assimilation and consumption by the plant. Furthermore, the obtained average phosphate concentration in the effluent (2.79 ± 1.07 mg/L) showed higher values than the concentration plants can assimilate (0.5 mg/L). This might lead to think that there is an inhibition in the phosphate removal by the system saturation. Most plants preferably absorb nutrients through their roots; aquatic plants generally absorb nutrients directly from the water column through their stems and leaves (Sutton and Latham, 1996; Barko et al., 1988; Moeller et al., 1988). For the 62-day operation, the average results of nutrients and contaminant concentrations for the input (influent) and output (effluent) are reported in Table 2.

In relation to $N\text{-}NH_4$, it is observed that $N\text{-}NH_4$ concentrations in the SF-CWs were comparatively lower than that for $N\text{-}NO_3$ (3 and 6 mg/L, respectively). This can be explained by the action of nitrifying bacteria in the water, said water was taken from the Jamapa river in the municipality of Boca del Rio, Veracruz; and by the aerobic conditions that prevailed in the SF-CWs (4.52 ± 1.03 mg DO/L) due to the continuous aeration through diffusers. On the other hand, it should be noted that several authors point out that 2 mg/L of DO are sufficient to efficiently carry out the nitrification process (Sawaittayothin and Polprasert, 2007; Tanner and Kadlec, 2003).

The average $N\text{-}NO_2$ concentration of the influent to the SF-CWs was 3 mg/L, which is consistent with that expected and reported for similar environments (Burrell et al., 1998). The rapid decrease in the $N\text{-}NO_2$ concentration could be explained by the rapid conversion into $N\text{-}NO_3$ between days 7 and 21 of the operation (Figure 1), under favorable environmental conditions, such as temperature (32 °C), and the presence of nitrifying bacteria, and dissolved oxygen. For the 62-day treatment assessment, the removal efficiency (η, %) of $N\text{-}NO_2$ was approximately 82% (Table 3), being the highest removal efficiency obtained among all the study parameters.

The COD in the influent was 295 mg O_2/L in the SF-CWs (Table 2). This can be explained as a result of the discharge of wastewater from some surrounding industries (sugar extraction industry), which in spite of not being too close to the study field, do carry contaminants into the Jamapa river and into the sea through Boca del Rio, Veracruz. This is in addition to the substantial contaminating (pesticides) contributions made by diffuse discharges of wastewater by farms and households, as well as those resulting from tourism activities taking place in the area. Based on these results, Figure 3 and Table 3 report the average efficiency of removal after 62 days of operation. The average value for COD was 43%, achieving the maximum efficiency level at the end of day 62, i.e. 80%. Likewise, the results for BOD_5 removal efficiency are slightly above the average for COD, the average being 52% (Table 3), and 80% at day 62. After day 49 there was no significant difference between BOD_5 and COD removal efficiencies (Figure 3).

The quality of water in healthy rivers and lakes as per the international standards and the maximum values for ammonium ($N\text{-}NH_4$) should be between 0.5 and 1 mg/L, while for soluble orthophosphate (PO_4^{-3}) between 0.5 and 1.5 mg/L (Beavers and Tully, 2005; EEA, 2000). For Mexico, the Mexican Official Standard NOM-001-SEMARNAT-1996 indicates for the protection of aquatic life a maximum allowable limit for the monthly average concentration of 5 mg/L, as total phosphorus, and 15 mg/L as total nitrogen, for discharges into rivers and reservoirs (Table 2). Moreover, the Mexican Official Standard NOM-127-SSA1-1994 for environmental health, water for human use and consumption, sets out a maximum allowable limit of lindane of 0.2 mg/L.

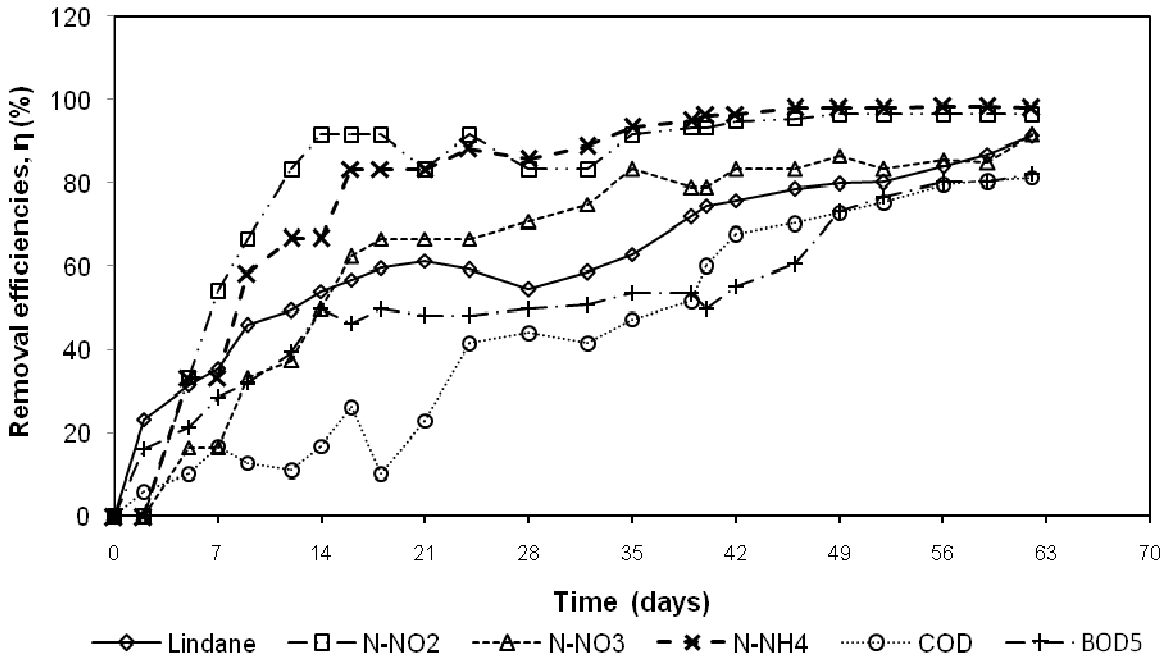

Figure 3. Removal efficiencies of Lindane, Nitrites ($N\text{-}NO_2$), Nitrates ($N\text{-}NO_3$), Ammonium ($N\text{-}NH_4$), Chemical Oxygen Demand (COD), and Biochemical Oxygen Demand (BOD_5) in contaminated constructed wetlands to 62 days of operation and a HRT of 3 days.

Therefore, the use of constructed wetlands to treat polluted rivers and lakes contaminated with pesticides is technically an economical and environmentally feasible system. Among the advantages of these systems compared to conventional treatments, are the low operating costs, and low generation of unwanted byproducts. The treating process of this approach is based on the interaction of its major constituents: support medium, vascular plants, and microorganisms. The combined activities of these three components allow the removal of pollutants, including pesticides, by means of physical, chemical and biological mechanisms, leading in turn, to the establishment of an ecosystem equivalent to a natural wetland (Beavers and Tully, 2005).

CONCLUSION

The increase in dissolved oxygen concentration enhances the degradation of lindane (10.58-0.91 mg/L), and accelerates the consumption of nutrients. The average phosphate concentration in the effluent (2.8 mg/L) was higher than the concentration plants can assimilate (0.5 mg/L), leading us to believe there is an inhibition of the removal activity. In 62 days of treatment, the individual average removal efficiency for lindane, COD, and BOD_5 were 63, 43, and 52%, respectively. On the other hand, the $N\text{-}NO_2$ removal efficiency was approximately 82%, which was the highest among all other study parameters.

ACKNOWLEDGEMENTS

The authors wish to thank PAIP (FQ-VMLP-2009, 2010) and PAPIIT (IN107209) for the financial support given to research work in wetlands, as well as to DGAPA - UNAM for the Post-Doctoral Fellowship awarded to Dr. Ortega-Clemente.

REFERENCES

Andersson PL, Berg AH, Bjerselius R, Norrgren L, Olsen H, Olsson PE, Orn S, Tysklind M (2001). Bioaccumulation of selected PCBs in zebrafish, three-spined stickleback and Artic char three different routes of exposure. Arch. Environ. Contam. Toxicol. 40: 519-530.

Aulagnier F, Poissant L (2005). Some pesticides occurrence in air and precipitation in Québec, Canada. Environ. Sci. Technol. 39: 2960-2967.

Ayas Z, Ekmekci G, Ozmen M, Yerli SV (2007). Histopathological changes in the livers and kidneys of fish in Sariyar Reservoir, Turkey. Environ. Toxicol. Pharm. 23: 242-249.

Barko JW, Smart RM, McFarland DG, Chen RL (1988). Interrelationships between the growth of Hydrilla verticillata. Royle and sediment nutrient availability. Aquat. Bot. 32: 205-216.

Beavers PD, Tully IK (2005). Nutrient reduction evaluation of sewage effluent treatment options for small communities. Water Sci. Technol. 51(10): 221- 229.

Boone MD, James SM (2003). Interactions of an insecticide, herbicide, and natural stressors in amphibian community mesocosms. Ecol. Appl. 13(3): 829-841.

Braeckevelt M, Reiche N, Trapp S,Wiessner A, Paschke H,Kuschk P, Kaestner M (2011). Chlorobenzene removal efficiencies and removal processes in a pilot-scale constructed wetland treating contaminated groundwater. Ecol. Eng. 37 (6): 903-913.

Burrell PC, Keller J, Blackall J (1998). Microbiology of a nitrite oxidizing

bioreactor. Appl. Environ. Microbiol. 64(5): 1878-1883.
Chen S, Lili S, Zhengjun S, Qiuhui H(2007). Determination of organochlorine pesticide residues in rice and human and fish fat by simplified two-dimensional gas chromatography. Food Chem. 104: 1315-1319.
Dabrowski JM, Schulz R (2003). Predicted and measured levels of azinphosmethyl in the Lourens river, South Africa: comparison of runoff and spray drift. Environ. Toxicol Chem. 22: 494-500.
De la Vega-Salazar MY, Martinez-Tabche L, Macias-Garcia C (1997). Bioaccumulation of methyl parathion and its toxicology in several species of the freshwater community in Ignacio Ramirez Dam in Mexico. Ecotoxicol. Environ. Saf. 38: 53-62.
EEA (2000).Nutrients in European ecosystems. Environmental Assessment Report No 4. European Environment Agency, Copenhagen, Denmark.
Garmouma M, Poissant L (2004). Occurrence, temperature and seasonal trends of a- and g-HCH in air (Quebec, Canada). Atmos. Environ. 38: 369-382.
Greenway M, Simpson JS(1996). Artificial wetlands for wastewater treatment, water reuse and wildlife in Queens Land, Australia. Water Sci. Technol. 33(10-11): 221-229.
Greenway M (1997). Nutrient content of wetland plants in constructed wetlands receiving municipal effluent in tropical Australia. Water Sci. Technol. 35(5): 135-142.
Gregoire C, Elsaesser D, Huguenot D, Lange J, Lebeau T, Merli A, Mose R, Passeport E, Payraudeau S, Schutz T, Schulz R, Tapia-Padilla G, Tournebize J, Trevisan M, Wanko A (2009). Mitigation of agricultural non-point-source pesticide pollution in artificial wetland ecosystems. Review. Environ. Chem. Lett. 7(3): 205-231.
Holvoet KMA, Seuntjens P, Vanrolleghem PA(2007). Monitoring and modeling pesticide fate in surface waters at the catchment scale. Review. Ecol. Model. 209: 53-64.
Jaeken P, Debaer C (2005). Risk of water contamination by plant protection products (PPP) during pre- and post treatment operations. Annu. Rev. Agric. Eng. 4: 93-114.
Kalyoncu L, Arca I, Aktunmsek A (2009). Some organochlorine pesticide residues in fish species in Konya, Turkey. Chemosphere. 74: 885-889.
Kreuger J, Nilsson E (2001). Catchment scale risk-mitigation experiences key issues for reducing pesticide transport to surface waters. Pesticide behaviour in soil and water, vol. 78. BCPC Symposium Proceedings. pp. 319-324.
Leu C, Singer H, Stamm C, Muller SR, Schwarzenbach RP (2004). Variability of herbicide losses from 13 fields to surface water within a small catchment after a controlled herbicide application. Environ. Sci. Technol. 38 (14): 3835-3841.
Liu G, Li J, Yin L, Liu L, QiH, Wu Y (2012). Distribution characteristics of organochlorine pesticides in surface and vertical sediments from the Zha Long Wetland, China. Environ. Sci. Pollut. Res.DOI: 10.1007/s11356-012-0762-y.
Moeller RE, Burkholder JM, Wetzel RG(1988). Significance of sedimentary phosphorous to a rooted submersed macrophyte (Najas flexilis) and its algal epiphytes. Aquat. Bot. 32: 261-281.
NOM–001–SEMARNAT–(1996). Queestablece los límitesmáximospermisibles de contaminantes en lasdescargas de aguasresiduales en aguas y bienesnacionales.DiarioOficial de la Federación, 6 deenero de 1997. México, D.F.
NOM-127-SSA1-(1994). Saludambiental, aguaparauso y consumohumano-limitespermisibles de calidad y tratamientos a quedebesometerse el aguaparasupotabilizacion.
Peterson HG, Boutin C, MartinPA, Freemark KE, Ruecker NJ, Moody MJ (1994). Aquatic phyto-toxicity of 23 pesticides applied at expected environmental concentrations. Aquat. Toxicol. 28(3/4): 275-292.
Poissant L, Koprivnjak JF (1996). Fate and atmospheric concentration of α- and γ-hexachlorocyclohexane in Québec, Canada. Environ. Sci. Technol. 30: 845-851.
Poissant L, Beauvais C, Lafrance P, Deblois C (2008). Pesticides in fluvial wetlands catchments under intensive agricultural activities. Sci. Total Environ. 404: 182-195.
Reichenberger S, Bach M, Skitschak A, Frede HG (2007). Mitigation strategies to reduce pesticide inputs into ground- and surface water and their effectiveness: A review. Sci. Total Environ. 384: 1-35.
Rose MT, Crossan AN, Kennedy IR (2008). The effect of vegetation on pesticide dissipation from ponded treatment wetlands: Quantification using a simple model. Chemosphere. 72: 999-1005.
Rose MT, Sanchez-BayoF, Crossan AN, Kennedy IR (2006). Pesticide removal from cotton farm tailwater by a pilot-scale ponded wetland. Chemosphere. 63: 1849-1858.
Sawaittayothin V, Polprasert C (2007). Nitrogen mass balance and microbial analysis of constructed wetlands treating municipal landfill leachate. Bioresour. Technol. 98: 565–570.
Sheoran AS, Sheoran V (2006). Heavy metal removal mechanism of acid mine drainage in wetlands: A critical review. Miner. Eng. 19: 105-116.
Standard methods for the examination of water and wastewaters (1985). 16th ed., American Public Health Association/America Water Works Association/Water Environment Federation, Washington DC, USA.
Stefanelli P, Muccio AD, Ferrara F, Barbini DA, Generali T, Pelosi P, Amendola G, Vanni F, Muccio SD, Ausili A (2004). Estimation of intake of organochlorine pesticides and chlorobiphenyls through edible fishes from the Italian Adriatic Sea during 1997. Food Control. 15: 27-38.
Sutton DL, Latham WGH(1996). Analysis of interstitial water during culture of Hydrilla verticillata with controlled release fertilizers. Aquat. Bot. 54: 1-9.
Tanner CC, Kadlec RH (2003). Oxygen flux implications of observed nitrogen removal rates in subsurface-flow treatment wetlands. Water Sci. Technol. 48(5): 191- 198.
Waliszewski SM, Szymczynski GA, Rogowska Z (1985). Rapid and Low Cost Method for Monitoring Determination of Selected Chlorinated Pesticides in Food Mixture. Bull. Environ. Contam. Toxicol. 34: 518-526.
Warren CS, Mackay D, Bahadur NP, Boocock DGB (2002). A suite of multisegment fugacity models describing the fate of organic contaminants in aquatic systems: application to the Rihand Reservoir, India. Water Res. 36: 4341-4355.
Warren N, Allan IJ, Carter JE, House WA, Parker A (2003). Pesticides and other micro-organic contaminants in freshwater sedimentary environments. a review. Appl. Geochem. 18: 159–194.
Zaga A, Little EE, Rabeni CF, Ellersieck MR (1998). Photoenhanced toxicity of a carbamate insecticide to early fife stage anuran amphibians. Environ. Toxicol Chem. 17(12): 2543-2553.
Zaranko TD, Griffiths RW, Kaushik NK (1997). Biomagnification of polychlorinated biphenyls through ariverine food web. Environ. Toxicol. Chem. 16: 1463-1471.