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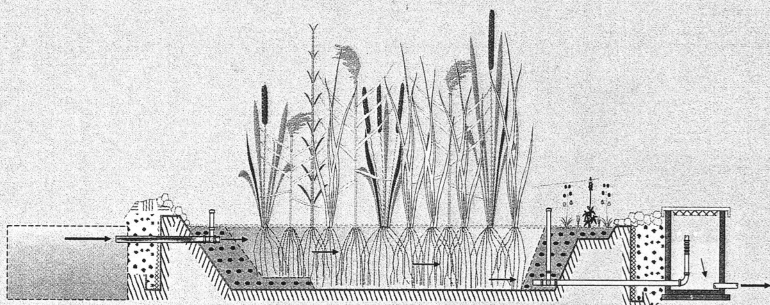
1st INTERNATIONAL SEMINAR ON

THE USE OF AQUATIC MACROPHYTES FOR WASTEWATER TREATMENT IN CONSTRUCTED WETLANDS



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SWINE WASTE TERTIARY TREATMENT FOR NITROGEN REMOVAL BY A CONSTRUCTED WETLAND

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ABSTRACT

Swine waste is an important agricultural and environmental concern because of the huge amounts of wastes generated and pollution of surface and ground waters. A study of the treatment of swine wastewater using constructed wetlands has been set up in Spain, in July 2002. The system consists of two series of three tanks, with different flow [Free Water Surface (FWS) and Subsurface Flow (SSF)] and vegetation: (cattail/common reed) and (cattail/willow). Wastewater and effluents samples has been analysed for: total and volatile suspended solids (TSS, VSS); five-day biochemical oxygen demand (BOD_5); chemical oxygen demand (COD); total Kjeldahl nitrogen (TKN); ammonium nitrogen; nitrate and nitrite nitrogen. The aim objective was to determine nitrogen removal efficiency in both series: *Typha-Phragmites* (S1) and *Typha-Salix* (S2). The system is still in stabilization phase, but data seem to suggest that S1 is better at removing N in the NH_4-N form (92%) than S2, but this is more effective at removing total N (64%) as S1 (60%).

KEYWORDS

Constructed wetland; nitrogen removal; swine wastes; tertiary treatment.

INTRODUCTION

Land application of animal wastes has been traditionally an environmentally safe practice because a few animal producers were scattered across the landscape. But nowadays, managing the waste produced by confined animal feeding operations is a major agricultural and environmental challenge. Confinement of hogs in large-scale production units generates huge amounts of animal waste. Currently, most livestock producers apply both liquid and solid waste to land for terminal treatment. Application of liquid swine wastes to land has several problems such as nuisance odour, high solids content, high nutrient concentrations and limited pumping distances (Hunt et al., 1995).

A growing environmental concern is pollution of surface and ground waters from animal wastes. High nitrogen (N) loading rates to soils and waters can be associated with intensive animal operations. Concentrations of N in excess of 10 mg L^{-1} in the nitrate (NO_3-N) form render groundwater unsuitable as drinking water for humans (Alexander, 1972). High N concentrations entering streams or lakes may contribute to eutrophication.

Traditional methods of recycling animal manures to land crop production are not always satisfactory because of the large concentration of animals. To decrease pollution while maintaining or increasing productivity, confined animal operators need practical ways to either prevent wastewater from entering surface water and groundwater, or treat the water before it leaves the farm. Operators want wastewater management systems that are affordable, reliable, and practical to build and operate.

Today, various technologies are available to treat wastewater in ways that use the natural chemical, physical and biological processes of the environment and that rely on nature's energies (Kadlec et al., 2000). One of these technologies is a constructed wetland system. Constructed wetlands are being utilized to treat a great variety of wastewaters including municipal sewage, industrial wastes and acid mine drainage (Moshiri, 1993; Hammer, 1989). The application of constructed wetlands for treatment of animal wastes is rather recent (DuBowy and Reaves, 1994; Hunt et al., 1995). They can be used in confined animal feeding operations before discharge or the application of wastewater to the land.

Results from existing treatment wetlands on farms suggest that these systems can help in several ways (DuBowy & Reaves 1994; DuBowy 1997). Treatment wetlands are a cost-effective and technically feasible approach to the management of wastewater and runoff for several reasons: they are less expensive to build, low operation and maintenance expenses, decrease labour costs associated with hauling and applying effluent, they can remove solids and nutrients, help to minimize odour problems, they are able to tolerate fluctuations in flow and to treat wastewaters with low organic load, they also provide habitat for many wetland organisms and can be built to fit harmoniously into the landscape. Constructed wetlands can be integrated into the farm in a way that benefits the operator and the neighbors (Kadlec et al., 2000).

Wetland macrophytes are the dominant structural component of most wetland treatment systems. The term macrophyte includes vascular plants that have tissues and are easily visible. A wide variety of macrophytic plants occur naturally in wetlands environments (Kadlec & Knight, 1996).

Treatment systems based on macrophytic plants consist generally of a monoculture or policulture of vascular plants ready in tanks, lagoons or little deep ditches, and with a retention time longer than conventional systems.

The type of vegetation to establish in such systems depends on its adaptability to the climate of the region, its ability to transport oxygen into the rhizosfere, its tolerance to high concentrations of polluting agent and the capacity to assimilate it.

Plant species should also be commonly found, easy to establish and manage, and resistant to insects and diseases (Ansola, 2001).

Most frequently used species in constructed wetlands are common reed (*Phragmites australis*), cattail (*Thypha* spp) and bulrush (*Scirpus* spp).

The objectives of this study are: 1) to reduce the different nitrogen forms concentration in the liquid swine wastes by a constructed wetland which allows the effluent to meet the quality objectives and provisions of the Council Directive 91/271/CEE, concerning urban waste water treatment (OJ. L-135, 1991); and 2) to evaluate the treatment performance of two different combinations of vegetation, *Typha-Phragmites* (S1) and *Typha-Salix* (S2), in the nitrogen removal from swine wastewater.

The election of these systems of depuration set against the conventional owes to its low cost, easy handling and the great integration in the environment.

METHODS

The study has been set up in Fompedraza (Valladolid, Spain), a small village of 145 inhabitants in the northwest of the Iberian Peninsula. It is a rural area, with an agricultural economy based on vine and cereal crop.

The experimental system consists of six fiberglass tanks, with a capacity of 0.5 m^3 and a surface area of 1 m^2, connected in two series of three tanks (fig.1).

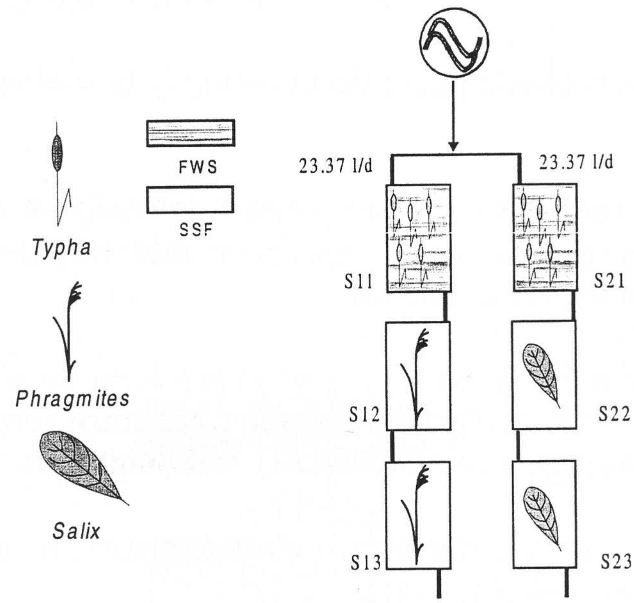

Figure- 1
Schematic representation of the wetland system, where FWS means Free Water Surface and SSF, Subsurface Flow

The first tank in each series, with a 0.30 m depth gravel-bed (6-8 mm diameter) covered with 0.25 m of the water to treat, presents a free water surface (FWS) (fig. 2). The rest of the tanks, with 0.60 m of gravel-bed depth, have subsurface flow (SSF) (fig. 3).

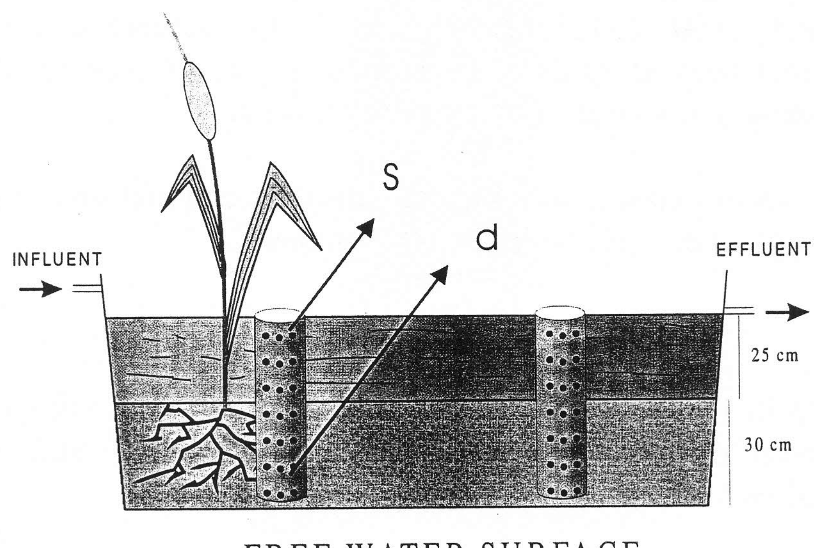

Figure- 2
Schematic representation of the Free Water Surface (FWS) tanks, where s means surface sample point, and d, depth sample point

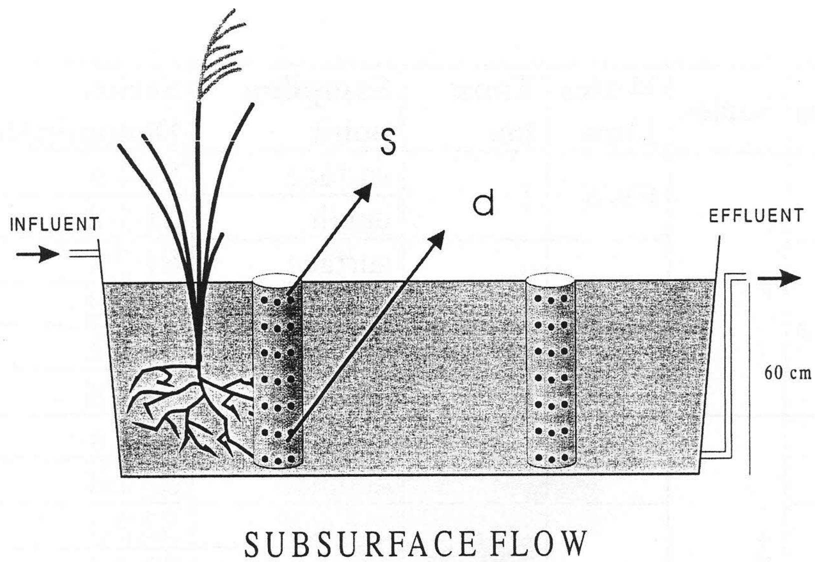

Figure- 3
Schematic representation of the Subsurface Flow (SSF) tanks, where s means surface sample point, and d, depth sample point

The FWS tanks has been planted with cattail (*Typha latifolia*), while the SSF tanks contain common reed (*Phragmites australis*) in the series 1 but willow (*Salix atrocinerea*) in the series 2 (Table 1).

Two yellow vinyl (PVC) sampling tubes, perforated at different heights to allow the water to flow through them but not the gravel, have been driven into each tank (fig. 2 and 3).

The theoretical wastewater inflow rate is 23.37 ± 0.5 l d^{-1}. Effluent is continually pumped into the system by a peristaltic pump, with a daily control.

The system began working in July 2002, so it is currently in phase of stabilisation.

Considering the theoretical retention time, estimated in 55 days, sampling has been taken every 40-50 days; hydraulic loading rate (HLR) is 0.78 cm d^{-1} (2.85 m yr^{-1}).

Table- 1
Series denomination considering vegetation, tank number and sampling point

Vegetation	Series	Water Flow	Tank no.	Sampling point	Series Denomination
Typha	1	FWS	1	surface	S1 1 s
				depth	S1 1 d
Phragmites		SSF	2	surface	S1 2 s
				depth	S1 2 d
			3	surface	S1 3 s
				depth	S1 3 d
Typha	2	FWS	1	surface	S2 1 s
				depth	S2 1 d
Salix		SSF	2	surface	S2 2 s
				depth	S2 2 d
			3	surface	S2 3 s
				depth	S2 3 d

All analyses have been performed according to *Standard Methods for the Examination of Water and Wastewater* (APHA, 1989).

All samples has been analysed for: total and volatile suspended solids (TSS, VSS); five-day biochemical oxygen demand (BOD_5); chemical oxygen demand (COD); total Kjeldahl nitrogen (TKN); ammonium nitrogen; nitrate and nitrite nitrogen. COD is measured after filtration, leading to measures of soluble COD.

During the monthly sampling, dissolved oxygen (DO), conductivity, pH, temperature, oxidation-reduction potential (ORP), have been determined by ion-selective electrodes at the wastewater sampling points. These parameters have been measured at different heights in the sampling tubes: on the surface (s) and at 0.3-0.4 m depth (d) (figures 2 and 3).

RESULTS AND DISCUSSION

Water losses to the atmosphere from a wetland occur from the water and soil (evaporation) and from the emergent portions of the plants (transpiration); the combination of the two processes is termed evapotranspiration (ET) (Kadlec and Knight, 1996). During spring, summer and fall, ET can be an important water loss in northern climates, so that flow rate must be increased. The flow rate pumped into the system varies from about 23 to 25 l d^{-1} (fig. 4), corresponding the lowest values to winter period, when it also decreases evapotranspiration (ET).

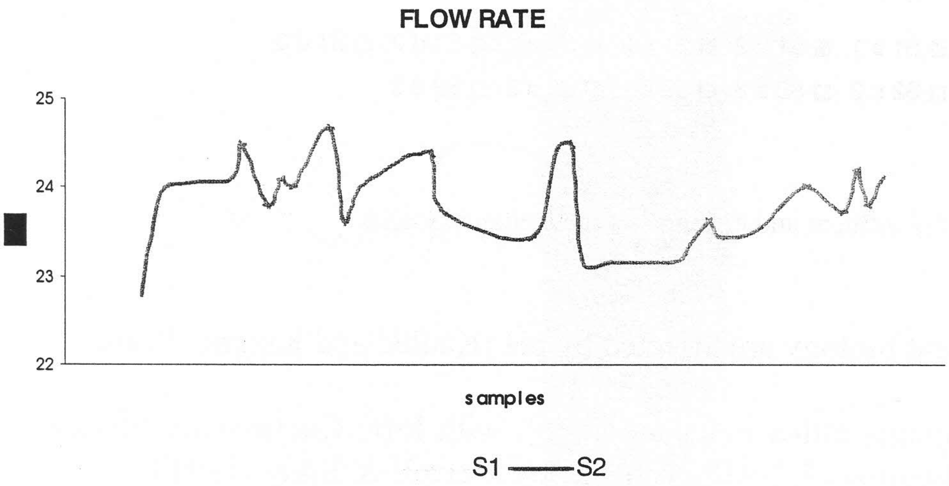

Figure- 4
Daily flow rate variation for the constructed wetland

The physical and chemical environment of a wetland affects all biological processes. In turn, many wetland biological processes modify this physical/chemical environment. Three of the most widely fluctuating and important abiotic factors are dissolved oxygen (DO), hydrogen ion concentration (pH) and temperature (Kadlec and Knight, 1996).

Wetland water temperatures can influence some pollutant removal and conversion processes (Kadlec et al., 2000). Some biochemical processes, notably the microbially mediated nitrogen processes, are temperature sensitive (Kadlec and Knight, 1996). As shown in figure 5, influent temperature varies from 24, in summer, to hardly 3ºC in winter, with a high increase at the end of February (30ºC). Effluent temperature in the two compared series also decreased in winter, but slowly starts increasing in February (fig.5).

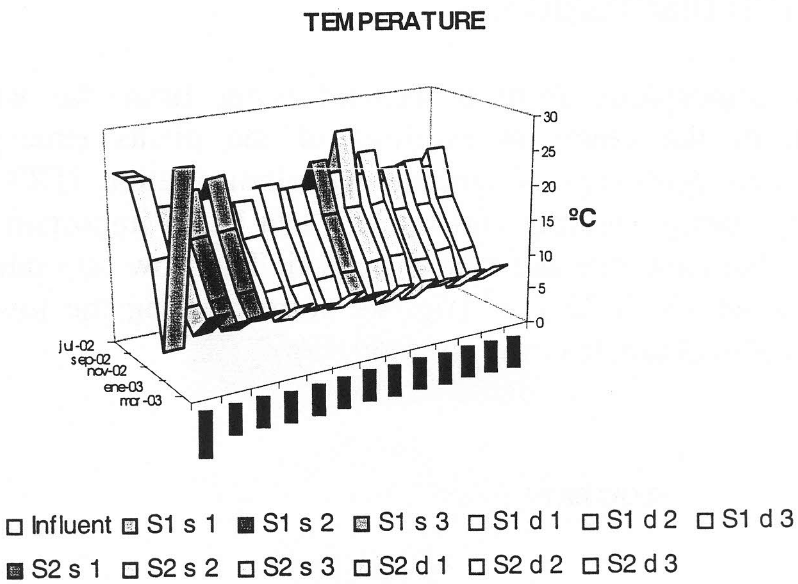

Figure- 5
Temperature measurements for the influent and effluents of the wetland system

Wetland water chemistry and biology are affected by pH (Kadlec and Knight, 1996).

The pH in the wetland remains rather constant (fig.6), with little fluctuations between the optimal levels for nitrification (7.2-9.0) proposed by Metcalf & Eddy (1991).

Oxygen, although abundant in the atmosphere, has a limited solubility in water, so it is frequently a limiting factor for the growth of plants and animals in wetlands (Kadlec and Knight, 1996). Low dissolved oxygen (DO) levels have been measured in winter, when the system has been temporarily covered by an ice layer (5-10 cm) for almost a month and a half. Monthly measures show that the average DO levels decrease along the system (fig.7), but anyway, effluent levels in both series are higher than in the influent.

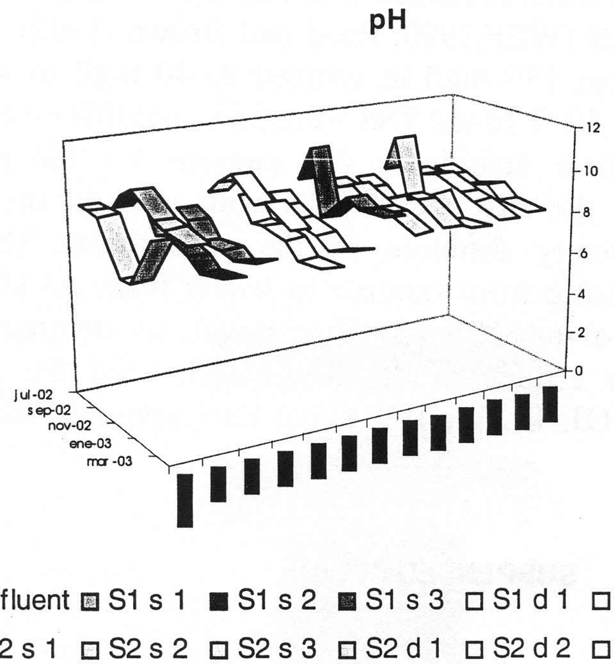

Figure- 6
Monthly pH measurements for the influent and effluents

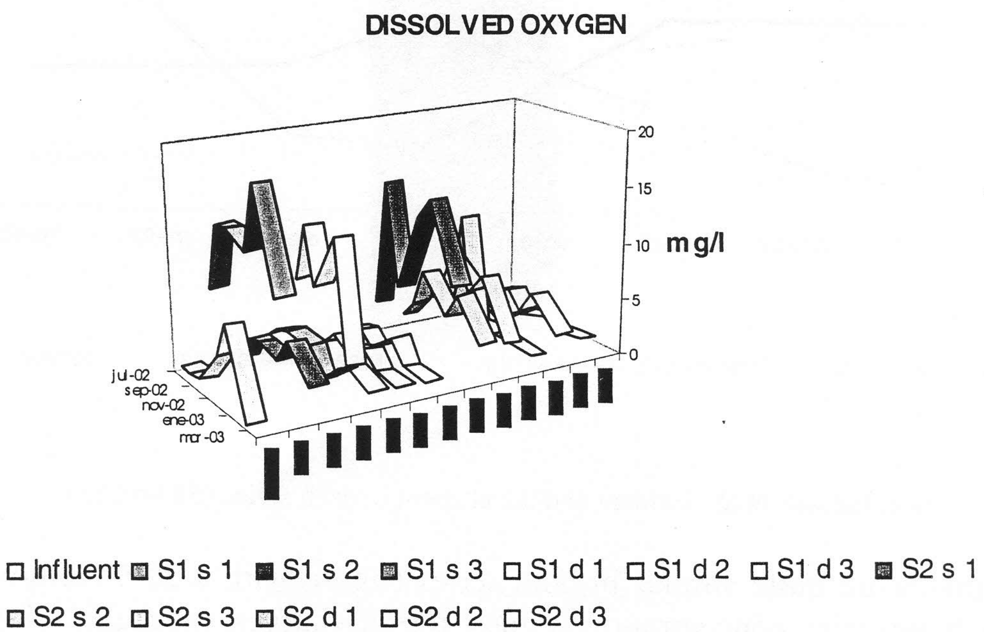

Figure- 7
Dissolved oxygen (DO) measurements for the influent and effluents

Suspended solids are one manifestation of natural wetland processes, as well as being common contaminants in feed waters (Kadlec et al., 2000). Wetlands are reported to be very efficient at removing TSS (WEF,1990; Reed and Brown, 1992). Total suspended solids in the influent vary from 130 mg/l in summer to 40 mg/l in winter. When the system began working, hardly 50 % of the TSS were SSV, but this percentage continues to grow as the system become stabilised. The patterns for the suspended solids concentrations are similar in the first series (S1), with a slight increase during the dormant season, but completely different in the second one (S2), where TSS concentration has been increasing from summer to winter time. As shown in figure 8, S2 is more effective at removing TSS (66 % removal) as compared to S1 (37 % removal). TSS levels in the effluent, in both series, exceed the requirements for discharges of the Directive (OJ. L-135, 1991), but the system is still in stabilization phase.

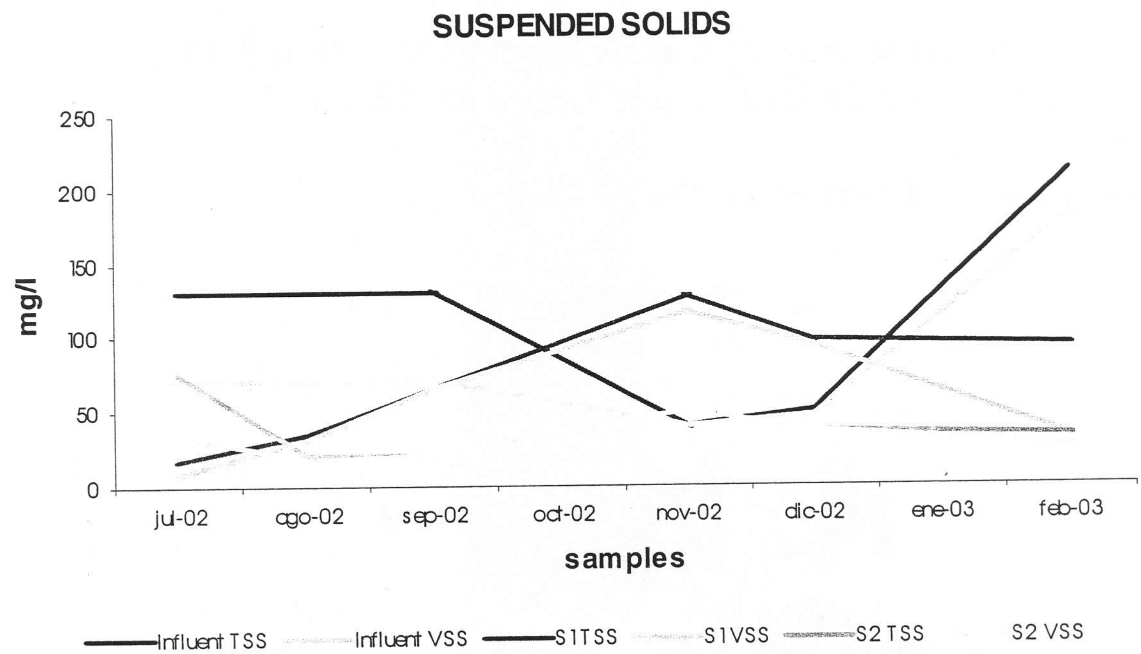

Figure- 8
TSS and VSS concentrations in the influent and the effluent of each series (S1 and S2)

BOD_5 removals are quite similar in both series (fig.9), with a 28 % removal. Effluent levels are below the concentration values for discharges (25mg/l) allowed by the European Union (OJ. L-135, 1991).

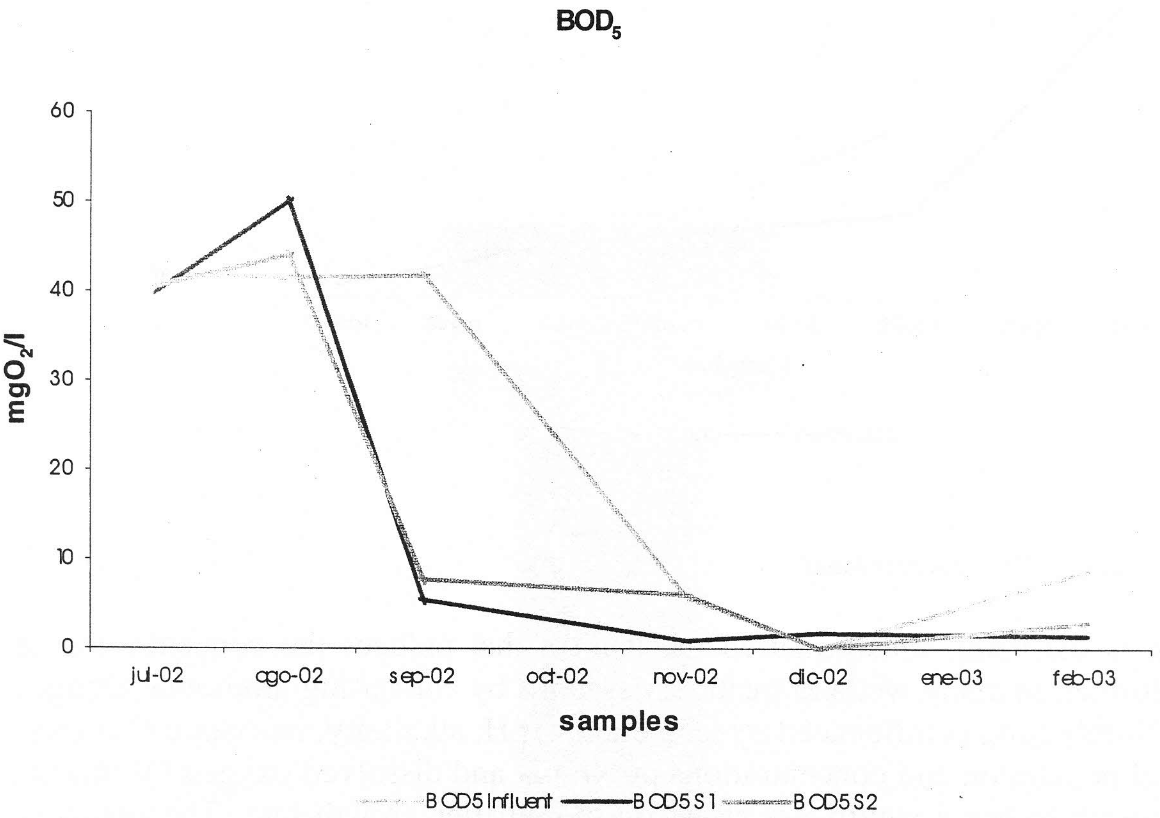

Figure- 9
Influent and effluents BOD_5 concentrations

Figure 10 reports COD data; effluent mean levels in series 1 and series 2 are 64 and 82 mg/l, respectively; none of them exceed the discharge requirements of the European Union (OJ. L-135, 1991).

In municipal wastewaters, the ratio BOD_5 to COD is typically 0.4-0.8 (Metcalf and Eddy, 1991). Industrial wastewaters may have lower ratios. In this study, influent ratio is 0.45, whereas in the effluent it seldom reaches 0.2.

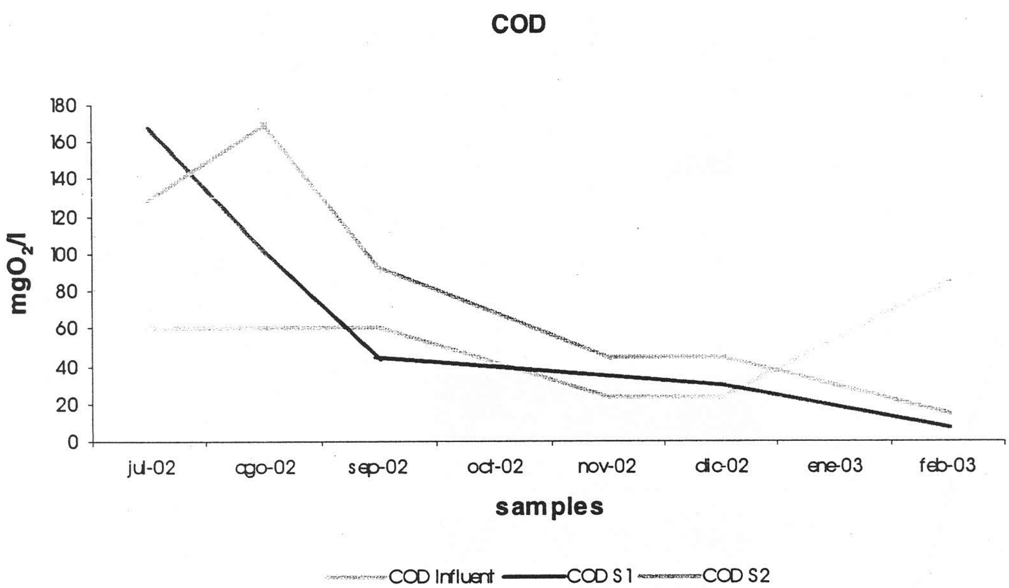

Figure- 10
Influent and effluents COD concentrations

Nitrification is the main transformation mechanism that reduces the concentration of ammonia nitrogen in many wetland treatment systems by converting ammonia nitrogen to nitrates. Nitrification is influenced by temperature, pH, alkalinity, inorganic C source, the microbial population and concentrations of NH_4-N and dissolved oxygen (Vymazal, 1995). Temperature has a significant effect on the nitrifier growth rate. The optimum temperature range for nitrification in pure bacterial cultures is from 25 to 35°C. Temperatures lower than 15°C are reported to lower nitrifiers growth rates drastically (Reddy and Patrick, 1984). At water temperatures above 30°C, the nitrification rate begins to decline sharply.

An expression that can be used to model the disappearance of ammonia nitrogen in wetland treatment systems is the area-based, first-order model. The areal rate constant for ammonium reduction is determined by the following equation:

$$\ln\left[\frac{Co - C*}{Ci - C*}\right] = -\frac{k}{q}$$

where Ci is inlet NH_4-N concentration (mg/l), Co is outlet NH_4-N concentration (mg/l), C* is background NH_4-N concentration (mg/l), k is first order areal rate constant (m/yr) and q is Hydraulic loading rate (m/yr) (Kadlec and Knight, 1996).

The rate constant varies from season to season. The values of rate constants for this model are shown in table 2.

Analyses of wastewater samples collected monthly during the study showed that the average total N concentration in the influent was 112 mg/l, with most of the N in the NH_4-N form (85 %). The same patterns are observed in the effluent, with 85 % of the total N in the NH_4-N form in S1 and 96 % in S2. Concentrations of NH_4-N and total N are shown in figures 11 and 12.

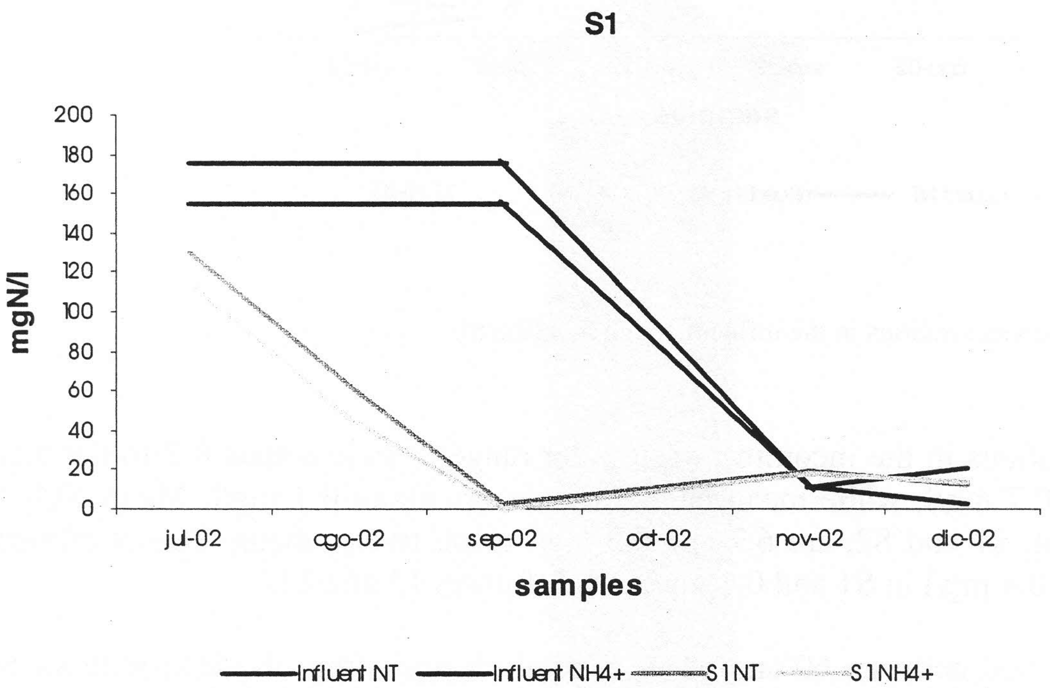

Figure- 11
Total N and NH_4-N concentrations in the influent and in S1 effluent

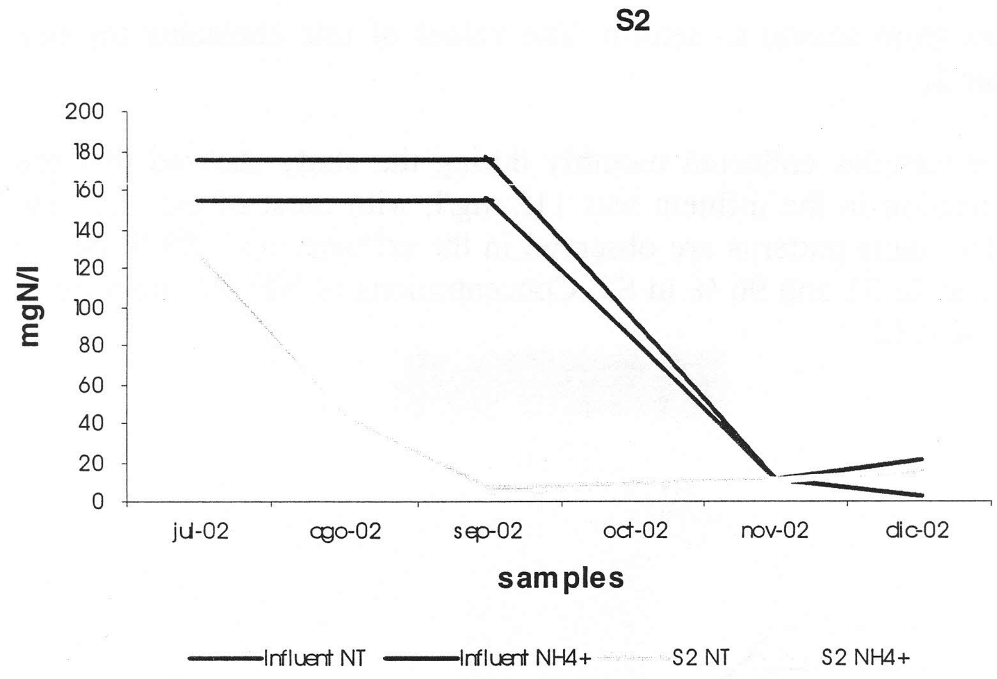

Figure- 12
Total N and NH_4-N concentrations in the influent and in S2 effluent

Nitrate concentrations in the incoming wastewater range from less than 0.2 to 0.9 mg/l with a mean of 0.7 mg/l, while mean NO_2-N values are about 0.1 mg/l. Mean NO_3-N values in effluent, S1 and S2, are 6.7 and 0.3 mg/l respectively; mean NO_2-N effluent values are about 0.8 mg/l in S1 and 0.1 mg/l in S2 (figures 13 and 14).

The system removed between 60 and 64 % of total nitrogen from the incoming swine waste; S2 is more effective at removing total N as compared to S1, whereas S1 is better at removing N in the NH_4-N form (92 %) than S2 (88 %).

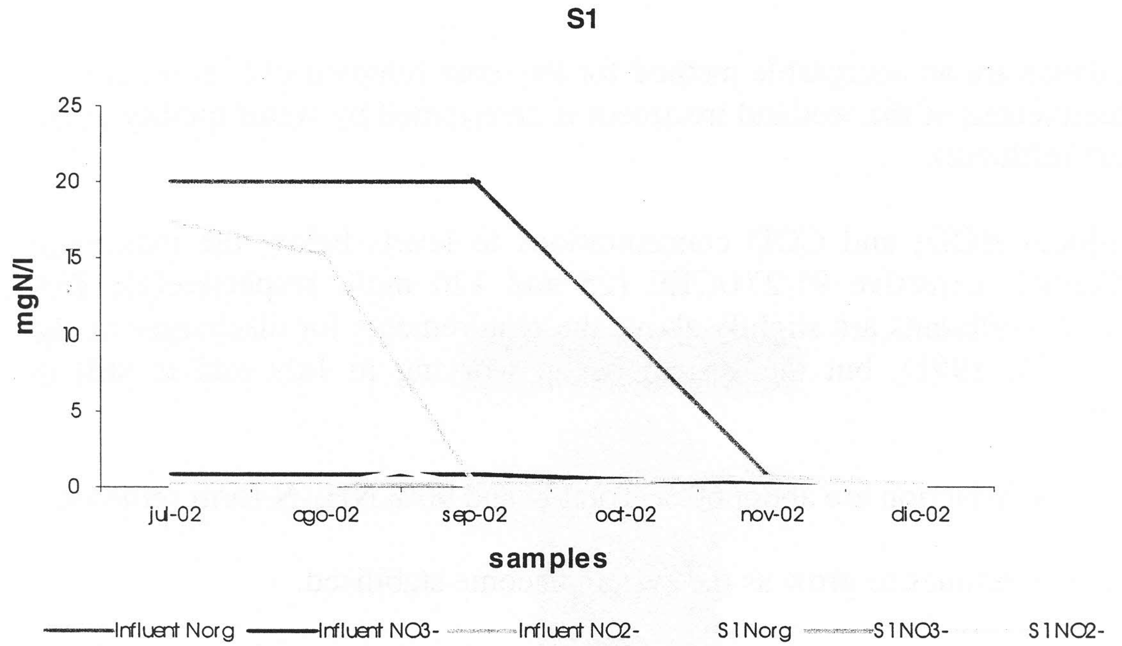

Figure- 13
Influent and S1 effluent concentrations of organic N, NO_3-N and NO_2-N

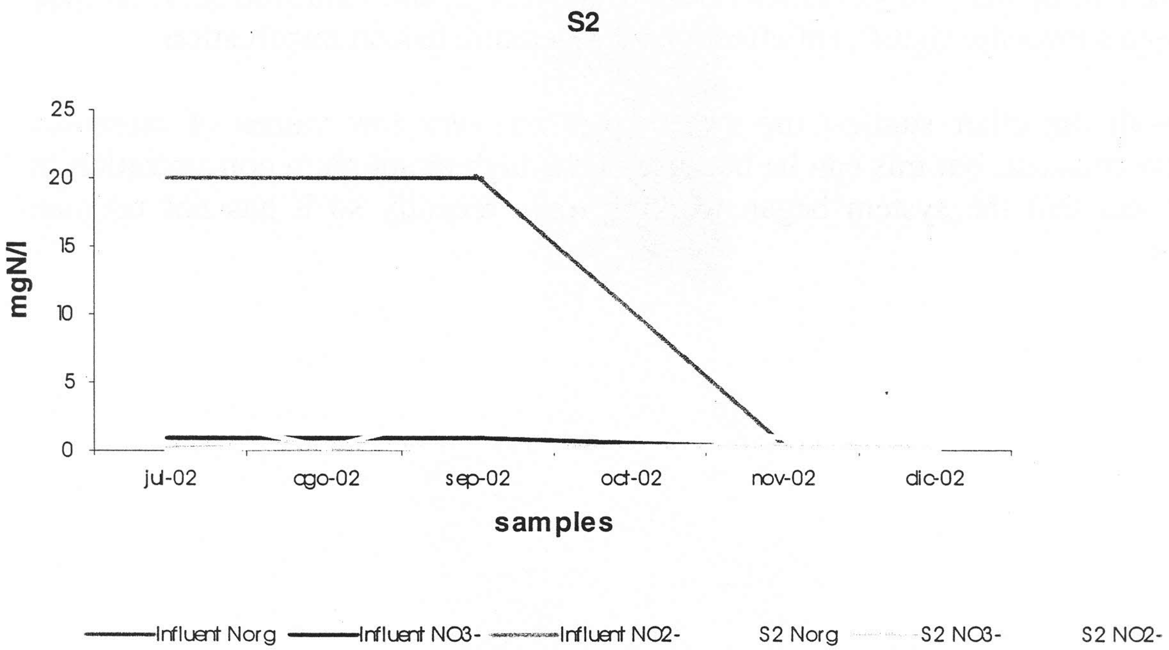

Figure- 14
Influent and S2 effluent concentrations of organic N, NO_3-N and NO_2-N

CONCLUSIONS

Constructed wetlands are an acceptable method for the mass removal of N from animal wastes. The effectiveness of the wetland treatment is determined by water quality at the end of the system (effluent).

The wetland reduces BOD_5 and COD concentrations to levels below the maximum limits of the Council Directive 91/271/CEE (25 and 120 mg/l, respectively); TSS concentrations in the effluents are slightly above the requirements for discharges of the Directive (OJ. L-135, 1991), but the system began working in July and is still in stabilization phase.

Specific values for N reduction are about 60 % Total N and 90% NH_4-N form removal.

But this percentage continues to grow as the system become stabilised.

Series 2 is more effective at removing TSS (66 %) than Series 1 (37 %), and also at removing total N (64%); but is in S1 where the greatest NH_4-N removals have been managed (92%).

The area-based ammonia disappearance rate constant has greater values in summer than in winter; what shows the significant effect that temperature has on nitrification.

Compared with the other studies, the system presents very low values of ammonia reduction rate constant, but this can be because of the high ammonium concentration in the influent and that the system began working quite recently so it has not become stabilized yet.

Table- 2
Example Area-Based, First-Order Rate Constants for Ammonium Reduction for Treatment Wetlands (Adapted from Kadlec and Knight, 1996)

Location	Wetland	Type	HLR (cm/d)	NH_4-N In (mg/l)	NH_4-N Out (mg/l)	Apparent k (m/yr)	T (°C)	Data Ref.
Surface-flow marshes								
Listowel, Ontario	1	SF con marsh	2.67	7.40	5.30	3.3	8.0[a]	Herskowitz, 1986
	2	SF con marsh	2.84	7.40	5.66	2.8	8.0[a]	Herskowitz, 1986
	3	SF con marsh	1.92	7.40	4.37	3.7	7.84	Herskowitz, 1986
	4	SF con marsh	1.95	8.58	6.43	2.1	8.02	Herskowitz, 1986
	5	SF con marsh	2.60	8.58	8.45	0.1	8.0[a]	Herskowitz, 1986
Houghton Lake, MI		SF nat marsh				22.1	16[a]	Kadlec, 1979-1994
Benton, KY	1	SF con marsh	1.71	5.04	7.89	-2.8	16.14	Choate et al.,1990a
	2	SF con marsh	1.71	5.04	6.43	-1.5	17.75	Choate et al.,1990a
Gustine, CA	5 Wetlands	SF con marsh	3.20	17.65	19.97	-1.4	17[a]	Walker and Walker, 1990
Richmond, NSW	1	SF open water	6.41	35.20	17.50	15.4	19.70	Bavor et al., 1988
Floating/submergent aquatics								
New Zealand	2 Wetlands	SF con FAP	5.52	60.00	37.00	9.7	15	van Oostrom, 1994
Richmon, NSW	4	SF con SAP	7.34	35.20	24.00	10.3	17.16	Bavor et al., 1988
Hyacinth Pilot	1	SF con FAP	40.80	27.42	8.25	178.9	ca. 25	Weber and Tchobanoglous, 1986
Hyacinth Pilot	1	SF con FAP	44.00	6.45	1.65	218.9	ca. 25	Weber and Tchobanoglous, 1986
Soil-based reed beds								
Denmark	44 Wetlands	Soil-based *Phragmites*	4.53	21.00	14.10	6.6	6.5[a]	Schierup et al., 1990b

Table- 2
Continued

Location	Wetland	Type	HLR (cm/d)	NH_4-N In (mg/l)	NH_4-N Out (mg/l)	Apparent k (m/yr)	T (°C)	Data Ref.
Subsurface-Flow reed beds								
U.K.	11 Wetlands	SSF *Phragmites*	18.30	4.60	2.80	33.2	9.5[a]	Green and Upton, 1993
Gravel beds								
Benton, KY	3	SSF bulrush	5.22	4.84	8.62	-11.0	16.74	Choate et al.,1990a
Hardin, KY	1	SSF cattail	8.81	7.95	9.32	-5.1	16.15	Choate et al.,1990a
	2	SSF bulrush	7.34	6.86	5.92	3.9	13.36	Choate et al.,1990a
North America	8 Wetlands	SSF	4.66	5.70	3.20	9.8		NADB, 1993
Richmond, NSW	2	Gravel only	3.83	35.20	19.20	8.5	18.57	Bavor et al., 1988
	3	SSF bulrush	5.08	35.20	19.40	11.0	18.20	Bavor et al., 1988
	5	SSF cattail	4.61	35.20	19.80	10.6	18.37	Bavor et al., 1988
Free water surface/ Subsurface flow								
Spain	1	FWS cattail/ SSF common reed	0.78	154.30	79.85	1.87	18.9	Cordero et al., 2003
		FWS cattail/ SSF common reed	0.78	6.70	15.80	-2.44	7	Cordero et al., 2003
	1	FWS cattail/ SSF willow	0.78	154.30	81.85	1.80	18.9	Cordero et al., 2003
		FWS cattail/ SSF willow	0.78	6.70	12.75	-1.83	7	Cordero et al., 2003

[a] Estimated as mean daily air temperature.
Note: The value of C* is taken as 0.0.

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