Université de Montréal

Effet de l’espèce de plante en marais filtrants artificiels selon la saison, le type de marais filtrant et la nature des polluants

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Cette thèse intitulée :

Effet de l'espèce de plante en marais filtrants artificiels selon la saison, le type de marais filtrant et la nature des polluants

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Résumé

Les marais filtrants artificiels sont communément utilisés pour le traitement des eaux usées municipales, mais ils peuvent aussi traiter une large gamme d'effluents à l'aide de différents types de marais filtrant et ce sous une variété de climats. Les plantes jouent un rôle central dans l'épuration des eaux, mais on peut se demander si ce rôle est le même sous toutes les conditions et ce pour les différentes espèces de plantes. Dans le cadre de cette étude, j'ai évalué l'influence de la présence ainsi que de l'espèce de plantes selon des conditions extrêmement différentes : 1) traitement de rejet de serre hydroponique par un marais filtrant à flux horizontal en conditions hivernales, 2) traitement des boues par un marais à flux vertical en conditions estivales.

Le traitement des rejets hydroponiques est problématique puisque contrairement aux effluents municipaux, ils sont fortement concentrés en nitrate, mais ne contiennent aucun carbone organique nécessaire pour la dénitrification. Toutefois, les plantes peuvent fournir du carbone par leurs exsudats racinaires, dont la quantité varie selon l'espèce. J'ai donc testé l'effet de trois espèces de plante (*Typha* sp., *Phragmites australis* et *Phalaris arundinacea*) ainsi que l'ajout de carbone sur le traitement en conditions hivernales. Les résultats montrent que les marais plantés sont plus efficaces que les marais témoins non plantés, même en hiver alors que les plantes sont en dormance. Les marais plantés de *Phalaris* avaient une efficacité supérieure. Toutefois, l'apport de carbone par les plantes était insuffisant pour une épuration adéquate des nitrates et l'ajout d'une source de carbone externe s'est avéré nécessaire.

Les boues sont des effluents qui peuvent être jusqu'à 100 fois plus concentrées que les eaux usées municipales. Les lits de séchages de boues plantés de macrophytes (Lisam) sont des marais à flux vertical qui sont utilisés pour déshydrater et minéraliser les boues. La qualité de l'eau en sortie des Lisam a généralement peu d'importance, puisque le rejet est envoyé les stations de traitement des eaux usées. Toutefois, lorsque les boues sont produites loin des stations de traitement, les Lisam doivent non seulement traiter la boue, mais aussi limiter le rejet de polluants. Pour ce volet, j'ai évalué l'influence des différentes espèces de plante (*Phragmites australis*, *Typha angustifolia* et *Scirpus fluviatilis*) sur l'épuration de l'eau ainsi que sur la déshydratation et la minéralisation de la boue. Les systèmes plantés sont meilleurs avec significativement moins de polluants en sortie des Lisam plantés de *Phragmites*, suivi par *Typha* et *Scirpus*. La boue accumulée en surface des Lisam plantés de *Phragmites* était plus sèche et minéralisée comparativement aux autres espèces. L'influence des plantes s'explique par la séquestration de polluants dans les végétaux et par l'effet positif de la rhizosphère sur la dégradation des polluants. La filtration et l'évapotranspiration ont aussi joué un rôle majeur dans l'épuration.

On peut donc conclure que les plantes jouent un rôle tout aussi important dans l'épuration des eaux, et que le choix de l'espèce est fondamental, même sous des conditions très différentes de celles rencontrées lors du traitement d'eaux usées municipales.

Mots clés

Marais filtrant artificiel, lit de séchage de boue planté de macrophytes, traitement des eaux, rejet de serre hydroponique, boue piscicole, influence de l'espèce de plante, évapotranspiration, espèce invasive

Abstract

Constructed wetlands (CWs) are commonly used to treat municipal wastewater, but they can also handle a wide range of effluents by using different types of CWs and under a variety of climatic conditions. Plants play a central role in CWs, but it is still unknown if the presence of plants or the choice of the species have the same influence under different conditions. In this study, I evaluated the influence of the presence and the species of plants under two very different conditions: 1) treatment of hydroponic wastewater by a horizontal flow CW in winter conditions, 2) sludge treatment by a vertical flow CW in summer conditions.

Treatment of hydroponics wastewater, which is composed fertiliser, is problematic since, unlike municipal effluents, they are highly concentrated in nitrate, but contain no carbon required for denitrification. However, plants may provide carbon by their root exudates, whose quantity varies with the plant species. Therefore, I tested the effect of three plant species (*Typha* sp., *Phragmites australis* and *Phalaris arundinacea*), and the addition of carbon on the treatment of hydroponics wastewater under winter conditions. The results show that planted CWs were more effective than the unplanted control, even under winter condition when the plants are dormant. CWs planted with *Phalaris* were the most efficient in treating nitrate pollution compared to the other plant species. However, the carbon from the plants exudates was insufficient for satisfactory treatment and thus an external source of carbon was necessary for complete nitrate removal.

The sludge is an effluent that can be up to 100 times more concentrated than municipal wastewater. It can be treated in sludge treatment wetlands (STWs), which is a type of vertical flow CW specialised in the dewatering and mineralisation of the sludge. The water quality at the outlet of STWs is usually not an issue, since the water is sent back to the wastewater treatment plant (WWTP). However, in cases where sludge's are produced in remote areas far from WWTP, such as fish farm, STWs must not only treat the sludge, but also limit pollutant discharge into the environment. The purpose of this segment of my thesis was to evaluate the influence of different plant species (*Phragmites australis*, *Typha angustifolia* and *Scirpus fluviatilis*) on water quality at the outlet of STWs as well as on the dehydration and mineralization of the fish farm sludge. Higher water quality were found at the outlet of STWs planted with *Phragmites*, followed by *Typha* and *Scirpus*. STWs planted with *Phragmites* had also dryer and more mineralised sludge compared to other species. The influence of plants was due to the sequestration of pollutants in plants tissues and by the positive effect of the rhizosphere on the degradation of pollutants. Filtration and evapotranspiration also played a major role in the in the sludge dewatering and pollution removal.

Therefore, we can conclude that plants play a significant in water treatment, and that the choice of plant species in fundamental, even under conditions very different from those usually encountered during the treatment of municipal wastewater.

Keywords

Constructed wetlands, sludge treatment wetland, wastewater treatment, hydroponics wastewater, fish farm sludge, influence of plant species, evapotranspiration, invasive species

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Liste des abréviations et des acronymes

Abréviations et acronymes*	Définitions françaises	Définitions anglaises
Ø	Diamètre	Diameter
C/N	Ratio carbone sur azote	Carbon to nitrogen ratio
(CW)	Marais filtrant artificiel	Constructed wetland
DBO_5 (BOD_5)	Demande biochimique en oxygène sur 5 jours	Five day biochemical oxygen demand
DCO (COD)	Demande chimique en oxygène	Chemical oxygen demand
ET	Evapotranspiration	Evapotranspiration
HRT	Temps de rétention hydraulique	Hydraulic retention time
HSSF	Sous surfacique à flux horizontal	Horizontal subsurface flow
Lisam (STW)	Lit de séchage planté de macrophyte	Sludge treatment wetland
MS (TS)	Matière sèche	Total solids
MES (TSS)	Matière en suspension	Total suspended solids
MV (TVS)	Matière volatile	Total volatile solids
MVES (TVSS)	Matière volatile en suspension	Total volatile suspended solids
NH_4-N	Azote sous forme d’ammoniaque	Ammonia nitrogen
NO_x-N	Azote sous forme de nitrite et nitrate	Nitrite and nitrate nitrogen
NO_3-N	Azote sous forme de nitrate	Nitrate nitrogen
N-tot (TN)	Azote total	Total nitrogen
(TKN)	Azote total Kjeldahl	Total Kjeldahl nitrogen
P-tot (TP)	Phosphore total	Total phosphorus
PO_4-P	Phosphore sous forme de phosphate	Phosphate phosphorus
REDOX	Potentiel de réduction et d’oxydation	Reduction oxidation potential
(TC)	Carbone total	Total carbon
(TOC)	Carbone organique total	Total organic carbon

* Forme anglaise mise entre parenthèses

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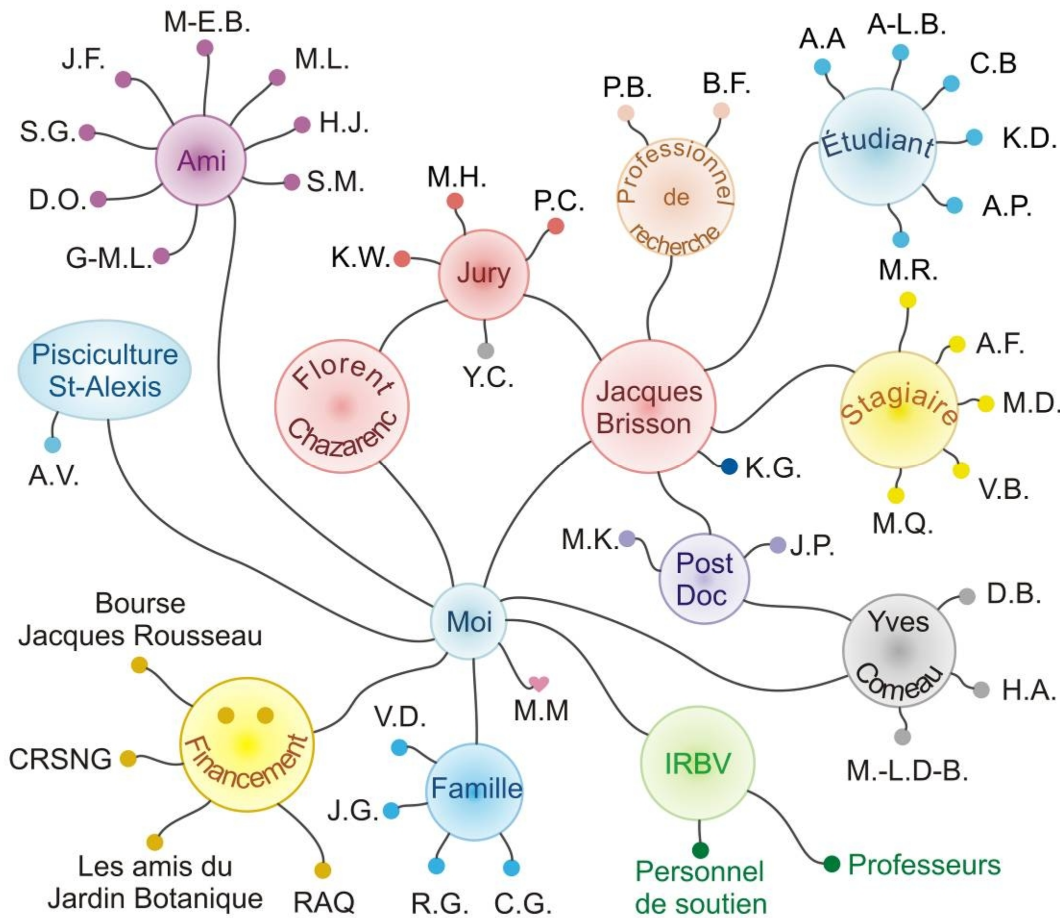

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Vous qui me lisez…

Chapitre 1 : Introduction générale

1.1 Mise en contexte de l'étude

Inspirés des marais en milieu naturel, les marais filtrants artificiels sont des écosystèmes récréés par l'être humain afin de traiter les eaux usées (Kadlec et Wallace, 2009). Mettant à profit les interactions entre les plantes, les microorganismes et le sol, les marais artificiels favorisent l'enlèvement des polluants par de multiples procédés biologiques et physicochimiques (Stottmeister *et al.*, 2003). De plus, comparativement aux procédés traditionnels, les marais filtrants artificiels nécessitent peu d'énergie, s'intègrent bien dans le paysage et ont un faible coût de construction et d'opération (Brix, 1994; Werker *et al.*, 2002).

Communément utilisés pour le traitement des eaux usées domestiques et municipales, les marais filtrants peuvent aussi traiter une large gamme d'effluents, dont la composition et la concentration en polluants peuvent être extrêmement variées (Table 1).

Tableau 1.1 Caractéristiques de certains effluents traités par marais filtrant artificiel

Polluants (mg/L)	**Rejet de serre hydroponique**	**Eau de ruissellement**	**Eau usée municipale**	**Boue activée décantée**	**Boue de fosse septique**
MES	6	101	220	9 493	35 185
MVES	-	-	165	7 594	23 926
DCO	25	73	500	11 840	47 051
DBO_5	-	10	220	-	-
N-tot	-	1,9	40	739	1 555
NH_4-N	9	-	25	15	302
NO_x-N	277	0,8	0	0	-
P-tot	69	0,4	8	229	699
PO_4-P	65		5	38	46
Références	Prystay et Lo 2001; Koide *et al.,* 2004; Grasselly *et al.,* 2005; Park *et al.*, 2008	Burton et Pitt, 2002	Kadlec et Wallace, 2009	Troesch *et al.,* 2009a	Troesch *et al.,* 2009b

Notamment, certains effluents ont une forte concentration en polluants inorganiques et presqu'aucune matière en suspension (rejet hydroponique), tandis que d'autres sont très riches en matière organique et ont une très forte teneur en solides (les boues).

Afin de traiter ces eaux usées, dont la composition peut être extrêmement différente, plusieurs types de marais filtrants ont été conçus avec des architectures et des modes d'alimentation qui favorisent certains processus d'épuration. Par exemple, les marais sous surfaciques à flux horizontal sont des systèmes saturés en eaux (Figure 1.1), dont les conditions sont généralement anaérobies avec certaines zones d'anoxies et d'aérobies près du système racinaire des plantes (Kadlec et Wallace, 2009). Ce type de marais est préconisé lorsque l'effluent est chargé en nitrate, puisque les conditions d'anaérobies du système favorisent le processus de dénitrification et donc la transformation des nitrates en azote gazeux (Vymazal, 2005).

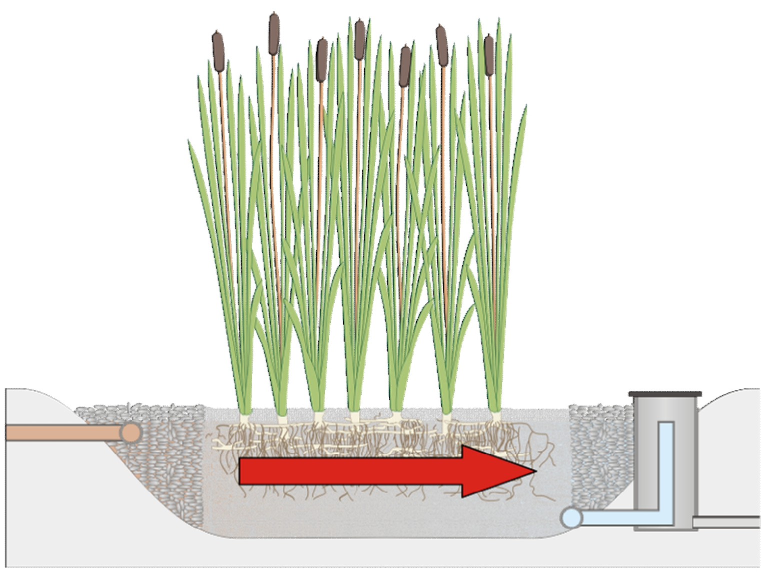


Figure 1.1 : Vue en coupe d'un marais sous surfacique à flux horizontal

Par contre, ce type de marais a des limitations pour le traitement de certains polluants. Par exemple, le traitement d'un effluent ayant une forte concentration en ammoniaque peut être problématique, car la quantité d'oxygène fournie par les plantes est souvent insuffisante pour permettre la nitrification (IWA, 2000). De plus, ce type de marais n'est pas conçu pour le traitement de fortes teneurs en matière en suspension, puisque l'effluent est alimenté sous la surface du marais, ce qui le rend vulnérable au colmatage (Vymazal, 2005).

Le marais sous surfacique à flux vertical est un autre type de marais filtrant, dont les processus d'épuration complémentent les limitations des marais à flux horizontal. Ce marais est constitué d'un lit de sable ou gravier, dont l'apport en effluent se fait à la surface du lit et percole verticalement au travers de la matrice plantée (Figure1.2). Puisque le système n'est pas saturé d'eau, l'effluent est oxygéné par son passage dans le gravier, ce qui favorise la biodégradation de certains polluants, dont l'ammoniaque et la matière organique (IWA, 2000). De plus, le marais à flux vertical peut traiter de fortes teneurs en matières en suspension, dont des eaux usées brutes (Molle *et al.*, 2005). Il peut même traiter des résidus de boue activée, grâce à une variante des marais verticaux nommée le « lit de séchage de boue planté de macrophytes » (Uggetti *et al.*, 2010). Toutefois, l'épuration peut être limitée par l'absence de zones anaérobies nécessaires pour certains processus de biodégradation, dont la dénitrification.

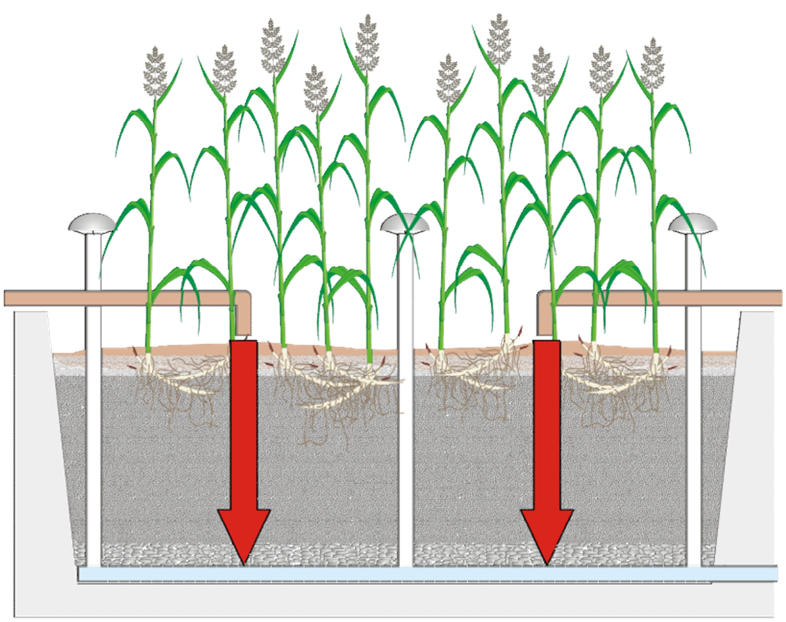


Figure 1.2 : Vue en coupe d'un marais sous surfacique à flux vertical

La présence de plantes dans les marais filtrants, en plus d'être esthétique, joue un rôle central dans le traitement des eaux usées. Il a été montré que les marais plantés ont généralement une meilleure performance épuratoire que les témoins non plantés (Tanner, 2001; Vymazal, 2011). Ceci s'expliquerait entre autres par la séquestration d'une fraction des polluants (N, P, métaux) dans les tissus végétaux, ainsi que par la prévention du colmatage des systèmes grâce au mouvement des tiges et racines (Brix, 1997). Toutefois, le rôle le plus important des plantes serait de créer un environnement favorable au développement des microorganismes responsables de la biodégradation des polluants (Hatano *et al.*, 1993). La rhizosphère favoriserait l'établissement de microorganismes en leurs procurant une surface pour se développer, une micro-zone aérobie grâce à l'oxygène libéré par les racines, ainsi qu'une source de carbone organique via les exsudats racinaires (Zhu et Sikora, 1995; Münch *et al.*, 2005; Gagnon *et al.*, 2007). De plus, la présence de plantes favoriserait l'évapotranspiration, ce qui augmenterait le temps de rétention hydraulique, permettant ainsi un plus long contact entre les polluants et les microorganismes de la rhizosphère (Faulwetter *et al.*, 2009). L'influence bénéfique des plantes sur l'épuration serait donc due à une combinaison de plusieurs facteurs, dont ceux-ci pourraient varier selon les espèces de plantes utilisées. En effet, une revue de la littérature par Brisson et Chazarenc (2009) a montré qu'en générale l'efficacité de traitement variait selon l'espèce de plante et ce, pour au moins un des polluants mesurés dans chaque étude. Toutefois, les raisons qui expliquent ces différences entre les espèces restent encore peu connues.

Le climat peut aussi être un facteur influençant le traitement des eaux en marais filtrant, puisque l'épuration est en grande partie basée sur des processus biologiques. Les marais sous surfaciques à flux horizontal ont été montrés comme généralement efficaces en condition hivernale, puisque l'eau s'écoule sous la surface du marais et est donc en partie isolée contre les basses températures de l'air (Werker *et al.*, 2002). Par contre, la sénescence des végétaux à l'automne pourrait avoir un impact sur l'apport en oxygène et les basses températures pourrait ralentir l'activité microbienne (Ouellet-Plamondon *et al.*, 2006). En effet, une variation saisonnière a été observée en marais filtrants avec une efficacité supérieure en été et en automne comparativement à l'hiver et au printemps (Werker *et al.*, 2002). La présence de plantes et particulièrement de certaines espèces de plante serait bénéfique en condition hivernale, puisque l'épuration serait moins susceptible de varier avec la température comparativement au marais non planté (Riley *et al.*, 2005; Ouellet-Plamondon *et al.*, 2006; Taylor *et al.*, 2011).

Les marais filtrants artificiels sont des systèmes d'épuration extrêmement versatiles, puisqu'ils peuvent traiter une très large gamme de polluants, à l'aide de différents types de marais filtrant, plantés de différentes espèces et ce sous une variété de climats. Les plantes ont un rôle central à jouer dans l'épuration des eaux, mais on peut se demander si ce rôle est le même sous toutes ces conditions et quel est l'effet spécifique de l'espèce de plante. La grande majorité des études ayant évalué le rôle des végétaux ont été réalisées dans des conditions de traitement d'eaux usées domestiques, soit des conditions qu'on pourrait qualifier d'intermédiaires considérant l'éventail des effluents traités par marais filtrants. On connaît beaucoup moins ce rôle des végétaux dans les conditions d'effluents plus extrêmes. Dans le cadre de la présente étude, je vais évaluer l'influence de la présence ainsi que de l'espèce de plantes sur le traitement en marais filtrant artificiel selon des conditions inhabituelles, soit, à une extrémité, des rejets de serre hydroponique traités par un marais filtrant à flux horizontal, en condition hivernale et, à l'autre extrémité, des boues piscicoles traitées par un marais à flux vertical en conditions estivales (tableau 1.2). Pour la suite de la présente introduction, je vais détailler la problématique associée à chacune de ces conditions, formuler mes objectifs et hypothèses de recherche, et présenter brièvement l'approche méthodologique de mon étude.

Tableau 1.2 : Conditions expérimentales

	Rejet de serre hydroponique	**Boue piscicole**
Caractéristiques	Polluant minéral : Forte concentration de NO_3 et PO_4, faible en DCO et MES	Polluants organiques : Forte concentration de MES, MVES, DCO et NH_4, faible en NO_3
Type de marais	Marais filtrant sous surfacique à flux horizontaux (anaérobie)	Marais filtrant sous surfacique à flux vertical (aérobie)
Saison	Simulation de condition hivernale	Estivale

1.2 Traitement des rejets hydroponique par marais filtrant

La culture hydroponique de légumes en serre nécessite une grande quantité d'eau et de fertilisant chimique afin d'optimiser le rendement. Il a été estimé qu'environ 25 à 45% du volume de fertilisant est ajouté en excès et conséquemment une quantité substantielle de polluant est rejetée par ce type d'agriculture (Prystay et Lo, 2001; Grasselly *et al.*, 2005). Ceci vient du fait que la concentration en nutriments de la solution hydroponique devient débalancée avec le temps et la solution doit être changée fréquemment afin d'éviter une accumulation de sels (Koide et Satta, 2004; Prystay et Lo, 2001). Non traité, ce rejet engendre une source de pollution ponctuelle, dont la forte concentration en azote et en phosphore peut causer l'eutrophisation des cours d'eau et des lacs (Ansari *et al.*, 2010) ainsi que la contamination des eaux souterraines (Almasri, 2007).

Il existe plusieurs techniques pour le traitement des rejets de serre hydroponique, dont l'utilisation de membranes de filtration à échange ionique ainsi que l'osmose inverse (Koide et Satta, 2004). Toutefois ces techniques sont énergivores et ont un coût élevé d'installation et d'utilisation. Les marais filtrants artificiels ont été proposés comme alternative extensive et économique au traitement des rejets de serre hydroponique (Grasselly *et al.*, 2005). Les eaux usées de serre hydroponique se distinguent des eaux usées domestiques (Table 1) par le fait qu'elles sont composées exclusivement de polluants sous forme minérale, avec une très forte concentration de nitrate et un niveau élevé de phosphate et d'ammoniaque, mais aucune matière organique (Prystay et Lo, 2001; Park *et al.*, 2008). Les marais filtrants sous surfaciques à flux horizontal offrent l'avantage d'avoir des conditions anaérobies favorables aux processus de dénitrification. Toutefois, l'absence de carbone organique dans l'effluent pourrait avoir un impact important sur l'épuration des eaux, car les microorganismes responsables de la dénitrification ont besoin d'une source carbonée pour transformer les nitrates en azote gazeux (N_2) (Faulwetter *et al.*, 2009). Cependant, il a été montré que les plantes, via leurs exsudats racinaires, peuvent fournir une source de carbone organique et favoriser le processus de dénitrification (Zhu et Sikora, 1995). Ces exsudats racinaires, qui sont en majorité des composés carbonés, représentent entre 5% et 11% du carbone fixé par la plante (Uren *et al.*, 2007; Lambert *et al.*, 2009; Jones *et al.*, 2009). Néanmoins, la quantité de carbone apportée par les exsudats racinaires en marais filtrant reste encore peu étudiée et on ignore si la quantité est suffisante ou bien si une source externe de carbone serait nécessaire. Des études menées par Prystay et Lo (2001), ont montré que le traitement des nitrates par marais filtrant plantés de *Typha latifolia* était faible (13-27%) en raison du manque de carbone nécessaire pour la dénitrification. Toutefois, la production d'exsudats racinaires ainsi que la capacité de dénitrification varie selon les espèces végétales (Lin *et al.*, 2002). Conséquemment, d'autres espèces de macrophytes pourraient donner des résultats différents. Par exemple, il a été montré que *Phalaris arundinacea* pouvait produire suffisamment d'exsudats racinaires riche en carbone organique pour permettre une bonne dénitrification (enlèvement de 78%) d'un effluent moyennement chargé en nitrate (NO_3 : 48 mg N/l) comparativement à *Typha latifolia* (<30%) (Zhu et Sikora, 1995). Toutefois, l'ajout d'une source de carbone externe peut être nécessaire pour avoir une épuration optimale lorsque la concentration en nitrate est élevée (Lin *et al.*, 2002; Zhu et Sikora, 1995). Plusieurs sources de carbone externe ont été utilisées afin de favoriser la dénitrification, dont le méthanol, l'éthanol, l'acide acétique, le glucose, le fructose et l'amidon (Her et Huang, 1995; Huett *et al.*, 2005; Park *et al.*, 2008). Cependant, pour que la dénitrification soit complète, l'effluent doit contenir un ratio spécifique entre le carbone et l'azote (C/N), lequel peut varier selon la source de carbone utilisé et sa biodégradabilité (Her et Huang, 1995). Des ratios C/N variant entre 1,13 à 5 ont été utilisés avec un enlèvement de nitrate de plus de 90% (Huett *et al.*, 2005).

La production de légumes en serre se fait aussi durant la période hivernale et donc le traitement des rejets par marais filtrant doit être efficace même lorsque les températures sont sous le point de congélation. Les marais sous surfaciques à flux horizontal ont été démontrés comme une technologie propice au traitement des eaux usées municipales en climat froid (Jenssen *et al.*, 1993), mais pour le moment aucune étude n'a été faite concernant le traitement des rejets hydroponiques. L'épuration des eaux en marais filtrant est possible en condition hivernale puisque l'effluent a une certaine chaleur et passe sous la surface du marais où il est en partie isolé contre le froid (Kadlec et Wallace, 2009). L'activité microbienne diminue avec la température, incluant le processus de dénitrification, dont l'activité a été mesurée jusqu'à un minimum de 4°C (Sirivedhin et Gray, 2006). De ce fait, l'enlèvement des nitrates dans les rejets hydroponiques pourrait être limitant en condition hivernale puisque le taux de dénitrification diminue avec la température et la concentration de nitrate élevée dans l'effluent. De plus, l'ajout de carbone organique via les exsudats racinaire est important pour le processus de dénitrification, mais il n'est pas connu si les exsudats sont produits lorsque les plantes sont en dormance durant la période hivernale.

1.3 Traitement des boues par marais filtrants

Les boues sont des déchets ayant une forte teneur en eau et sont produites par différentes sources telles que le traitement des eaux usées municipales et industrielles, en agriculture par la production de certains fumiers ainsi que par le dragage de voies maritimes (DeMaeseneer, 1997; Summerfelt *et al.*, 1999). La gestion des boues est un problème important puisqu'en raison de leur forte teneur en eau, elles ont un volume considérable et engendrent des coûts importants liés au transport et à leur disposition (Kim et Smith, 1997). De plus, les boues peuvent contenir une quantité plus ou moins grande de polluants, sous forme liquide et solide, qui peuvent être néfastes pour l'environnement (DeMaeseneer, 1997). Il existe plusieurs procédés industriels afin de réduire le volume des boues, dont l'utilisation de floculant chimique ainsi que des procédés mécaniques comme les presses, centrifugeuses et filtres à bandes et à succion (Kim et Smith, 1997; Edwards *et al.*, 2001). Toutefois, ces procédés sont coûteux, énergivores et nécessitent une main d'œuvre spécialisée (Barbieri *et al.*, 2003). Les lits de séchage de boues plantés à macrophytes (ou Lisam) offrent une alternative aux systèmes chimiques et mécaniques, permettant ainsi la réduction du volume des boues par un système extensif nécessitant peu d'énergie et dont les installations sont simples et peu coûteuses. Les lits de séchages de boue plantés de macrophytes offrent d'autres bénéfices que la simple déshydratation, tels que la minéralisation des polluants de boues et une épuration du lixiviat de boue à la sortie des lits de séchage.

Les Lisam sont constitués d'une matrice filtrante, ayant une granulométrie croissante de la surface vers le fond, sur laquelle sont disposées les boues et dans laquelle les plantes se développent (Figure 1.3).

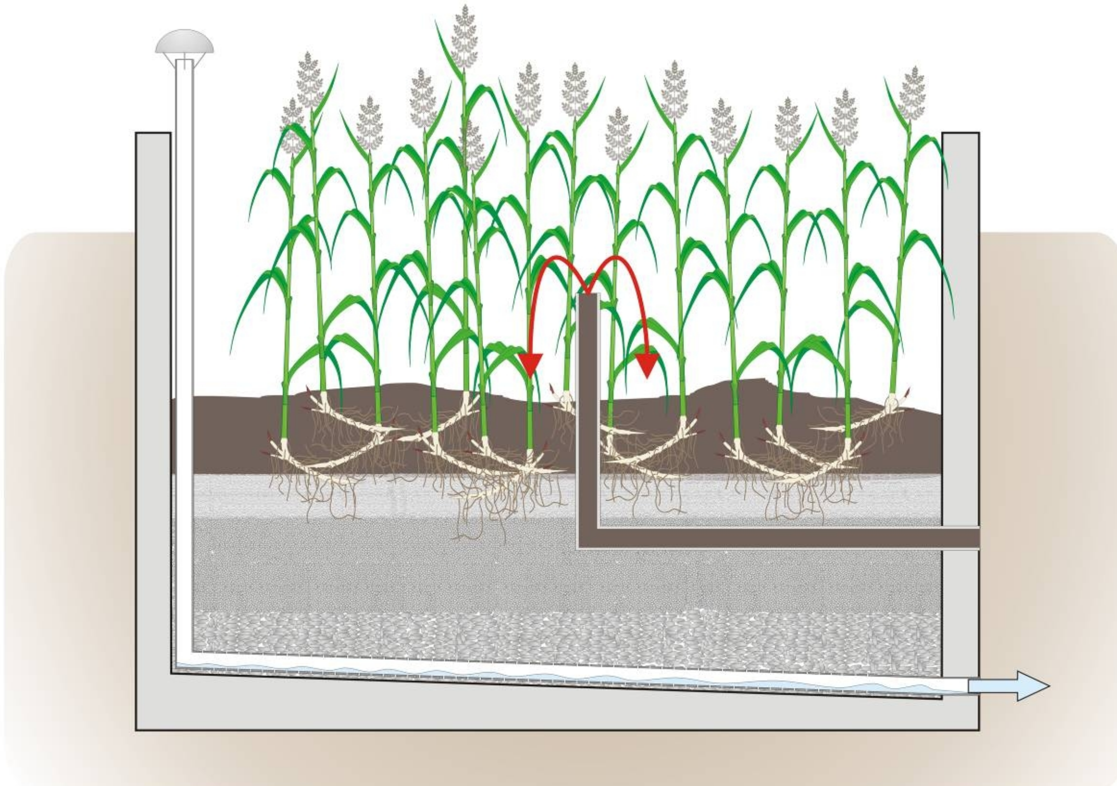


Figure 1.3 : Vue en coupe d'un lit de séchage de boue planté de macrophytes

Ces types de systèmes ont une architecture similaire aux marais filtrants à flux vertical (IWA, 2000), à la différence près qu'ils ont une plus haute paroi, d'environ 0,9 à 1,8 mètre de hauteur, afin de permettre un stockage des résidus de boue accumulée avec les années (Schmid *et al.,* 2003; Cooper *et al.,* 2004). Les Lisam sont habituellement des systèmes complètement drainés et dont l'effluent à la sortie (le lixiviat de boue) est envoyé à la station d'épuration lorsque cela est possible (DeMaeseneer, 1997; Kengne *et al.*, 2008). La fréquence d'application des boues est périodique, avec une application d'un volume défini suivie d'une période de repos allant de 7 à 65 jours (DeMaeseneer, 1997; Schmid *et al.*, 2003; Cooper *et al.*, 2004). Plusieurs Lisam sont utilisés en parallèle (2 à 25 lits) afin que le traitement des boues puisse se faire sans délais causés par la période de repos (Uggetti *et al.,* 2010). Les charges appliquées sur les lits de séchage plantés de macrophytes sont habituellement de 60 à 65 kg de matière sèche par mètre carré par année (MS m^{-2} an^{-1}), mais peuvent varier entre 12 et 200 kg MS m^{-2} an^{-1} selon le type de boue et la saison (DeMaeseneer, 1997; Kengne *et al.*, 2009; Uggetti *et al.*, 2010). Cependant, il est suggéré d'appliquer une plus faible charge de boues lors des deux premières années, c'est-à-dire entre 10 et 18 kg MS m^{-2} an^{-1}, afin d'assurer un démarrage adéquat du système (Liénard *et al.*, 1995; Burgoon *et al.*, 1997). Certaines recherches proposent même une période d'un an sans application de boue afin de permettre l'établissement des plantes (DeMaeseneer, 1997). Le climat, l'âge du système ainsi que les caractéristiques de la boue sont les facteurs qui peuvent influencer la charge maximale de boue qui peut être appliquée sur les lits (Schmid *et al.*, 2003). Les lits de séchage peuvent être utilisés pendant une période de 8-12 ans avant que les systèmes ne soient vidangés, ce qui correspond à une hauteur de résidu de boue d'environ 1,2 à 1,6 mètre (Schmid *et al.,* 2003, Nielsen et Willoughby, 2005)

L'une des fonctions principales des Lisam est de réduire le volume des boues, qui est généralement constituée de 85 à 99,5% d'eau (*i.e.* : 15 à 0,5% de matière sèche) (Uggetti *et al.*, 2010). Pour ce faire, plusieurs mécanismes physiques et biologiques entrent en jeux. Premièrement, l'assèchement des boues résulte de la rétention de la matière solide sur la matrice de filtration et la percolation d'une partie du lixiviat de boue à travers le système (Kengne *et al.*, 2008). La percolation du lixiviat de boue est particulièrement importante lors de l'alimentation du lit et diminue rapidement après la première journée (DeMaeseneer, 1997). Lors de la période de repos, la déshydratation du résidu de boue continue via l'absorption et la transpiration de l'eau par les plantes ainsi que par évaporation (Cooper *et al.,* 2004). La taille de la biomasse aérienne des plantes a été rapportée comme étant positivement corrélée au taux de transpiration des plantes (Wang *et al.*, 2009). De plus, la présence de plantes permettrait de limiter le colmatage des lits grâce à la formation, par les tiges et racines, de tunnels permettant le drainage de l'eau (Schmid *et al.*, 2003). La réduction du volume des boues varierait de 60% à 93% (Stefanakis *et al.*, 2009) et le pourcentage d'humidité du résidu de boue en surface des lits varierait entre 40% à 80% (DeMaeseneer, 1997; Uggetti *et al*., 2010). Il est à noter que les lits de séchage de boue sont presque exclusivement plantés de roseau commun (*Phragmites australis*), une plante ayant une forte biomasse aérienne, un bon développement racinaire ainsi qu'une tolérance élevée aux stress (DeMaeseneer, 1997). Toutefois, le *Phragmites australis* est considéré comme une plante envahissante dans plusieurs pays et il serait donc déconseillé de l'utiliser dans des régions sensibles (Edwards *et al.*, 2006). Cependant, il existe peu d'information sur l'influence de l'espèce de plante sur la déshydratation des boues, particulièrement sur la différence entre *Phragmites australis* et d'autres espèces pouvant être utilisées comme substituts. La morphologie des diverses espèces de plantes, tant au point de vue racinaire que de la biomasse aérienne, pourrait jouer un rôle important dans la diminution de volume des boues. De plus, d'autres facteurs peuvent aussi influencer la réduction du volume des boues, dont la composition de la boue. Un problème majeur auquel les lits de séchage font face est le colmatage de la matrice filtrante (Kengne *et al.*, 2009), ce qui limite la percolation du lixiviat de boue et diminue drastiquement la déshydratation. Par exemple, la présence de particules fines pourrait avoir un impact négatif important sur la déshydratation en favorisant le colmatage des lits (Healy *et al.,* 2007). Or, les boues produites par plusieurs procédés, comme en étangs aérés de station d'épuration et en étangs piscicoles, contiennent des particules fines puisque les étangs sont souvent imperméabilisés avec de l'argile qui se retrouve dans la boue lors de la vidange.

Un autre processus important dans les lits de séchage plantés de macrophyte est la minéralisation et l'oxydation des polluants de la boue (Liénard *et al.,* 1995; Nielsen, 2005; Cui *et al.,* 2008). La minéralisation de la boue vise à transformer la matière organique en composés minéraux généralement moins nocifs (Senesi et Plaza, 2007; Cui *et al.,* 2008).

La minéralisation de la boue se traduit par une diminution de la matière organique, mesurée par la réduction du contenu en matière volatile ainsi que la réduction de l'azote et du phosphore total (Cui *et al.,* 2008; Stefanakis *et al.,* 2009). Le contenu initial en matière volatile (*i.e.* : matière organique) des boues représente généralement 50 à 80% de la matière sèche de la boue fraîche (Uggetti *et al.,* 2010). Une fois traité par lit de séchage planté de macrophytes, la teneur en matière volatile du résidu de boue varie habituellement entre 35 et 50% de la matière sèche (Uggetti *et al.,* 2010; Melidis *et al.,* 2010). Dans les lits de séchage, les plantes permettent de créer des conditions favorables à la minéralisation par la diffusion d'oxygène dans la boue, via des tunnels créés par les tiges et racines ainsi que par le transport actif d'oxygène à la rhizosphère (Edwards *et al.,* 2001). De plus, la minéralisation de la boue entraînerait une diminution de son volume (Schmid *et al.,* 2003), puisqu'une fraction de la matière organique peut être biodégradée en éléments plus simples, comme en gaz et en eau. Également, le type de boue pourrait avoir une influence sur la minéralisation, car une boue contenant de l'argile, par exemple, pourrait colmater le lit de séchage et favoriser des conditions anaérobies et conséquemment ralentir ou modifier le type de minéralisation.

La qualité de l'eau en sortie des Lisam est un aspect rarement considéré comme important, puisque les Lisam sont majoritairement utilisés pour le traitement de boues de station d'épuration et le lixiviat de boue est généralement retourné en tête de traitement (Nielsen, 2005). Cependant, les Lisam sont parfois situés en milieu isolé, comme par exemple en région agricole, et le niveau d'épuration du lixiviat de boue est crucial afin de limiter les rejets de polluants dans l'environnement. Il est à noter que l'eau à la sortie des Lisam a généralement une concentration en polluants inférieure à celle issue de procédés mécaniques de déshydratation (Liénard *et al.,* 1995). La présence de plantes dans les lits de séchage de boue a été montrée comme ayant un effet positif sur l'enlèvement des polluants du lixiviat (Hofmann, 1990; Wang *et al.,* 2009; Wang *et al.,* 2010). L'espèce de plante semble aussi avoir un impact sur la qualité du lixiviat en sortie des Lisam avec une meilleur épuration associée à *Phragmites australis* et *Typha latifolia* comparativement à *Iris pseudacorus* (Wang *et al.,* 2009; Wang *et al.,* 2010). Il est présumé que l'épuration serait effectuée par des microorganismes situés dans la couche de boue ainsi que sur le sable et gravier et serait favorisée par la rhizosphère des plantes (Liénard *et al.,* 1995). De plus, malgré l'influence positive des plantes sur le traitement, la concentration des polluants à la sortie des lits de séchage est souvent trop élevée pour être rejetée directement dans l'environnement (DeMaeseneer, 1997). L'efficacité des Lisam pourrait être améliorée par une plus longue rétention du lixiviat dans la matrice filtrante au lieu d'un drainage immédiat (Burgoon *et al.,* 1997; Edwards *et al.,* 2001). Une rétention plus longue du lixiviat permettrait un contact prolongé entre les polluants et les microorganismes de la rhizosphère. En outre, la qualité de l'eau en sortie des lits de séchage de boue dépendra du lixiviat produit lors de la minéralisation des boues ainsi que du traitement de ce lixiviat à l'intérieur de la matrice de filtration.

1.4 Objectifs et hypothèses de recherche

L'objectif général de cette étude est de déterminer l'influence qu'ont les plantes ainsi que l'effet spécifique de l'espèce sur le traitement en marais filtrant artificiel sous des conditions extrêmement différentes. Ceci est fait selon trois volets de recherche ayant chacun des objectifs et hypothèses qui leurs sont propres. Le volet 1 porte sur le traitement des rejets hydroponiques par marais filtrant, le volet 2 porte sur la qualité de l'eau en sortie des Lisam et le volet 3 sur l'assèchement et la minéralisation des boues ainsi que le destin des polluants en Lisam. Le détail des objectifs et hypothèses est présenté selon les différents volets dans le tableau suivant :

Tableau 1.3. Objectifs et hypothèses de recherche

Volets	Objectifs	Hypothèses
Volet #1 : **Traitement des rejets hydroponiques par marais filtrant**	1.1 Déterminer l'influence de la présence de plantes sur l'enlèvement des polluants	1.1 Les marais plantés sont plus efficaces que les marais non plantés à l'enlèvement des polluants
	1.2 Déterminer l'influence de l'espèce de plante sur l'enlèvement des polluants	1.2 L'enlèvement des nitrates est supérieur dans les marais dont les plantes sont reconnues comme ayant de riches exsudats racinaires (ex. : *Phalaris*)
	1.3 Déterminer l'effet de l'ajout d'une source de carbone externe sur l'enlèvement des polluants	1.3 Les marais pour lesquels une source de carbone est ajoutée à l'effluent a une meilleure épuration des nitrates
Volet #2 : **Traitement des boues par Lisam :** **Qualité de l'eau en sortie des systèmes**	2.1 Déterminer l'influence de la présence de plantes sur la qualité de l'eau en sortie des Lisam	2.1 La présence de plantes dans les Lisam favorise la qualité de l'eau en sortie de ces systèmes
	2.2 Déterminer l'influence de l'espèce de plante sur la qualité de l'eau en sortie des Lisam	2.2 L'espèce de plante ayant la plus forte biomasse (ex. *Phragmites*) ont une meilleure épuration
	2.3 Déterminer l'influence de l'ajout d'argile sur la qualité de l'eau en sortie des Lisam	2.3 L'ajout d'argile diminue la performance épuratoire dû au colmatage des systèmes
Volet #3 : **Traitement des boues par Lisam :** **Assèchement et minéralisation de la boue ainsi que le destin des polluants**	3.1 Déterminer l'influence de la présence de plantes sur l'assèchement et la minéralisation de la boue en Lisam	3.1 La présence de plantes dans les Lisam favorise l'assèchement et la minéralisation de la boue.
	3.2 Déterminer l'influence de l'espèce de plante sur l'assèchement et la minéralisation de la boue en Lisam	3.2 L'espèce de plante ayant une forte densité et une haute évapotranspiration (ex. *Phragmites*) assèche et minéralise la boue plus efficacement
	3.3 Déterminer le destin des polluants dans les Lisam selon les espèces de plantes	3.3 La majorité des polluants restent dans la boue en surface des Lisam ou bien est minéralisée et peu de polluants sont attendus en sortie des Lisam plantés de *Phragmites*

Bien que l'influence positive des plantes en marais filtrant commence à être de plus en plus établie, les connaissances acquises sont principalement basées sur le traitement des eaux usées municipales en condition estivale. L'originalité de cette étude vient donc du fait qu'elle tentera de répondre à plusieurs questions concernant le rôle des plantes ainsi que de l'influence spécifique de l'espèce sur le traitement des polluants sous des conditions qu'on peut qualifier d'extrêmes. Par exemple, le traitement des rejets hydroponiques en condition hivernal est problématique, car ce type de rejet est fortement chargé en nitrate, mais ne contient aucun carbone organique nécessaire pour la dénitrification. De plus, le processus microbien de dénitrification diminue avec la température et peut donc être limité en hiver. Il est reconnu que les plantes, en particulier certaines espèces, peuvent favoriser la dénitrification en libérant du carbone organique par leur exsudat racinaire. Toutefois, la capacité des plantes à fournir suffisamment de carbone pour le traitement des nitrate sous des conditions hivernales est encore inconnue. Cette étude est donc unique puisqu'elle tentera de mesurer l'effet des plantes selon des facteurs qui poussent les limites physiologiques de celles-ci tout en respectant des conditions réelles rencontrées lors du traitement des eaux en marais filtrant. Le traitement des boues est aussi un défi, car la concentration en polluant peut être 100 fois plus élevé qu'un effluent municipal conventionnel. De ce fait, l'influence des plantes sur la qualité de l'eau en sortie reste à être clairement démontrée. De plus, l'effet des plantes sur la minéralisation et déshydratation des boues n'est pas bien établi, puisque le peu d'étude réalisée à ce sujet sont contradictoires. L'aspect novateur de cette étude est de tester l'influence de l'espèce de plante sur la boue via la méthode conventionnelle du pourcentage de polluants par matière sèche, mais aussi en effectuant un bilan de masse des polluants dans le système.

1.5 Méthodologie, traitement des rejets hydroponiques (volet 1)

Les expériences ont été effectuées en conditions contrôlées dans les serres de recherche du Jardin Botanique de Montréal, sous une simulation d'un climat hivernal. L'installation expérimentale consistait de mésocosmes de 1 m^2 plantés en monoculture de *Phalaris arundinacea*, *Phragmites australis*, *Typha* sp. ainsi que d'un témoin non planté, le tout en duplicata. Les marais ont été alimentés avec 30 litres par jour d'une solution d'engrais imitant les rejets d'une culture hydroponique. Une moitié de l'installation a été alimentée avec seulement la solution d'engrais, tandis que l'autre moitié a été alimentée avec la solution d'engrais additionnée d'une source carbonée (Figure 1.3).

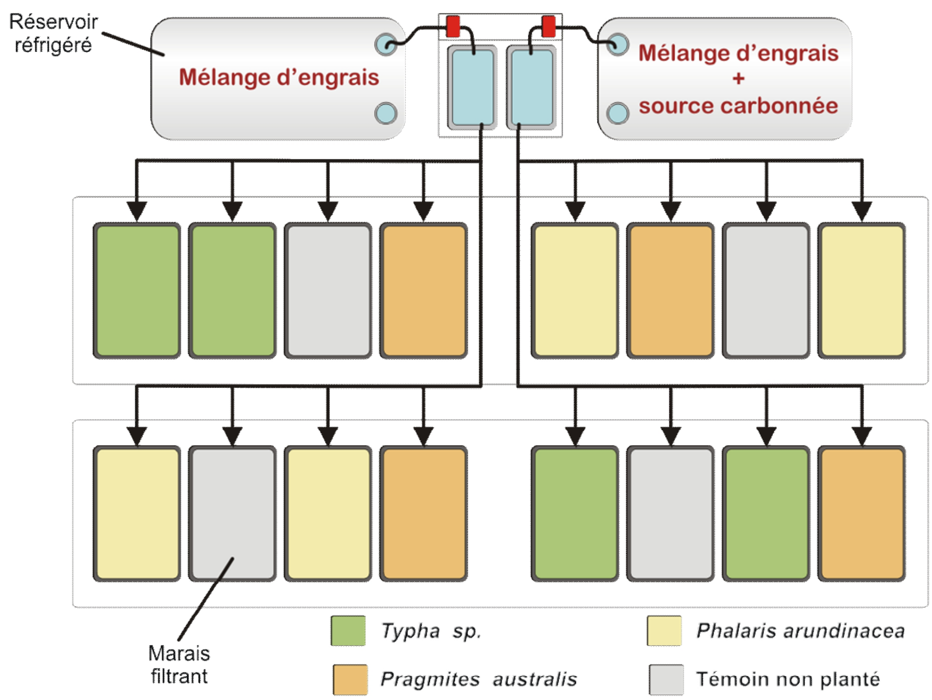


Figure 1.4. Vue en plan des installations expérimentales avec le détail de la répartition de l'effluent selon l'espèce de plante.

1.6 Méthodologie, traitement des boues (volets 2 et 3)

Les expériences a été effectuées avec un total de 16 mésocosmes (hauteur 1 m; rayon 0,3 m) plantés en monoculture de trois espèces de macrophyte, *Phragmites australis*, *Typha* sp., et *Scirpus fluviatilis* ainsi que des témoins non plantés. Les mésocosmes ont été alimentés avec deux types de boue piscicole (avec ou sans argile) durant l'été 2008, 2009 et 2010 (Figure 1.5). La boue a été appliquée de façon intermittente (1 jour de d'alimentation suivi par six jours de repos) à une charge de 412 g MS m^{-2} semaine (2008, 9 $semaines^{-1}$), 338 g MS m^{-2} semaine (2009, 12 $semaines^{-1}$) et enfin 575 g MS m^{-2} semaine (2010, 14 $semaines^{-1}$) pour un total de 0,59 m^3 m^{-2} de boues.

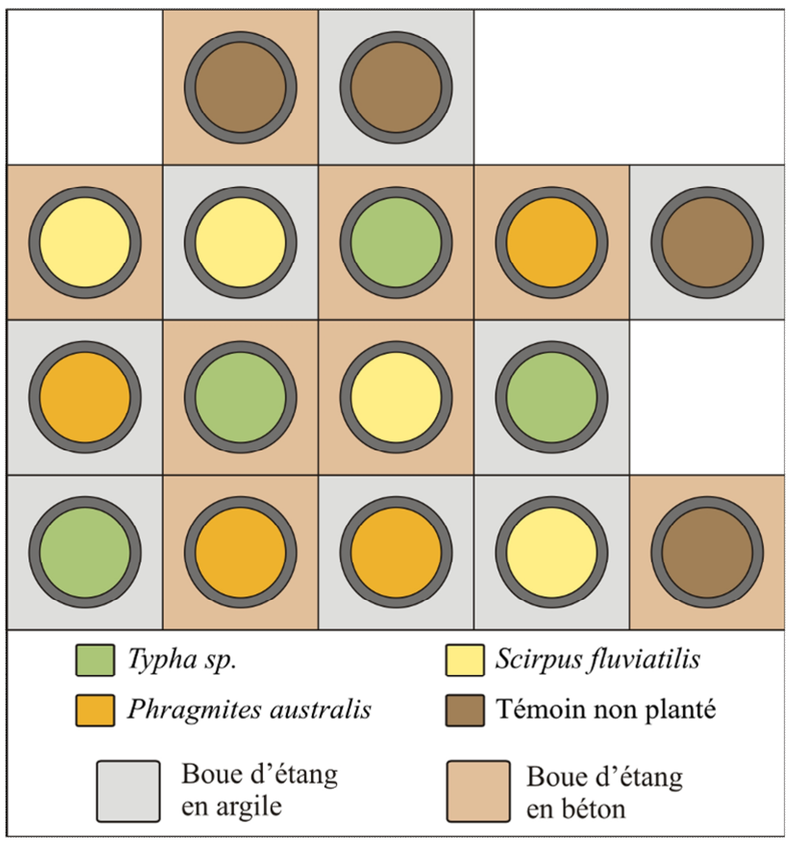


Figure 1.5. : Vue en plan des lits de séchage de boues plantés de macrophyte

1.7 Organisation de la thèse

Le chapitre 1 a présenté une introduction générale, une revue de la littérature, les hypothèses et objectifs de recherche ainsi qu'une brève aperçu de la méthodologie utilisée. Les chapitre 2-3-4 constituent le corps principal de la thèse et relatent les principaux résultats des recherches sous forme de trois articles scientifiques. Le chapitre 2 présentera le traitement de rejets hydroponiques par marais filtrant en condition hivernale, tandis que les chapitres 3 et 4 portera sur le traitement des boues par lit de séchage à macrophytes (Lisam) en condition estivale. Plus spécifiquement, le chapitre 3 exposera l'influence des plantes sur la qualité de l'eau en sortie des Lisam et le chapitre 4 ce concentrera sur l'effet des plantes sur la boue. Les annexes présenteront une multitude de détails qui n'ont pas été intégrés au corps de la thèse afin de ne pas alourdir le texte. L'annexe 1 est un tableau synthèse présentant une revue de la littérature portant sur les caractéristiques des rejets hydroponiques. L'annexe 2 montre les données sur le traitement des rejets hydroponiques. L'annexe 3 est un tableau synthèse présentant une revue de la littérature sur les caractéristiques des boues. L'annexe 4 montre les données des polluants en sortie des Lisam. L'annexe 5 montre les bilans de masse des polluants dans les Lisam de 2008 à 2010. Finalement, l'annexe 6 porte sur la comparaison entre l'efficacité épuratoire retrouvée dans la littérature et les valeurs trouvées par cette étude.

1.8 Apport du postulant au Doctorat et de ses coauteurs aux articles

Le chapitre 2 a été publié en 2010 sous le titre « *Treatment of a hydroponics wastewater using constructed wetlands in winter conditions* », dans la revue Water, Air, and Soil Pollution[1]. Le chapitre 3 intitulé *« Effect of plant species on water quality at the outlet of a sludge treatment wetland* » a été publié dans la revue Water Research[2] en octobre 2012. Le chapitre 4 est prêt pour soumission à la revue Ecological Engineering. Tous les articles ont été en totalité écrits par le postulant au Doctorat et sont basés sur ses résultats d'expériences. Les coauteurs ont fourni le support intellectuel nécessaire pour la planification et l'exécution des expériences ainsi que pour la relecture de l'article.

[1] Gagnon, V., Maltais-Landry, G., Puigagut, J., Chazarenc, F. and Brisson, J. (2010). Treatment of Hydroponics Wastewater Using Constructed Wetlands in Winter Conditions. Water Air and Soil Pollution 212(1-4), 483-490.

[2] Gagnon, V., Chazarenc, F., Kõiv, M. and Brisson, J. (2012). Effect of plant species on water quality at the outlet of a sludge treatment wetland. Water Research 46(16-15), 5305-5315.

Chapitre 2: Treatment of a hydroponics wastewater using constructed wetlands in winter conditions.

V. Gagnon, G. Maltais-Landry, J. Puigagut, F. Chazarenc and J. Brisson

2.1 Abstract

Hydroponics culture generates large amounts of wastewater that are highly concentrated in nitrate and phosphorus but contains almost no organic carbon. Constructed wetlands (CWs) have been proposed to treat this type of effluent, but little is known about the performance of these systems in treating hydroponic wastewater. In addition, obtaining satisfactory winter performances from CWs operated in cold-climates remains a challenge as biological pathways are often slowed down or inhibited. The main objective of this study was to assess the effect of plant species (*Typha* sp., *Phragmites australis*, *Phalaris arundinacea*) and the addition of organic carbon on nutrient removal in winter. The experimental setup consisted of 16 subsurface flow CWS mesocosms (1 m^2, HRT of 3 days) fed with 30 L d^{-1} of synthetic hydroponics wastewater, with half of the mesocosms fed with an additional source of organic carbon (sucrose). Carbon addition had a significant impact on pollutant removal in all systems, with the means of nitrate removal passing from 0.5 to 4.9 g $m^{-2}d^{-1}$ and the means of phosphate passing from 0.3 to 0.5 g $m^{-2}d^{-1}$ in planted CWs when carbon was added. Planted mesocosms were generally more efficient than unplanted controls. Furthermore, we found significant differences among plant treatments for NO_3-N (highest removal with *Phalaris*) and COD (highest removal with *Phragmites*/*Typha* sp.). Overall, planted wetlands with added organic carbon represent the best combination to treat hydroponics wastewater during the winter.

2.2 Introduction

Hydroponics culture requires large quantities of water and chemical fertilizers to optimize plant production. Consequently, this type of agriculture produces large amounts of point source pollution highly concentrated in nitrate (200-300 mg NO_3-N L^{-1}) and phosphorus (30-100 mg PO_4-P L^{-1}), but containing almost no organic carbon (Park *et al.*, 2008; Prystay and Lo, 2001). Current technologies for hydroponics wastewater treatment – ion-exchange membranes, reverse osmosis – are efficient (Koide and Satta 2004) but have high operational and maintenance costs compared to standard wastewater treatment systems. Constructed wetlands (CWs), especially horizontal subsurface flow constructed wetlands (HSSF CWs), have been proposed as an efficient and inexpensive alternative to treat hydroponics wastewater (Grasselly *et al.,* 2005).

The complex and positive contribution of macrophytes to overall pollutant removal in HSSF CWs is currently widely accepted (Brix 1997). For instance, plants may supply organic carbon to stimulate denitrification (Zhu and Sikora 1995) when wastewaters are rich in nitrate but poor in organic carbon (*e.g.* hydroponics wastewater). However, carbon compounds from root exudates could be insufficient to completely remove high concentrations of nitrate. Indeed, Prystay and Lo (2001) reported low nitrate removal (13-27%) that was comparable to unplanted systems in HSSF CWs planted with *Typha latifolia* and treating hydroponics wastewater. Also, as production of root exudates and denitrification capacity differs among plant species (Lin *et al.*, 2002), other macrophyte species could yield different results. Hence, determining which species are more efficient under a variety of conditions remains necessary (Brisson and Chazarenc, 2009).

In addition to the organic carbon supplied by plants, an external source of organic carbon may be required to obtain satisfactory nitrate removal when wastewater nitrate concentrations are high (Lin *et al.*, 2002; Zhu and Sikora, 1995). Methanol, ethanol, acetic acid, glucose, fructose and starch are common external carbon sources used to enhance denitrification in carbon-poor effluents (Her and Huang, 1995; Huett *et al.*, 2005; Park *et* *al.*, 2008). However, complete denitrification requires a specific ratio of carbon:nitrogen that varies depending on carbon type and lability (Her and Huang, 1995). Optimal nitrate removal in constructed wetlands was observed at a COD:N ratio of 3.5 using fructose as a carbon source (Lin *et al.*, 2002) and a C:N ratio of 3 when methanol was used (Huett *et al.*, 2005). However, plant species, effluent type and loading could interact with the effects of carbon addition, resulting in different ratios than those previously reported.

CWs operated in cold climates face another challenge, as winter removal efficiency must remain satisfactory with reduced rates of biological removal pathways and plant dormancy (Kadlec et Wallace, 2009). Although denitrification rates decline with colder temperatures, nitrate removal in HSSF CWs was observed at temperatures as low as 4°C (Sirivedhin and Gray, 2006). Also, significant differences observed among plant species during summer may become non-significant during winter (Riley *et al.*, 2005). This highlights the need to better quantify the effects of winter on nutrient removal and potential interactions with other factors (*e.g.* effects of plants).

We used a mesocosm experiment to assess the individual and combined effects of plant species (*Typha* sp., *Phragmites australis*, *Phalaris arundinacea*) and the addition of organic carbon (sucrose) on nutrient removal from hydroponics wastewater in HSSF CWs operated in winter conditions. Given the constraints of winter operation and composition of hydroponics wastewater, its optimal year round treatment with HSSF CWs represents a real challenge, and suitable operating conditions can only be identified with the simultaneous evaluation of several factors as we report in this study.

2.3 Methods

2.3.1 Experimental setup

The mesocosm experiment ran from November 2007 to April 2008 in a controlled greenhouse located at the Botanical Garden of Montreal. The experiment simulates a HSSF CWs under a temperate climate with cold winters (no plant activity). In this case, the belowground environment of constructed wetlands does not freeze due to soil insulation, flowing wastewater, and often insulation by snow cover. To mimic those conditions, the air temperature in the greenhouse (and thus, mesocosm temperature) was kept at a minimum of 5°C (average 8°C). To further simulate winter conditions, at the beginning of the experiment, the aboveground portion of the macrophytes (most of which was already in senescence) was harvested and removed. The greenhouse was also shaded with a large tarp to reduce light and temperature as an additional measure to prevent any aboveground plant activity.

The experimental setup consisted of 16 subsurface flow mesocosms of 1 m^2 (1.25 m long, 0.8 m wide, and 0.3 m deep) operated at a HRT of 3 days. The mesocosms were filled with granitic river gravel (Ø = 10-15 mm), with a narrow section at the inlet filled with granitic coarse gravel (Ø = 30-40 mm) to facilitate water distribution. Water level was kept 4 cm under the substrate surface. The mesocosms were planted in monocultures of *Typha* sp., *Phragmites australis* and *Phalaris arundinacea*, in addition to unplanted controls with four mesocosms for each of the four species treatments. The treated effluent was collected daily at the outlet of each mesocosm. The mesocosms were planted in 2002 from rhizomes and were used to treat fish farm wastewater until 2006 (Ouellet-Plamondon *et al.*, 2006). In the summer of 2007, the mesocoms were fed with reconstituted hydroponics wastewater for acclimatization before the first experimental measurements were conducted in November 2007.

2.3.2 Wastewater characteristics

Two types of wastewater were used, one with reconstituted hydroponics wastewater only and the other with the same hydroponics wastewater but supplemented with organic carbon (sucrose). The hydroponics wastewater was reconstituted using chemical fertilizers following effluent nutrient concentrations published by Prystay and Lo (2001), Park *et al.,* (2008), Koide and Satta (2004), as well as information collected by personal communication with greenhouse producers (see Table 2.1 for concentrations and loading). The wastewater with organic carbon had the same fertilizer concentration but was supplemented with sucrose to reach a final concentration of 815 mg COD L^{-1} (C/N: 1.3; COD/N: 3.5). During the experiment, half of the mesocosms received chemical fertilizers only while the other half received fertilizers supplemented with organic carbon. This resulted in two mesocoms being attributed to each of the 8 combinations of species/wastewater treatments. The mesocosms received wastewater in an intermittent batch mode of 30 L m^{-2} d^{-1}. New wastewater solutions were prepared approximately every week and stored in refrigerated bulk tanks (4ºC) until delivery to prevent major changes in wastewater composition.

2.3.3 Physical-chemical analyses

Inlet and outlet samples (24 hours composite sampling) were collected 1 to 4 times a month for a total of 13 samples per mesocosm. The following variables were measured according to Standard Methods (APHA *et al.*, 1998): COD, NO_3-N, NH_4-N, PO_4-P. Evapotranspiration (ET) was estimated on every sampling date by measuring the total inflow and outflow. Removal efficiencies were calculated based on mass balance.

2.3.4 Modeling

Treatment of hydroponics wastewater by HSSF CWs should be primarily based on the removal of nitrogen rather than organic carbon as the latter is intentionally added to stimulate denitrification. Hence, treatment performances for nitrogen were simulated using a first order kinetic plug flow model (Kadlec et Wallace, 2009) to estimate the effect of carbon and plant species on degradation kinetics:

$$\frac{Cs}{Co} = e^{(-K_{VT} \cdot \tau)} \quad (1)$$

$$K_{VT} = K_V \cdot (\theta)^{(T_{avg} - 20)} \quad (2)$$

Cs: mass flow at outlet (mg d^{-1})

Co: mass flow at inlet (mg d^{-1})

K_{VT}: first-order volumetric rate constant at temperature T° (d^{-1})

τ: experimental residence time distribution (d)

K_V: first-order volumetric rate constant at 20°C (d^{-1})

T_{avg}: average daily air temperature (°C)

θ: empirical abiotic constant, set to 1.06 following Rousseau *et al.* (2004).

Experimental results were fitted using this model with T_{avg} = 8°C, and gravel porosity = 30%. Evapotranspiration was taken into account as mass flow was used instead of concentration. We calculated K_{V20} for total N (TN = NO_3-N + NH_4-N), NO_3-N and NH_4-N removal for systems with added carbon as they were the only ones with satisfactory N removal. R^2 were also computed between observed values and those predicted by the model to evaluate goodness of fit.

2.3.5 Statistical analyses

We used a two-way repeated measures analysis of variance (ANOVA) followed by a test of multiple comparisons of means (Tukey HSD) to test the effects of macrophytes species and carbon addition. All variables met the normality assumptions, and a square-root-transformation was applied for COD data to reduce the heterogeneity of the variances as measured by the Levene test. However, ANOVA was not performed for NH_4-N because no data transformation could adequately reduce the heterogeneity of the variances, except rank transformation, which is not recommended for designs including interactions (Quinn and Keough, 2003). Statistical tests were considered significant at the 0.05 level and were performed with SPSS 16.0 statistical software.

2.4 Results and discussion

2.4.3 Nitrate removal

Nitrate removal from hydroponics wastewater under winter conditions ranged from 4% (*Phragmites* no carbon) to 79% (*Phalaris* with carbon). Carbon addition had a highly significant impact on nitrate removal ($F_{1,8}$ = 1607.7, P<0.001) in all treatments, with outlet concentrations 3 times lower and removal efficiencies 10 times higher when carbon was added (Table 2.1). Planted mesocosms had slightly higher nitrate removal than unplanted controls with carbon addition, but removal was generally similar without carbon addition (Table 2.1). Wetlands planted with *Phalaris* removed more nitrate than wetlands planted with *Phragmites* regardless of carbon addition ($F_{3,8}$ = 4.4, P<0.05), whereas the other two treatments did not differ from either *Phalaris* or *Phragmites* (Figure 2.1a).

Table 2.1. Inlet concentration, loading, and removal of pollutants expressed as percentage of removal according to plant presence (mean of all species) and carbon

				Planted		**Unplanted**	
		Loading ($g\ m^{-2}d^{-1}$)	Inlet ($mg\ L^{-1}$)	Outlet ($mg\ L^{-1}$)	Removal	Outlet ($mg\ L^{-1}$)	Removal
Carbon	NO_3-N	6.90	228 ± 7	65 ± 20	72%	72 ± 31	69%
	NH_4-N	0.51	17 ± 2	6 ± 3	68%	10 ± 5	44%
	PO_4-P	1.68	57 ± 4	40 ± 10	30%	44 ± 9	21%
	COD	24.68	814 ± 55	28 ± 12	97%	55 ± 20	93%
No carbon	NO_3-N	6.84	225 ± 8	211 ± 24	7%	213 ± 28	7%
	NH_4-N	0.52	16 ± 2	6 ± 3	63%	8 ± 5	54%
	PO_4-P	1.73	58 ± 3	52 ± 7	10%	54 ± 7	6%
	COD	0.70	24 ± 8	15 ± 9	33%	17 ± 10	25%

Carbon addition was the main factor that influenced nitrate removal in our study, as it greatly increased nitrate removal from a mean of 0.5 g N m^{-2} d^{-1} (no carbon) to 4.9 g N m^{-2}

d^{-1} (with carbon). However, even with the addition of carbon, denitrification was probably saturated as outlet nitrate concentrations remained high (range: 50 to 73 mg NO_3-N L^{-1}) and COD removal was almost complete (93 to 97%). This suggests that denitrification could be further stimulated by increasing the C/N ratio beyond the COD/N ratio used for this experiment (COD/N ratio: 3.5). This ratio was used by Lin *et al.* (2002) and they reported nitrate removal efficiencies higher than 90% compared to 69-72% for our system. This lower performance could be caused by greater competition for the added carbon between denitrifying bacteria and other wetland bacteria along with other factors (residence time, carbon source, and season). Therefore, the optimal COD/N ratio does not seem to be universal and more ratios should be tested in order to optimize external carbon addition. As there is only a small range of COD/N ratio within which the effluent concentration of both nitrate and COD remain low (Park *et al.,* 2008), finding the optimal ratio is a delicate but necessary task that remains to be done more exhaustively.

Although their effect was more subtle, plants species also increased nitrate removal. Mesocosms planted with *Phalaris* were the most efficient but the differences were only marginally significant ($p = 0.042$). Plants are thought to stimulate nitrate removal through plant uptake, but this should be negligible during the winter. Hence, the differentiation among plant species is more likely to be a function of carbon and oxygen availability. *Phalaris* has a shallow (but dense) root system (Gagnon *et al.*, 2007). The larger culms and belowground structures of *Typha* and *Phragmites* could result in higher oxygen diffusion from their cut stumps to the rhizosphere, which may partially inhibit denitrification. Also, *Phalaris* may release higher quantities of carbon through root exudates and decaying biomass (Zhu and Sikora, 1995), which could stimulate denitrification.

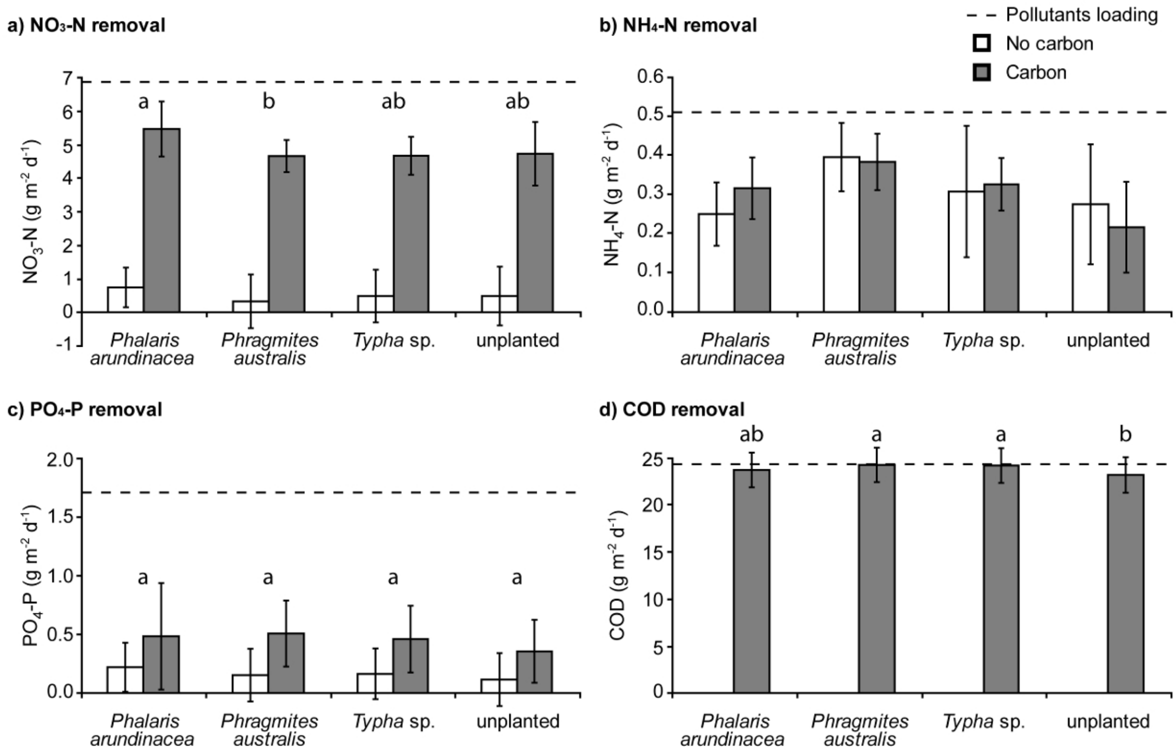


Figure 2.1: Mean removal (in g $m^{-2}d^{-1}$) according to plant species and carbon addition for a) NO_3-N, b) NH_4-N, c) PO_4-P, and d) COD. Error bars are standard deviations of the mean (n=13). Lower-case letters represent a significant difference at α=0.05 among plant treatments following an ANOVA and Tukey HSD. COD removal without carbon is not represented in the figure because the values were extremely low compared to the removal found with carbon addition and could not be adequately represented in the Figure (see Table 2.1 for details on inlet/outlet concentrations). ANOVA was not performed for NH_4-N because of the high heterogeneity of the variances.

The relative effect of plants is further supported by K_{V20} values for NO_3, which were highest with *Phalaris* and lowest with *Typha* (Table 2.2). As NO_3 was the dominant species of nitrogen in this study, the pattern among treatments in K_{V20} values for TN is similar to that of NO_3 (Table 2.2). Overall, the kinetic rate observed for NO_3 removal was up to ten times greater compared to others studies conducted in horizontal CWs, where K_{V20} for TN ranged from 0.06 to 0.16 day^{-1} (Rousseau *et al.*, 2004). This suggests that denitrification was the main biological path for TN and COD removal in our experiment. Values of K_{V20} also highlight that CWs planted with *Phalaris* would be more efficient per unit area than other plant treatments and could be the species of choice for this effluent given the large quantities of wastewater generated by the hydroponics industry.

Table 2.2: K_{V20} (day $^{-1}$) values for TN (NO_3 + NH_4), NO_3 and NH_4 according to plant treatments in wetlands with carbon addition. R^2 values between observed and predicted values are given in parentheses.

	Phalaris arundinacea	***Phragmites australis***	***Typha* sp.**	**Unplanted**
TN	1.13 (R^2 = 0.80)	0.88 (R^2 = 0.94)	0.85 (R^2 = 0.91)	0.91 (R^2 = 0.71)
NO_3	1.16 (R^2 = 0.77)	0.87 (R^2 = 0.93)	0.85 (R^2 = 0.90)	0.96 (R^2 = 0.68)
NH_4	0.67 (R^2 = 0.71)	1.05 (R^2 = 0.94)	0.72 (R^2 = 0.85)	0.37 (R^2 = 0.64)

2.4.4 Ammonium removal

No statistical test could be performed for ammonium: interpretation of the results should be considered with care as they are based only on visual examination of the patterns (Figure 2.1b). Planted mesocosms removed more ammonium than unplanted controls, and removal efficiencies ranged from 44% (unplanted with carbon) to 79% (*Phragmites* no carbon) depending on conditions. Regardless of carbon addition, *Phragmites* seemed to perform better than other plant treatments. Also, carbon addition appears to stimulate slightly NH_4 removal in planted mesocosms (increase from 63% to 68%) and reduced NH_4 removal in unplanted mesocosms (decrease from 54% to 44%) (Table 2.1).

Higher ammonia removal in planted wetlands is often attributed to the enhancement of nitrification via root oxygen release and direct ammonia uptake by plants (Tanner *et al.*, 2002), but plant uptake should be negligible in winter as it was for nitrate. As ammonia removal appears to vary among plant species – *Phragmites* being the most efficient – this suggests that oxygen availability in the wetland matrix differs as a function of plants species (Stottmeister *et al.*, 2003), even during winter. A differential oxygen release capacity among plants also supports the pattern we found among plants for NO_3 removal, and K_{V20} values for NH_4 (Table 2.2). Indeed, K_{V20} values were highest for *Phragmites* and lowest for unplanted wetlands. Also, while *Phalaris* was the most efficient treatment at removing NO_3, it was the least efficient plant to remove NH_4. This suggests that species choice in CWs should be dependent on N speciation in the wastewater, as not all plants are equally efficient at all stages of treatment or all seasons. Our results also suggest that using different plants at different stages of treatment could optimize nitrogen removal if reduced N forms (organic or NH_4) are dominant. For example *Phragmites* could be used to stimulate nitrification in a first CW and *Phalaris* could be used to stimulate denitrification in a second CW, at least in winter conditions.

Carbon addition generally enhanced ammonia removal in planted mesocosms, whereas it reduced ammonia removal in unplanted controls. Lower ammonia removal in unplanted mesocosms was expected as lack of plant mediated oxygen release and high competition for oxygen between ammonia oxidizing and heterotrophic bacteria at high COD concentrations may inhibit nitrification (Michaud *et al.,* 2006). However, the increase in ammonia removal with carbon addition in planted mesocosms does not follow classical trends in nitrification but could be due to an alternative process, heterotrophic nitrification, where the oxidation of ammonium is achieved by heterotrophic bacteria (Tanner *et al.*, 2002). Increased ammonia removal with higher COD concentration was reported by Riley *et al.* (2005) in winter conditions and was attributed to increased plant-mediated oxygen availability in winter versus summer. Furthermore, the intensity of root oxygen release seems to be controlled by the external oxygen demand of the rhizosphere, with higher oxygen release rates in lower redox conditions (Wiessner *et al.*, 2002). Therefore, higher oxygen availability at higher COD concentration could favor ammonia oxidizing bacteria and lead to higher ammonia removal when carbon is added to planted systems.

2.4.5 Phosphate removal

Phosphate removal ranged between 6% (unplanted with no carbon) and 30% (*Phragmites* with carbon), and planted mesocosms had slightly higher phosphate removal than unplanted controls (Table 2.1). We found no statistical difference in phosphate removal among plants species, with or without carbon addition (Figure 2.1c). Regardless of plant treatment, mesocosms supplemented with carbon had significantly higher phosphate removal ($F_{1,8}$ = 136.9, P<0.001) than mesocosms with no carbon supply (Figure 2.1c).

Although carbon addition enhanced phosphate removal from a mean of 0.2 g P m^{-2} d^{-1} (no carbon) to 0.5 g P m^{-2} d^{-1} (with carbon), outlet concentrations remained high with an average of 41 mg PO_4-P L^{-1}. Phosphorus removal is generally attributed to physical and chemical processes in HSSF CWs, with pH and redox often playing key roles (Vymazal, 2007; Watson *et al.,* 1989), although microbial uptake and storage can also be important (van Rijn *et al.*, 2006). Indeed, the substantial addition of organic carbon (24.42 g COD m^{-2} d^{-1}) given to some mesocosms may have stimulated phosphate removal by promoting the growth of heterotrophs and the sequestration of phosphorus in microbial biomass. On the contrary, mesocosms not supplemented in carbon may have experienced relatively limited microbial growth and consequently lower phosphorus removal. This is supported by Park *et al.* (2008) who attributed high phosphorus removal in denitrification filters with added carbon to storage in denitrifying bacteria. However, high carbon availability seems to be important for high P bacterial retention as Huett *et al.* (2005) reported that phosphorus removal was unaffected by external carbon addition with a COD supply about 800 times lower than ours. Furthermore, microbial phosphorus sequestration is often considered as a short term sink, as a substantial fraction of this phosphorus may be released at death (Vymazal 2007). Nevertheless, the production of refractory organic compounds of bacterial origin may lessen phosphorus release, due to the extremely slow biodegradation of these compounds (Gachter and Meyer, 1993), thus contributing to sustained removal of phosphorus.

Apart from carbon addition, plant presence also had a slightly positive effect on phosphate removal (Table 2.1). As phosphorus uptake by plant may be assumed to be negligible in winter, the contribution of macrophytes to phosphorus removal in our experiment was probably due to the indirect effect of plants on bacteria and redox.

2.4.5 Chemical oxygen demand

COD removal ranged from 24% (*Phragmites* no carbon) to 97 % (*Typha*/*Phragmites* with carbon), and plant presence stimulated COD removal, regardless of carbon addition (Table 2.1). Mesocosms planted with *Typha* or *Phragmites* had significantly higher COD removal than unplanted controls ($F_{3,8} = 7.8$, $P<0.05$), whereas *Phalaris* was intermediate and not statistically different from the other plant treatments (Figure 2.1d). Mesocosms supplemented with carbon had near complete COD removal (93% to 97%) and logically removed significantly more COD than the ones with no added carbon ($F_{1,8} = 60.3$, $P<0.001$) (Table 2.1). This near complete COD removal at high concentrations (COD: 24.42 g m^{-2} d^{-1}) reflects the high biodegradability of our carbon source, sucrose.

Higher COD removal in planted mesocosms has been reported previously (Ouellet-Plamondon *et al.*, 2006) and the difference between plants species could be explained by specificity of root oxygen release (Stottmeister *et al.*, 2003). The lower efficiency of *Phalaris* for COD removal may also be attributed to a greater carbon release, through production of root exudates (Zhu and Sikora, 1995) or decaying biomass. Finally, the COD fluctuation at the outlet of the mesocosms without added carbon, between 13-25 mg L^{-1}, can be attributed to normal organic carbon release by wetlands (Riley *et al.*, 2005).

2.5 Conclusion

Planted wetlands with added organic carbon represent the best combination for hydroponics wastewater removal, with winter pollution removal averaging 72%, 68%, 30% and 97% for nitrate, ammonium, phosphate and COD, respectively. Also wetlands planted with *Phalaris* were the most efficient at removing nitrate, likely as a function of higher carbon exudation and lower oxygen availability compared to other plants. Because pollutant removal in CWs is typically equal or higher in summer, our results confirm the efficiency of CWs for year-round treatment of hydroponics wastewater.

As in many other parts of the world, there is no unique effluent standard for hydroponics wastewater in the province of Quebec (Canada), each discharge requirement being determined individually based on the capacity of the receiving environment. Our experiment showed that pollutant removal of hydroponics wastewater by HSSF CWs remains important even under winter conditions. Nevertheless, phosphate concentrations at the CWs' outflow were still too high to reach most performance goals, indicating that other phosphorus removal systems must be considered. CWs greatly reduced nitrate discharge, with a mean of 50 mg NO_3-N L^{-1} at the outflow of CWs planted with *Phalaris* and supplemented with carbon. Increasing the C/N ratio could be an efficient way to further enhance nitrate removal if needed, as carbon seems to remain limiting in our experiment.

Although interesting from a mechanistic perspective, the addition of sucrose (refined sugar) to wastewaters cannot be recommended in full-scale HSSF CWs because the energy input is too high and contradicts the extensive nature of CWs. As the addition of carbon had a significant beneficial impact, further research should focus on investigating the effects of other carbon sources whose production is less energy-intensive than sucrose.

Chapitre 3: Effect of plant species on water quality at the outlet of a sludge treatment wetland

V. Gagnon, F. Chazarenc, M. Kõiv and J. Brisson

3.1 Abstract

Sludge treatment wetlands are mainly used to reduce the volume of activated sludge, and the pollutants at the outlet are generally returned to the wastewater treatment plant. However, in cases where sludges are produced far from treatment plants not only must the sludge be treated, but the discharge of pollutants into the surrounding environment must also be limited. The aim of this study was to evaluate the efficiency of different plant species in optimising pollutant removal in a decentralised sludge treatment wetland. In addition, a new system design was assessed, in which the wetland was not completely drained, and a saturated layer was created using an overflow. The experimental setup consisted of 16 mesocosms in total, planted with monocultures of *Phragmites australis*, *Typha angustifolia* and *Scirpus fluviatilis*, and unplanted controls, each in four replicates.

The experiment was conducted during the third summer of operation after setup. The system was fed with highly concentrated fish farm sludge at a load of 30 kg of total solids m^{-2} yr^{-1}. Results showed that such wetlands were highly efficient, with removal rates between 94% and 99% for most pollutants. Planted systems generally outperformed the unplanted control, with a significantly lower mass of pollutants at the outlet of the sludge treatment wetland planted with *Phragmites*, followed by those with *Typha* and then *Scirpus*. The distinct influence of plant species on pollution removal was explained by the sequestration of nitrogen and phosphorus in plant tissues and by the rhizosphere effect, which enhance the biodegradation of organic matter, allowed the nitrification process and created redox conditions favourable to the sorption of phosphorus. Filtration and

evapotranspiration rates played a major role in limiting the discharge of pollutants, and the impact was enhanced by the fact that the sludge treatment wetland was not completely drained.

3.2 Introduction

A sludge treatment wetland (STW) is a specialised type of vertical flow constructed wetland whose main function is to reduce sludge volume by dewatering and lowering organic matter content through mineralisation (Kim and Smith, 1997). Treatment processes include physical retention of sludge particles at the surface, and percolation of a fraction of the sludge water through the wetland's granular media (Nielsen, 2003). While pollutant concentrations are generally considered low at the outlet of an STW, compared to the extremely high concentrations present in the sludge (Uggetti *et al.*, 2010), they generally remain too important to permit discharge directly into the environment (DeMaeseneer, 1997; Cofie *et al.*, 2006). Yet, the water quality at the outlet of such systems is usually not an issue, because STWs are mainly used for dewatering activated sludge and the outflow is simply pumped back to the wastewater treatment plant (Uggetti *et al.*, 2010). However, in cases where sludges are produced far from treatment plants, such as on farms and by some industry, STWs must not only treat the sludge, but also limit pollutant discharge into the surrounding environment. It is therefore essential to better understand the processes that could increase water quality in a decentralised STWs – an aspect that has rarely been studied and could lead to the optimisation of this technology.

The presence of plants, and the choice of appropriate plant species, has been shown to generally improve pollutant removal from conventional constructed wetlands (Tanner, 2001; Akratos and Tsihrintzis, 2007; De Feo, 2007; Brisson and Chazarenc, 2009; Stefanakis and Tsihrintzis, 2012b) and this seems an important avenue to explore for optimising STWs efficiency. However, the concentration of pollutants in sludge can be up to 100 times greater than in raw municipal wastewater, and consequently it remains to be proven that plants can have a significant effect on pollutant removal in STWs. Studies by Hofmann (1990) and Liénard *et al.* (1995) have shown that the presence of plants lowers the concentration of total suspended solids (TSS) and chemical oxygen demand (COD) at the STWs outlet. Planted STW's were also shown to reduce the concentration of ammonia-nitrogen (NH_4-N) and orthophosphate (PO_4-P) (Stefanakis and Tsihrintzis, 2012a). Nonetheless, a study by Edwards *et al.* (2001), did not measure any differences in NH_4-N or PO_4-P concentration between planted and unplanted STWs, while an even higher concentration of TSS and biochemical oxygen demand (BOD) were detected at the outlet of the planted system. However, evaluating efficiency by assessing pollutant concentration is not ideal, since rain and evapotranspiration can alter pollutants concentration. Thus, comparative treatment efficiency should ideally be assessed on the basis of mass balance (Tanner, 2001). A study by Wang *et al.* (2009) used mass balance analysis to evaluate the treatment of sludge at a relatively low loading rate (mean of 16 kg of TS m^{-2} yr^{-1}). They found that TSS mass and total Kjeldahl nitrogen (TKN) were lower at the outlet of the planted system compared to the unplanted control. Similar findings were reached by Gustavsson and Engwall (2012) at a high loading rate (mean of 122 kg of TS m^{-2} yr^{-1}), for which a mass balance analysis showed that the planted STWs microcosms were more efficient in removing pollutants, including TSS, total volatile solids (TVS), COD, total organic carbon (TOC), BOD, total nitrogen (TN) and total phosphorus (TP).

Identifying the plant species most effective for pollutant removal could also be important for optimising the STW. However, to the best of our knowledge, full-scale STWs are almost exclusively planted with *Phragmites australis* (Hardej and Ozimek, 2002), a species considered invasive in North America and restricted or prohibited in certain regions for use in constructed wetlands (Vymazal and Kröpfelová, 2008). While other plant species have sometimes been used (*e.g.*: *Typha* sp., *Echinochloa pyramidalis*), comparative studies between plant species have rarely been performed. To date, only Wang *et al.* (2009) have shown that plant species has a specific effect on TKN removal, resulting in lower nitrogen mass at the outlet of the STWs planted with *Phragmites australis* and *Typha latifolia*, compared to one planted with *Iris pseudacorus*. Nevertheless, it remains to be determined whether the positive effect of different species is maintained when the STWs is fed at a higher solids loading rate.

Keeping the STWs in a partly saturated condition by installing an overflow rather than the usual completely drained system could also be a way to favour removal of pollutants (Panuvatvanich *et al.*, 2009). Hence, the STWs could act as a compact hybrid constructed wetland (Kantawanichkul and Somprasert, 2005) by combining a vertical flow in the upper portion (good nitrification) and a horizontal flow in the lower portion of the wetland (good denitrification). By not completely draining the system, the water lost through plant transpiration could also play an important role by increasing hydraulic retention time (HRT) and limiting the volume of pollutants at the wetland outlet (Katsenovich *et al.*, 2009). Finally, the presence of a saturated water layer in the wetland could limit the extreme drought conditions that can kill plants during hot, dry weather (Burgoon *et al.*, 1997).

The aim of this study was to evaluate the influence of plant presence as well as the performance of different plant species on the volume of water and the mass of pollutants discharged at the outlet of a decentralised sludge treatment wetland. In evaluating plant effects, we placed particular emphasis on evapotranspiration and nutrient uptake. Three macrophyte species, *Phragmites australis*, *Typha angustifolia* and *Scirpus fluviatilis*, were tested and compared to an unplanted control. The experimental STWs were not completely drained, and a saturated layer was retained at the wetland bottom. The experimental systems were fed with highly concentrated fish farm sludge and pollutant removal was evaluated by mass balance analysis.

3.3 Methods

3.3.1 Experimental design

The experiment was conducted at the Montreal Botanical Garden (Quebec, Canada), which has a semi-continental climate, with a warm, humid summer and a very cold winter. The mean monthly temperature reaches a maximum of 20.9°C in July and a minimum of -10.2°C in January. Average annual precipitation is 978.9 mm (22.2% as snow), and the growing season lasts for about 195 days, from mid-April to mid-September (Environment Canada, climate normals 1971-2000). The experimental setup consisted of 16 mesocosms (cylindrical shape, height: 1 m; diameter: 0.6 m) representing sludge treatment wetlands, each composed of four filter layers of different granular sizes (see Figure 3.1 for details). Contrary to conventional STWs, the experimental mesocosms were not completely drained, and a saturated layer was retained by placing an overflow at 25 cm from the bottom. All water coming out from the overflow was recovered in an outlet bucket for sampling. Mesocosms were planted with a monoculture of *Phragmites australis*, *Typha angustifolia, Scirpus fluviatilis*; there were also unplanted controls, each in four replicates. Two types of fish farm sludge were applied (Table 3.1), so that a duplicate of each condition, species and sludge type was obtained. A randomized block design was used for distributing the plant species and types of sludge among the mesocosms. Plants were already well-established when measurement began, since the experimental set-up was in its third summer of operation.

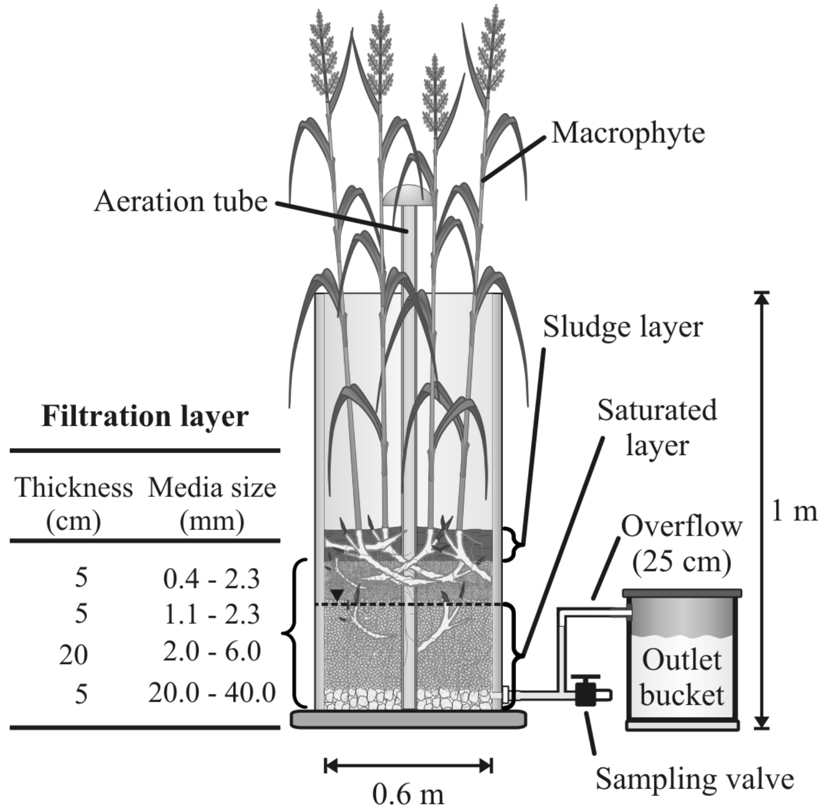


Figure 3.1. Cross-section of the mesocosm, with the detailed granular size of each filtration layer.

3.3.2 Pollutant characteristics and loading rate

The STWs were fed with fish farm sludge (settled fish manure), which is composed mainly of fish faeces and uneaten food (Naylor *et al.*, 1999) and is comparable to septage sludge in terms of pollutant composition and concentration (Troesch *et al.* 2009; Vincent *et al.* 2011), but generally more concentrated than surplus activated sludge (Stefanakis *et al.* 2009; Troesch *et al.* 2009). Two types of fish farm sludge, differing in origin, were used in this study. In earth ponds, a small fraction of clay from the pond lining is mixed in with the sludge, while in concrete ponds (raceways) no clay is present in the sludge. The presence of fine particles such as clay may lead to the clogging of the STW, since grain size distribution of mineral substrates is one of the main parameters influencing clogging in vertical flow constructed wetlands (Hua *et al.*, 2010). In this experiment, we used decanted fish farm sludge from a concrete pond (hereafter referred to as "sludge no clay"), and we added 77 g of dried clay per litre of sludge to simulate earth pond sludge (hereafter referred to as "sludge with clay"). Mesocosms were loaded according to the total solid contained in the sludge with no clay. Prior to the experiment, mesocosms were fed with fish farm sludge over the course of two summers at a load of 20 kg of total solids (TS) m^{-2} yr^{-1}. During the experiment, conducted over 14 weeks, from June to mid-September 2010, mesocosms were fed intermittently (1 day of feeding followed by six days of rest), with 30 kg of TS m^{-2} yr^{-1}. The two types of sludge had very similar pollutant characteristics (Table 3.1), with the exception of a higher amount of total solids in sludge with clay (TS = 5%) than in sludge without clay (TS = 3%). During extreme drought, tap water was added to the bottom of the wetland to prevent plant mortality, which represent up to 35% of the total water input for some *Phragmite* STWs.

Table 3.1. Average concentration of pollutant of fish farm sludge

Pollutants in sludge	**Average in mg/L (± SD)**
Total solid (TS)	28 500 ± 16 400
Total solid with added clay (TS)	50 500 ± 13 600
Total volatile solid (TVS)	20 500 ± 11 100
Chemical oxygen demand (COD)	68 000 ± 26 500
Total Kjeldahl nitrogen (TKN)	1 800 ± 600
Ammonia (NH_4-N)	470 ± 330
Nitrate (NO_3-N)	44 ± 33
Total phosphorus (TP)	820 ± 300
Phosphate (PO_4-P)	120 ± 40

3.3.3 Sampling

Water samples were taken from two locations: 1) the saturated layer of the STW and 2) the bucket at the system outlet (Figure 3.1). Sampling the saturated layer allowed us to measure variations in pollutant mass associated with each plant species throughout the summer. The sum of pollutants flowing sporadically from the outlet allowed us to measure the effect of plants on the total mass of pollutants discharged from the STWs during the experiment.

The saturated layer was sampled by opening a valve situated at the base of each mesocosm (Figure 3.1). A volume of 300 mL was collected weekly, just prior to the application of a new batch of sludge. The sample was stored at 4°C and analysis of pollutants was performed within a day of sampling, with the exception of total phosphorus (TP) and total Kjeldahl nitrogen (TKN), for which a subsample was kept frozen at -20°C until analysis. Water volume in the saturated layer was also estimated prior to each sampling using a calibrated water buoy installed in each STW; the volume was then used to calculate the mass of pollutants at sampling time.

The presence of water in the bucket at the outlet was verified each day (Figure 3.1). If water was present, the volume was measured and then transferred to a container stored at -20°C. At the end of each week, the container was thawed and mixed for pollutant analysis, with the exception of one sample kept frozen at -20°C and subsequently analysed for total phosphorus (TP) and total Kjeldahl nitrogen (TKN). Total volume discharged from the outlet for the week was used to calculate the mass of pollutants for this period.

It must be noted that the difference in sample storage between the water from the saturated layer (4°C) and the outflow (-20°C) may in certain cases have affected the concentration of ammonia, nitrate and phosphate of the frozen samples. Thus, comparison and interpretation of the two sampling sites must be performed with care.

3.3.4 Measurement and analysis

At the end of the summer, above-ground plant biomass was cut, dried and weighed. The measured weight of above-ground biomass was then divided by the surface area of the mesocosms and expressed in g m^{-2}. Below-ground biomass was assessed for only one replicate (species and sludge type), for a total of six mesocosms. Half of the volume of each mesocosm was excavated and the rhizome and roots were collected, dried and weighed. The measured mass was then divided by the excavated surface of the mesocosms and expressed in g m^{-2}.

Plant uptake of nitrogen and phosphorus was estimated by multiplying dry biomass (above- and below-ground) by the specific ratio of nutrients per dry biomass according to values determined by Tanner *et al.* (1995), Ennabili *et al.* (1998) and Smith *et al.* (2008). Since no phosphorus ratio was found for below-ground biomass of *Scirpus fluviatilis*, only above-ground phosphorus content is presented.

Water loss through evapotranspiration (ET) was calculated weekly by measuring total inlet volume, the variation of volume inside the mesocosms and total outlet volume (Equation 1).

$$ET = V_{In} - ((V_{d7} - V_{d1}) + V_{out}) \tag{1}$$

V_{In} = Inlet volume (Sludge + rain volume for the week)

V_{d1} = Volume inside the mesocosm, day 1

V_{d7} = Volume inside the mesocosm, day 7

V_{out} = Volume collected from the outlet for the week

Water on the sand and gravel media of the drained portion of the STW was not included in the ET calculation, since it was considered to be negligible by Stefanakis and Tsihrintzis (2011). Water retained in the sludge cake was also not included in the ET analysis, since it represents only a small fraction (1-3%) of the total water input to the STWs (data not shown). The calculated evapotranspiration was divided by the surface of the mesocosm and by the seven days of the week. Therefore, ET is expressed as L m^{-2} d^{-1}.

Analyses of the following pollutants were made according to Standard Methods (APHA, 1998): chemical oxygen demand (COD); orthophosphate (PO_4-P); total suspended solids (TSS) and total volatile suspended solids (TVSS). Ammonia-nitrogen (NH_4-N) and nitrate-nitrogen (NO_3-N) were measured with a selective ion electrode (Cole-Parmer, model: SI-27502-00 and CO-27504-22). Total Kjeldahl nitrogen (TKN) and total phosphorus (TP) were measured using a Quikchem automated flow injection analyser according to the manufacturer's instructions (Quikchem 8500, Lachat). The mass balance was calculated by multiplying the pollutant concentration by volume and dividing by the surface area of the mesocosms; therefore pollutants were expressed as g m^{-2}.

3.3.5 Statistical analysis

We used a two-way analysis of variance (ANOVA) followed by a test of multiple comparisons of means (Tukey HSD) to evaluate the effects of plant species and clay addition on pollutant removal. Furthermore, we used this statistical analysis to test the difference according to plant biomass, evapotranspiration, and outlet volume. The statistical results are presented in brackets in the results section and represent all the factors tested (ex.: plants species and unplanted control). In addition, statistical results are also presented as a column of capital letters below the figures, where each different letter corresponds to a statistical difference between plant species for a specific week. Variables that did not meet normality or heterogeneity assumptions were modified using the appropriate transformation (ln, square root or Box-Cox). However, ANOVA was not performed for most of the NO_3-N measurements, because no data transformation could adequately reduce the heterogeneity of the variances. Addition of clay to the sludge did not significantly influence pollutant removal or plant characteristics, and thus the results are presented without a distinction between sludge types, except for TP and PO_4, for which significant differences were measured. Statistical tests were considered significant at the 0.05 level and were performed with SPSS 16.0 statistical software (IBM Corporation).

3.4 Results

3.4.1 Plant parameters

Plant above-ground biomass was significantly different between species ($F_{2,12} = 63.73$, $p < 0.01$), with a higher value for *Phragmites*, followed by *Typha* and then *Scirpus* (Table 3.2). Below-ground biomass followed the same trend, although no statistical analysis was performed since only one sample (no replicate) was measured (Table 3.2). During the experiment, plants were generally healthy, with the exception of *Scirpus*, which did not grow well in sludge and generally had yellowed leaves.

Table 3.2. Average plant dry biomass and water loss by evapotranspiration according to plant species for the summer 2010. The letters presented beside the values represent statistical differences.

		Phragmites	*Typha*	*Scirpus*	Unplanted
Dry biomass	Above-ground	2 669^a	1 103^b	110^c	-
(g m^{-2})	Below-ground	4 369	2 781	907	-
Water loss by	Maximum	20.4	12.3	11.2	9.1
evapotranspiration	Average	10.3 ^{a}	5.9 ^{b}	3.3 ^{c}	2.8 ^{c}
(L m^{-2} d^{-1})	Minimum	1.7	0.9	0.0	0.0

Average evapotranspiration rate was significantly different between plant species ($F_{3,12} = 51.74$, $p < 0.01$), with a higher value for *Phragmites*, followed by *Typha* and finally *Scirpus*, which was not significantly different from the unplanted control (Table 3.2). A positive and linear correlation ($R^2 = 0.97$) was found between the volume of water lost by evapotranspiration and dry above-ground biomass (Figure 3.2). Consequently, total outflow volume from each mesocosm was also significantly different between plant species ($F_{3,16} = 40.18$, $p < 0.01$). Greatest volume reduction was measured at the

Phragmites outlet (98%), followed by *Typha* (83%) and then *Scirpus* (56%), which was similar to the unplanted control (52%; Table 3.3). However, the correlation between outlet volume and above-ground biomass was best explained by a negative exponential correlation (R^2 = 0.93; Figure 3.2). In other words, the volume present at the mesocosm outlet diminished exponentially with the increase of above-ground biomass.

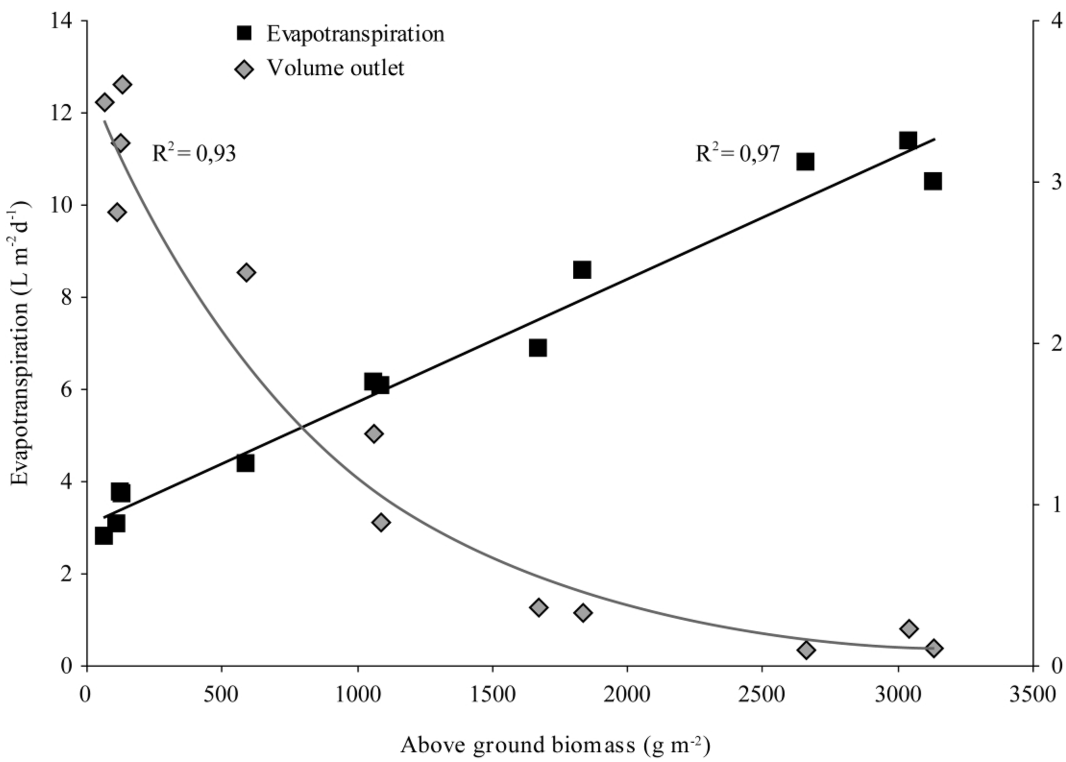


Figure 3.2. Correlations between water loss by evapotranspiration and above-ground dry biomass of all mesocosms (black squares); correlations between total volume at the outlet and above-ground dry biomass of all mesocosms (gray lozenges).

Nitrogen retention in the plant biomass at the end of the summer period was estimated at 131 g N m^{-2} for *Phragmites*, representing 26% of the total input of TKN by sludge; *Typha* had 61 g N m^{-2} with 12% and *Scirpus* 9 g N m^{-2} with only 1.9%. Phosphorus retention by plant biomass was estimated at 14 g P m^{-2} for *Phragmites*, representing 5.7% of the input of TP by sludge; 8 g P m^{-2} for *Typha* with 3.4% and finally 0.5 g P m^{-2} for *Scirpus* with 0.2% of the phosphorus input sequester in their biomass.

3.4.2 Pollutants in the saturated layer and at the STWs outlet

3.4.2.1 Total suspended solids and volatile solids

Removal of solids by the STWs was extremely efficient, with a reduction of 99% of TSS and TVSS at the outlet of both planted and unplanted mesocosms. Nonetheless, significant differences were observed between plant species (Table 3.3), with the lowest TSS and TVSS at the *Phragmites* and *Typha* outlets, followed by *Scirpus* and finally the unplanted control (TSS: $F_{3,12} = 86.27$, $p < 0.01$; TVSS: $F_{3,12} = 198.10$, $p < 0.01$).

Throughout the summer, the mass of TSS and TVSS in the saturated layer of the mesocosms was generally lower in the planted systems compared to the unplanted control (Figure 3.3 a-b). Significant differences between plant species were mainly observed at the beginning of the summer, with lower TSS in mesocosms planted with *Phragmites*, while *Typha* and *Scirpus* usually had values midway between *Phragmites* and the unplanted control. From the middle of the summer on, TSS and TVSS were low and there was generally no difference between plant species. The drop in TSS and TVSS on June 16 was caused by the emptying of the treatment wetland for the installation of a microbial sampling instrument. No data measurements were taken on August 4, due to a heat wave (Figure 3.3 a-b).

3.4.2.2 Chemical oxygen demand

The STWs were also highly efficient in removing COD, with a removal rate of 99% at the outlets of both planted and unplanted control mesocosms. Significant differences between plant species were observed, with the lowest mass of organic matter (g COD m^{-2}) at the *Phragmites* and *Typha* outlets, followed by *Scirpus* and finally the unplanted control ($F_{3,12} = 123.38$, $p < 0.01$).

Organic matter mass in the saturated layer of the mesocosms was significantly lower in the planted systems compared to the unplanted control (Figure 3.3 c). Furthermore, COD value varied between plant species depending on the period of the summer. Early in the summer, there was generally no statistical difference between species, however in mid-summer; *Phragmites* generally had a lower value than the *Typha* and *Scirpus* STW. The difference between species generally subsided at the end of the summer.

Table 3.3. Sum of the volume and mass of pollutants (g m^{-2}) at the inlet and outlet of the mesocosms according to plant species and unplanted control after vegetation period (total 3.5 months). The letters beside the values represent statistical differences.

	Species	Volume* (L/m²)	TSS** (g/m²)	TVSS (g/m²)	COD (g/m²)	TKN (g/m²)	NH_4-N (g/m²)	NO_3-N (g/m²)
Inlet	Sludge	900	8 000	5 900	19 700	500	135	15
Outlet	*Phragmites*	20^{a}	1.0^{a}	0.5^{a}	2.7^{a}	0.2^{a}	0.03^{a}	0.3^{a}
	Typha	140^{b}	6.1^{a}	2.4^{a}	21.2^{a}	1.2^{b}	0.3^{b}	1.5^{ab}
	Scirpus	370^{c}	16.5^{b}	5.0^{b}	54.8^{b}	3.7^{c}	1.7^{c}	4.8^{b}
	Control	400^{c}	38.6^{c}	16.0^{c}	175.5^{c}	24.0^{d}	24.0^{d}	0.9^{ab}

* Volume of sludge, precipitation and the average volume of fresh water added during drought condition

** TSS of the sludge without clay

3.4.2.3 Nitrogen

Planted STWs removed 99% of TKN and NH_4-N, while the unplanted control removed 95% of TKN and 82% of NH_4-N. Significant differences were observed between plant species (Table 3.3), with the lowest TKN and NH_4-N at the *Phragmites* outlet, followed by *Typha*, then *Scirpus* and finally the unplanted control (TKN: $F_{3,12} = 850.62$, $p < 0.01$; NH_4-N: $F_{3,12} = 84.70$, $p < 0.01$).

Mass of TKN and NH_4-N followed a similar pattern in the saturated layer of the STW, with significantly lower values in the planted mesocosms compared to a gradual accumulation of nitrogen in the unplanted control (Figure 3.3 d-e). Furthermore, the effect of plant species on TKN and NH_4-N was generally similar at first, but began to differentiate over the course of the summer. In Figures 3 d-e, we can see that, on average, *Phragmites* was significantly more efficient at removing TKN and NH_4-N in the saturated layer compared to other species. In the middle of the summer, for three consecutive weeks, the mass of NH_4-N was statistically different between all plant species, with a lower value for *Phragmites*, followed by *Typha* and then *Scirpus*. At the end of the summer, TKN and NH_4-N values were most similar between *Phragmites* and *Typha*.

Removal of NO_3-N differed greatly between plant species, with a reduction of 98% at the *Phragmites* outlet, 94% for the unplanted control, 89% for *Typha* and finally 65% for *Scirpus*. Significant differences were measured between plant species with lower NO_3-N at the outlets of mesocosms planted with *Phragmites*, while *Typha* and the unplanted control were midway between *Phragmites* and *Scirpus* (NO_3: $F_{3,12} = 5.30$, $p < 0.01$) (Table 3.3).

Mass of NO_3-N in the saturated layer was generally impossible to analyse statistically, due to the high heterogeneity variance; therefore, analysis was performed qualitatively (Figure 3.3 f). Higher mass of NO_3-N was present in mesocosms planted with *Scirpus* and *Typha* compared to *Phragmites* and the unplanted control. A high peak in NO_3-N was observed in the middle of the summer (July 21) for *Scirpus* and *Typha*, and to a lesser extent for *Phragmites* and the unplanted control. Low NO_3-N levels were observed at the end of the summer for all treatments.

3.4.2.4 Phosphorus

Removal of TP and PO_4-P varied from 94% to 99%, depending on the plant species and the type of sludge. The presence of clay in the sludge significantly reduced the amount of TP and PO_4-P at all mesocosm outlets (TP: $F_{1,8} = 5.35$, $p < 0.05$; PO_4-P $F_{1,8} = 5.07$, $p < 0.05$; Table 3.4). Furthermore, significantly lower TP and PO_4-P were measured at the outlets of mesocosms planted with *Phragmites* compared to the other species and the unplanted control (TP: $F_{3,8} = 26.12$, $p < 0.01$; PO_4-P: $F_{3,8} = 24.56$, $p < 0.01$).

In the saturated layer of the mesocosms, there were significantly lower amounts of TP and PO_4-P when clay was added to the sludge. Furthermore, the added clay only affected the relative amount of TP and PO_4-P, but the influence of plants remained similar, regardless of the type of sludge used. No statistical differences were observed between planted and unplanted mesocosms for the first part of the summer. However, from the middle to the end of the summer, *Phragmites* usually had significantly lower TP and PO_4-P, while *Typha* had values in between *Phragmites* and those of *Scirpus* and the unplanted control. Since the influence of plants on TP in the saturated layer was similar to findings for PO_4-P, only the latter is presented on Figure 3.4.

Table 3.4. Mass of total phosphorus and phosphate per square meter at the inflow and outlet of the mesocosms according to plant species and unplanted control. The letters presented beside the values represent statistical differences

	Species	TP (no Clay) (g/m^2)	TP (Clay) (g/m^2)	PO_4-P (no Clay) (g/m^2)	PO_4-P (Clay) (g/m^2)
Inlet	Sludge	230	230	32	30
Outlet	*Phragmites*	0.1^a	0.2^a	0.04^a	0.04^a
	Typha	3.7^b	0.8^b	1.00^b	0.21^b
	Scirpus	6.0^b	3.5^b	1.70^b	0.90^b
	Unplanted	7.5^b	4.4^b	1.83^b	1.27^b

3.5 Discussion

3.5.1 Overall pollutant removal

Excellent treatment capacities were observed at the outlet of the sludge treatment wetlands, with removal ranging from 94% to 99% for all pollutants, except for ammonia (82% to 99%) and nitrate (65% to 98%). These results are consistent with other studies that have shown removal efficiency superior to 90% at the outlet of sludge treatment wetlands (Burgoon *et al.*, 1997; Begg *et al.*, 2001; Wang *et al.*, 2009). However, due to the extremely high pollutant content of the sludge, even a removal rate of 90% can result in an outflow with a relatively high amount of pollutants. Consequently, enhancing pollutant removal by a percent or a fraction of a percent is not trivial and can significantly limit the quantity of pollutants discharged from the STW.

3.5.2 Plant parameter and water volume at the outlet

Water loss by plant transpiration, linked to plant-specific morphology, was an important factor influencing the volume of water and mass of pollutants at the outlet of the STW. A positive and linear correlation was found between evapotranspiration rate and above-ground dry biomass, as has been shown in other studies (Mueller *et al.*, 2005; Stefanakis and Tsihrintzis, 2011; Korboulewsky *et al.*, 2012). In contrast, the volume discharged at the outlet of the system diminishes exponentially with biomass size and not linearly, as would be expected based on evapotranspiration alone (Figure 3.2). Our new STWs design could explain this finding, since the presence of an overflow created a threshold such that outflow occurred less often with species exhibiting a high evapotranspiration rate. Panuvatvanich *et al.* (2009) have shown that ET was enhanced when water was retained in the saturated layer of the STW: water loss increased from less than 10% the first day to more than 40% after 6 days of impounding. In addition, evapotranspiration could also have favoured pollutant removal by increasing the HRT (Faulwetter *et al.*, 2009), which is discussed in more detail in the pollutant section.

Therefore, the choice of plant species is crucial for the optimal functioning of this new STWs design. In our experiment, *Phragmites* had highest evapotranspiration rate and were significantly more effective at reducing outlet volume, followed by *Typha* and finally *Scirpus* and the unplanted control. However, mesocosms planted with *Scirpus* did not differ from the unplanted control in terms of evapotranspiration and outlet volume reduction, since these plants did not grow well in the sludge layer. The grass-like morphology of *Scirpus* was not well-adapted to the sludge treatment, since the leaves were easily soiled during the feeding process, which led to the generally poor health of the plants. On the other hand, the high evapotranspiration rate of *Phragmites*, with a maximum of 20 L m^{-2} d^{-1}, can also be problematic, since it may create severe drought conditions within the system that can eventually be fatal to the plants (Burgoon *et al.*, 1997). Even with an overflow level at 25 cm from the bottom, the saturated layer of water was sometimes insufficient for preventing drought in hot, dry weather in the *Phragmites* bed. Drought could have resulted from the relatively low volume of sludge loaded on our system, and the small size of the mesocosm could have promoted higher evapotranspiration via "oasis" and "clothesline" effects (Allen *et al.*, 2011). In a full-size system, evapotranspiration rate is expected to be lower and drought should not be as frequent or severe. Nonetheless, the average evapotranspiration of *Phragmites* (10.3 L m^{-2} d^{-1}) in our system was in the range reported by Borin *et al.* (2011) for *Phragmites* in full-size constructed wetland.

3.5.3 Removal of solids

The highly efficient pollutant removal from the sludge treatment wetlands, planted and unplanted, can be explained by the effective physical filtration of the systems. In fact, pollutants in the fish farm sludge were mainly in particulate form (Naylor *et al.*, 1999) and were therefore retained efficiently at the surface of the STWs by the filtering media. In addition, the presence of plants significantly enhanced TSS and TVSS removal, with lower values in the saturated layer and at the outlet of the planted systems. The beneficial influence of plants on the retention of solids could be explained by the added filtration created by the network of dense roots and rootlets present in the STWs (Zurita *et al.*, 2009). The positive effect of plants on TSS removal was also observed by Wang *et al.* (2009) at a loading rate of 16 kg of TS m^{-2} yr^{-1} and by Gustavsson and Engwall (2012) at a

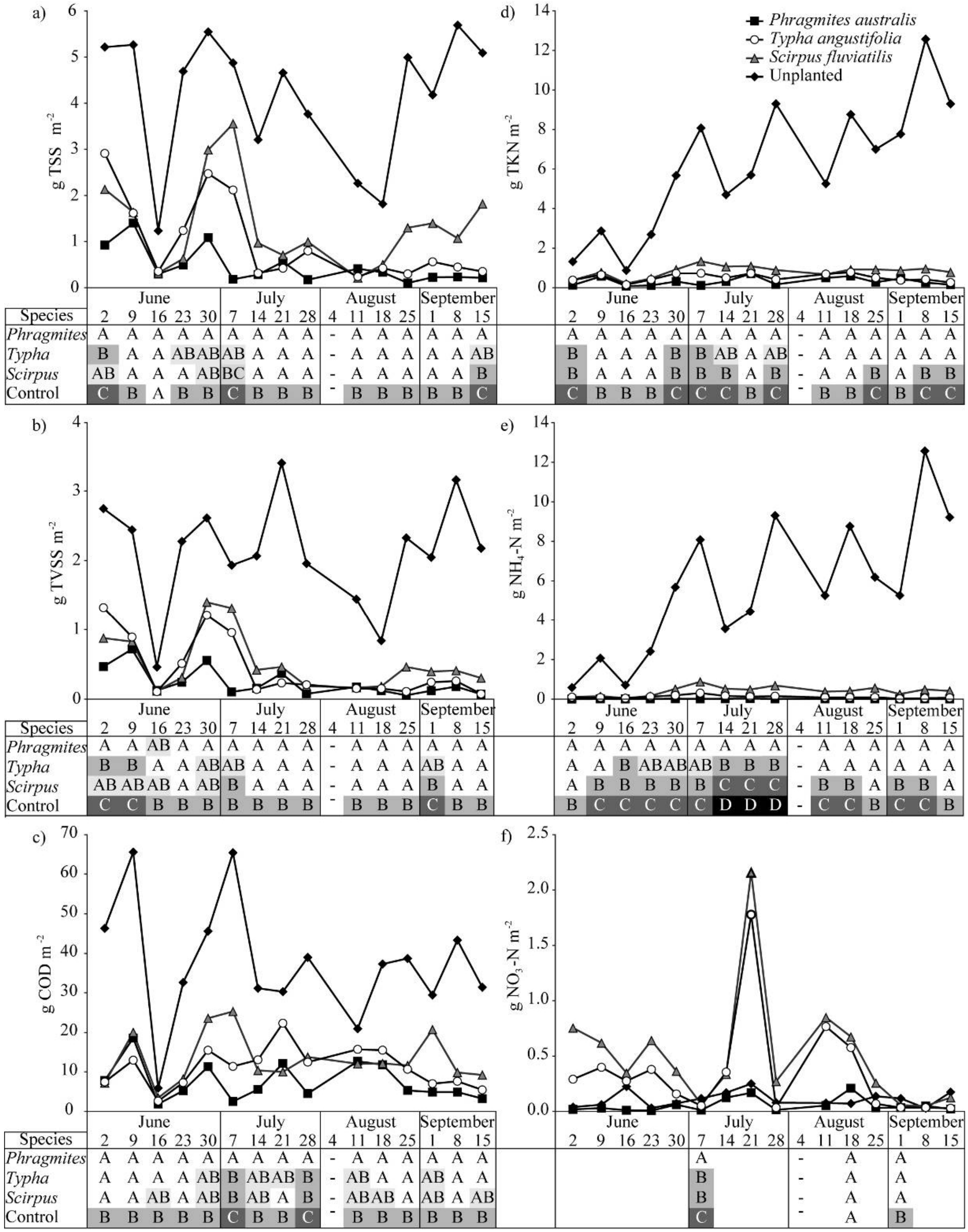


Figure 3.3. Mass of pollutants in the saturated layer of mesocosms according to plant species and time: a) Total suspended solids, b) Total volatile suspended solids, c) Chemical oxygen demand, d) Total Kjeldahl nitrogen, e) Ammonia nitrogen, f) Nitrate nitrogen. Statistical differences in pollutant mass between plant species for each week (table columns) are presented as different color-coded letters in the tables below the graphics.

loading rate of 122 kg of TS m^{-2} yr^{-1}, thus showing that plants remain effective at removing solids even when the STWs is highly loaded.

The amounts of TSS and TVSS also varied according to plant species, and differed between the saturated layer and the outlet of the STW. In the saturated layer, there were generally no differences between plant species, except at the beginning of the summer, when variations could be attributed to plant specific establishment and root development. In contrast, significant differences between plant species were measured at the outlet of the STW. These results differ from those reported by Wang *et al.* (2009), who did not measure any effect of plant species on TSS removal at the outlet of a totally drained STW. Our new STWs design could explain this difference in findings because the presence of a saturated layer favours water loss by evapotranspiration, thus limiting the volume of water discharged and consequently the mass of TSS at the outflow.

3.5.4 Removal of organic matter

Removal of organic matter, measured by COD, was highly efficient in the STWs and differs according to plant species. This reduction can be partly explained by the combination of physical filtration and reduction of the outflow volume, as seen previously, but can also be the result of biodegradation of the organic matter. In fact, levels of COD in the saturated layer were always lower in the planted system compared to the unplanted control. This could be the result of plant oxygen transfer to the rhizosphere, which may have enhanced microbial degradation of the organic matter (Nielsen, 2003; Gagnon *et al.*, 2007; Huang *et al.*, 2010). In addition, greater differences between plant species were observed in the middle of the summer, with lower COD values in systems planted with *Phragmites*. This summertime variation of COD could be linked to the plants' establishment, but also to the distinct oxygen transfer rate of each species over the course of the season.

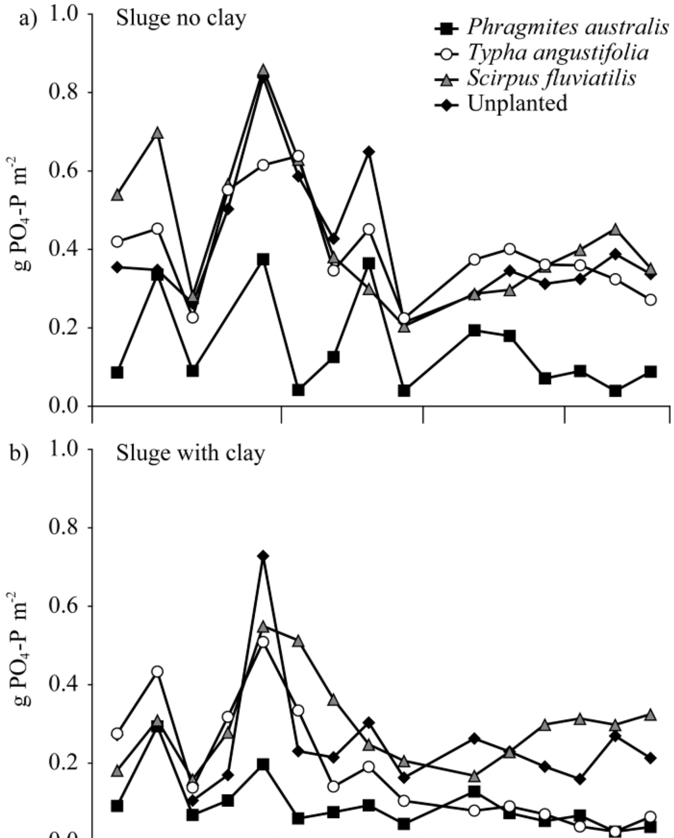
3.5.5 Removal of nitrogen

Nitrogen removal was greatly enhanced by the presence of plants, and differed according to species. Higher TKN and NH_4 removal was observed in wetlands planted with *Phragmites*, followed by *Typha* and finally *Scirpus*. In comparison, a study by Wang *et al.* (2009), in a completely drained STWs showed that *Phragmites* and *Typha* had similar TKN removal rates, while that of STWs planted with Iris was lower. The greater difference between plant species in our study may have been the result of the new STWs design, with the presence of a saturated layer enabling a longer contact time between nitrogen sources and plant rhizosphere, thus enhancing plant species' particular influence on the nitrification process and nutrient uptake. This positive effect of plants was also observed in the saturated layer, where low, stable values of TKN and NH_4 were measured, while a gradual accumulation of nitrogen occurred in the unplanted control. Furthermore, the influence of plant species varied with time, as shown by the different levels of TKN and NH_4 in mid-summer, when plants were at the peak of maturity. This could have been the result of the enhanced nitrification caused by plant oxygen transfer to the rhizosphere (Munch *et al.*, 2005), resulting in a lower NH_4 level in planted systems. In addition, the above-ground biomass has been shown to positively enhance oxygen transfer to the rhizosphere and nitrification process (Wiessner *et al.*, 2002), which accounts for the difference between plant species.

Nitrate level was high in *Typha* and *Scirpus*, but low for the unplanted control, confirming that nitrification was enhanced in the planted system. However, the saturated layer of the *Phragmites* system had low levels of both ammonia and nitrate. This can be partly explained by the uptake of nitrate and ammonia by *Phragmites*, whose biomass was estimated to account for up to a quarter (26%) of TKN input by the sludge, while *Typha* accounted for 12% and *Scirpus* for only 1.9%. However, since the experiment was conducted during the plants' growing season, we expect the fraction of nitrogen sequestered by the plant to be lower on a year-round basis. Nonetheless, similar results were found by Korboulewsky *et al.* (2012) using a lower load of nitrogen with a total of 23% of nitrogen input by sludge in *Phragmites* biomass and 15% for *Typha.*

3.5.6 Removal of phosphorus

High phosphorus removal was observed at the STWs outlet, with lower TP and PO_4-P at the *Phragmites* outlet compared to other species and the unplanted control, In contrast, Wang *et al.* (2009) found no comparable difference between plant species or the unplanted control. The significant effect of *Phragmites* in our study could be the result of having a saturated layer, which favoured evapotranspiration and lengthened the HRT, thus allowing more time for phosphorus removal through physical, chemical and biological processes. In the saturated layer of the STW, TP and PO_4-P were significantly lower in *Phragmites* and to a lesser extent in *Typha* from the middle to the end of the summer. These differences could be partly explained by the sequestration of phosphorus in plant biomass, which could account for about 5.7% and 3.4% of TP input by the sludge for *Phragmites* and *Typha* respectively. In comparison, *Scirpus* had a phosphorus uptake representing only about 0.2% of TP input. Removal could also have been caused by the sequestration of phosphorus in the microbial biomass (Gagnon *et al.*, 2010), which may have been favoured by the positive effect of the *Phragmites* rhizosphere on microbial density. Furthermore, the combination of plant-mediated oxygen transfer and water level fluctuations caused by high evapotranspiration could have enhanced redox-mediated phosphorus removal in *Phragmites* mesocosms. It has been suggested that elevated redox potential and humic root exudates may increase phosphorus sorption and co-precipitation on elements in the rhizosphere (Tanner *et al.*, 1995).



	June					July				August				September		
Species	2	9	16	23	30	7	14	21	28	4	11	18	25	1	8	15
Phragmites	A	A	A	A	A	A	A	A	A	-	A	A	A	A	A	A
Typha	A	A	A	A	B	B	AB	AB	B	-	A	A	AB	AB	AB	AB
Scirpus	A	A	A	A	B	B	B	AB	B	-	A	A	B	B	B	B
Unplanted	A	A	A	A	B	B	B	B	B	-	A	A	B	AB	B	B
Clay	A	A	A	A	A	A	A	A	A	-	A	A	A	A	A	A
No Clay	B	A	A	A	B	B	B	B	B	-	A	B	B	B	B	A

Figure 3.4. Mass of phosphate in the saturated layer of mesocosms according to plant species and time: a) mesocosms fed with sludge no clay, b) mesocosms fed with sludge with clay. Statistical differences in mass of PO_4-P between plant species for each week (table columns) are presented as different color-coded letters in the tables below the graphics.

Sludge with a clay content decreased the amount of TP and PO_4-P in the saturated layer and at the outlet of the sludge drying bed, perhaps due to adsorption and precipitation of phosphorus on calcium (Ca), aluminum (Al) and iron (Fe) elements present in the clay particles (Gerritse, 1993). This concurs with findings by Lefrancois *et al.* (2010) that some phosphorus can be sorbed to the clay sediment in a fish farm pond, thus lowering pond phosphorus content.

3.5.7 Sludge treatment wetland design and optimisation

The optimisation of pollutant removal in a sludge treatment wetland could be achieved by leaving a saturated layer (hybrid system), thus favouring water loss by plant transpiration and limiting the volume of water and mass of pollutants discharged at the outlet. However, a comparative study between a hybrid STWs and a completely drained STWs should be performed under the same experimental conditions to validate these results. Further investigation in a full scale system is necessary to identify the optimum balance between sludge loading rate, water loss by evapotranspiration and volume of the saturated layer (overflow height). Higher efficiency would be expected from this proposed design, fed with concentrated sludge (TSS: 20 000-30 000 mg L^{-1}), since the more diluted sludge used would imply higher percolate volume and thus a shorter HRT. Additionally, special attention should be paid to the ammonia concentration in the saturated layer, since ammonia may accumulate and reach concentrations lethal for plants. Therefore, ammonia input by the sludge should not exceed the nitrification capacity of the system. Recirculation of the outflow water to the inlet of the STW may be one strategy for lowering high ammonia concentration.

Selection of the most efficient plant species have been shown to be an important factor for the optimisation of the pollutant removal. In our study, *Phragmites* demonstrated greatest efficiency in pollutant removal. However, due to this species' invasive nature in many parts of the world, it can not be used in all regions (Vymazal and Kröpfelová, 2008). *Typha* could be a good alternative, since it was found to be as efficient as *Phragmites* for TSS, TVSS and COD removal and second best for nitrogen removal (TKN, NH_4, NO_3).

STWs planted with *Phragmites* and *Typha* were extremely efficient in pollutant removal, thus making them suitable for decentralised treatment of sludge. Nonetheless, due to the high evapotranspiration of these systems, the volume of water at the outlet was low and pollutant concentrations at the STWs outlets were moderately high. The dilution factor of the receiving environment should be adequately assessed or an after-treatment should be considered. A zero discharge willow evaporation bed could be an after-treatment option, where approximately 500 l m^{-2} yr^{-1} of wastewater could be treated, depending on regional climate (Gregersen and Brix, 2001).

3.6 Conclusion

- The presence of plants and particularly the choice of plant species were shown to be important for limiting the discharge of pollutants by a STW.
- Filtration, enhanced by the presence of plants, played an important role in pollutant removal, resulting in the retention of solids by the granular matrix.
- Plant transpiration greatly reduces the volume of water and the mass of pollutants at the outlet, an effect that was heightened by the fact that the wetland was not totally drained.
- Planted STWs were generally more efficient than the unplanted control with less pollutants at the *Phragmites* outlet, followed by *Typha* and then *Scirpus*, while the unplanted control usually proved to be the least effective.
- Results suggest that the plant rhizosphere favour the biodegradation of organic matter, allowed the nitrification process and created redox conditions favourable to the sorption of phosphorus.
- Plants sequestered a certain amount of nitrogen and phosphorus in their tissues, with up to a quarter of the nitrogen input by the sludge sequestered in *Phragmites*.

- The influence of plants on pollutants removal varies with time, with generally greater differences between species in mid-summer, an effect possibly related to plant establishment and maturity.
- Additional investigation should be conducted in full scale STWs to assess the influence of the saturated layer and identify the optimum balance between sludge loading rate, water loss by evapotranspiration and volume of the saturated layer.
- Further studies should look at the effect of cold winter climate on the accumulation of frozen sludge at the surface of the STWs and the role of emerging plants on pollutants in spring.

Chapitre 4: Effect of plant species on sludge dewatering and fate of pollutants in sludge treatment wetlands

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4.1 Abstract

Plants are assumed to play a central role in sludge treatment wetlands (STWs) by preventing clogging, favouring dewatering and improving mineralisation of the sludge. However, few comparative studies have been made to assess the influence of different plants or plant species on the treatment of sludge in STWs. Therefore, this study aims to evaluate the effect of three plant species on sludge dewatering and mineralisation, in addition to the general fate of water and pollutants in STWs. Experimental setup consisted of mesocosm sized STWs planted with monocultures of *Phragmites australis*, *Typha angustifolia* and *Scirpus fluviatilis*, in addition to an unplanted control, each in duplicate. The mesocosms were fed with settled fish farm sludge for three summers, and the effect of plants was assessed according to mass balance analysis of the pollutants.

Results revealed that pollutants were mainly retained within the sludge cake, while the rest was trapped inside the STWs or mineralised. Only a very small percentage of pollutants was discharged at the outlet (from <0.1% to 5% of total pollutant input). *Phragmites* had the highest sludge volume reduction and was the most efficient in sludge dewatering and mineralisation. In addition, a fraction of the nitrogen and phosphorus was sequestered in plant tissues, which represented close to a quarter of the nitrogen input by the sludge in *Phragmites* STWs. Nonetheless, sludge cakes of STWs planted with *Typha* and *Scirpus* generally had higher organic matter mass, as well as nitrogen and phosphorus levels, when compared to the unplanted control. This was attributed to the presence of plant litter in the sludge cake.

4.2 Introduction

Sludge treatment wetlands (STWs) are a phytotechnology specialised in the reduction of sludge volume. Plants are thought to play a central role in STWs, by preventing clogging, favouring dewatering and improving mineralisation of the sludge (Nielsen, 2007). They are assumed to enhance dewatering through plant transpiration and by creating drainage tunnels within the sludge layer through the movement of stems and roots (Nielsen, 2003). Furthermore, aeration from the tunnels as well as oxygen transfer from the plant to the rhizosphere are thought to favour microbial processes responsible for the mineralisation of the sludge cake (Uggetti *et al.*, 2010).

Although plants constitute a key element of this technology, few studies have tested the influence of plants or plant species on the dewatering and mineralisation of the sludge. All studies comparing planted to unplanted STWs were done using *Phragmites*, sometimes with contradictory findings. The presence of *Phragmites* has been shown to enhance sludge volume reduction, with 3-8% less volume in planted systems compared to unplanted controls (Edwards *et al.*, 2001; Stefanakis and Tsihrintzis, 2012). *Phragmites* can similarly favour dewatering, with an average of 2-6% more total solids (TS) in the sludge cake compared to unplanted (Edwards *et al.*, 2001; Stefanakis and Tsihrintzis, 2012). However, a study by Liénard *et al.* (1995) measured no TS difference between the sludge cake of planted and unplanted STWs. Sludge mineralisation was higher in planted systems, with 3-6% less volatile solid (TVS) per TS in the sludge cake (Liénard *et al.*, 1995; Stefanakis and Tsihrintzis, 2012), yet one study measured no difference (Edwards *et al.*, 2001). A lower percentage of nutrients has generally been found in the sludge cake of planted STWs, with 1-6% less total Kjeldahl nitrogen (TKN) and 0.3-3.5% less total phosphorus (TP) per TS (Liénard *et al.*, 1995; Stefanakis and Tsihrintzis, 2012). Little attention has been given to the effect of plant species in STWs. To date, the single study comparing the effect of plant species on the dewatering and mineralisation of sludge revealed no significant difference between *Phragmites australis* and *Typha* sp. in terms of volume reduction, chemical oxygen demand (COD), TS, TVS, TKN and TP removal (Uggetti *et al.*, 2012). However, the significance of any effect of plant presence or particular species in STWs is difficult to assess, since these experiments were conducted without replicated units. Variance for each treatment is therefore unknown or, if presented (spatial or temporal sub-sampling of the same STW units), it is usually too large to allow clear interpretation.

The pollutant content of sludge gives only the ratio of pollutants per solids, but not the specific mass of pollutants accumulated within the sludge cake of the STW. Therefore, the effect of plants in STWs could be better assessed using a mass balance analysis, which gives the percentage of water and pollutants retained in the sludge cake, sequestered in the plant, transformed or discharged at the outlet. Water balance analysis of STWs planted with *Phragmites australis* has shown that the larger proportion is eliminated through evapotranspiration (58-84%), most of the rest is discharged at the outflow (13-41%), and only a small fraction is retained in the sludge cake (1-4%) (Begg *et al.*, 2001; Stefanakis and Tsihrintzis, 2011). Water balance analysis of STWs planted with *Typha angustifolia* found a lower percentage of water loss through evapotranspiration (42%), with the remaining water considered discharged at the outlet (58%) (Panuvatvanich *et al.*, 2009). In an STWs planted with *Typha angustifolia*, total solids were retained mainly in the sludge cake (38-52%), with only 11-12% present at the outflow and the rest unaccounted for (36-50%) (Koottatep *et al.*, 2001). Another study found that nitrogen was mainly retained in the sludge cake (55%), a very small portion was sequestered in *Typha angustifolia* tissue (0.2%), and the rest was drained at the outflow (13%) or unaccounted for (13%) (Panuvatvanich *et al.*, 2009). None of these studies conducted comparative analysis of mass balance between plant species under the same experimental conditions. Consequently, the effect of plants species on the fate of water and pollutants in STWs remains inconclusive.

The aim of this work was to evaluate the effect of the presence of plants and specific plant species on sludge dewatering and mineralisation, and to determine the fate of water and pollutants in sludge treatment wetlands. The experiment was conducted over three summers in mesocosm sized STWs planted in monoculture of *Phragmites australis*, *Typha angustifolia* and *Scirpus fluviatilis*, and compared to an unplanted control, all in duplicates. The experimental STWs were not completely drained, and a saturated layer was retained at the wetland bottom to favour evapotranspiration and pollution removal. The experimental systems were fed with concentrated fish farm sludge, and efficiency was evaluated by sludge dewatering and mineralisation as well as by mass balance analysis.

4.3 Methods

4.3.1 Experimental design

The experiment was conducted at the Montreal Botanical Garden (Quebec, Canada), which has a semi-continental climate with a warm, humid summer and a very cold winter. The mean monthly temperature reaches a maximum of 20.9°C in July and a minimum of -10.2°C in January. Average annual precipitation is 978.9 mm (22.2% as snow), and the growing season lasts for about 195 days, from mid-April to mid-September (Environment Canada, climate normals 1971-2000). The experimental setup consisted of 8 mesocosms (cylindrical shape, height: 1 m; diameter: 0.6 m) representing sludge treatment wetlands, each composed of 4 filter layers of different granular sizes (see Figure 4.1 for details). Contrary to conventional STWs, the experimental mesocosms were not completely drained, and a saturated layer was retained by placing an overflow at 25 cm from the bottom. All water coming out from the overflow was recovered in an outlet bucket for sampling. Each mesocosms was planted with a monoculture of *Phragmites australis*, *Typha angustifolia, Scirpus fluviatilis* and a fourth remained an unplanted control; all STWs treatment were in duplicate for a total of 8 mesocosms. A randomized block design was used for distributing the plant species among the mesocosms.

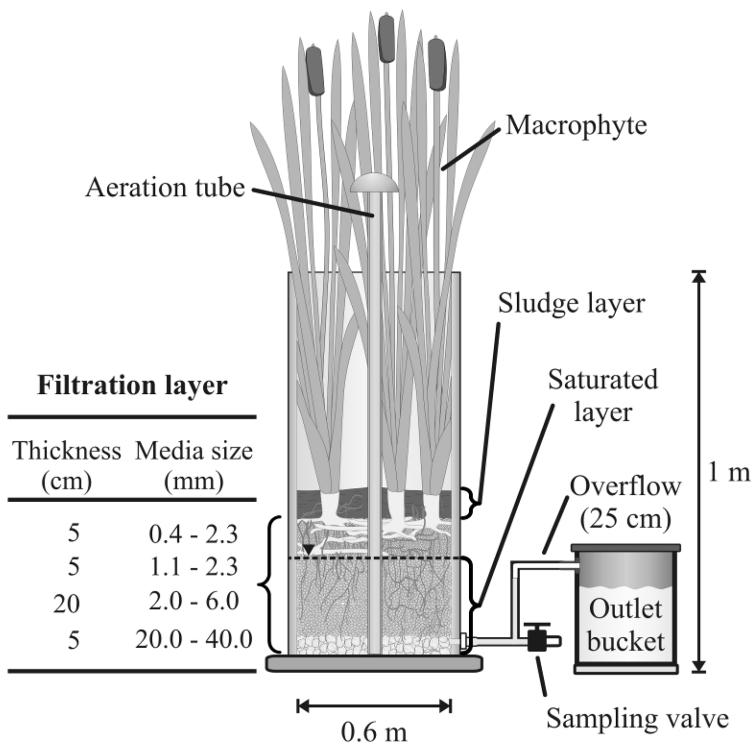

Figure 4.1. Cross section of the mesocosm, with the details of granular size of each filtration layer

4.3.2 Pollutant characteristics and loading rate

The mesocosms were fed with fish farm sludge, which is mainly composed of settled fish faeces and uneaten food (Naylor *et al.*, 1999) and is comparable to septage sludge in terms of pollutant composition and concentration (Troesch *et al.*, 2009; Vincent *et al.*, 2011). Average characteristics of the sludge and total input of pollutants are shown in Table 3.1. The mesocosms were planted at the end of the summer of 2007 and fed with fish farm sludge during the summers of 2008, 2009 and 2010. Loading was intermittent (1 day of feeding followed by six days of rest) with a weekly rate of 412 g TS m^{-2} wk^{-1} during the summer of 2008 (9 wk), 338 g TS m^{-2} wk^{-1} during the summer of 2009 (12 wk) and 575 g TS m^{-2} wk^{-1} during the summer of 2010 (14 wk) for a total of 0.59 m^3 m^{-2} of fish farm sludge. The loading rate of this study (30 kg m^{-2} yr^{-1} in 2010) was lower than the common loading rate of 50-60 kg m^{-2} yr^{-1} for septic sludge in STWs (Nielsen, 2003; Troesch *et al.*, 2009; Vincent *et al.*, 2011), since fish farm sludge contains a high level of ammonia (NH_4-N: 500 mg L^{-1}), which can be lethal to plants (Clarke and Baldwin, 2002). The mesocosms were not fed during winter, since the aim of this study was to measure the influence of plant species, which is expected to be minimal in STWs at freezing temperatures.

Table 4.1. Average of pollutant concentrations of the fresh sludge and total load of pollutants per surface of STW after the third summer of feeding.

Pollutants in sludge	**Concentration (mg l^{-1})**	**Total polluants input (kg m^{-2})**
Total solid (TS)	32 500 ± 14 000	17.4
Total volatile solid (TVS)	23 500 ± 13 000	12.4
Total Kjeldahl nitrogen (TKN)	20 000 ± 800	0.91
Total phosphorus (TP)	750 ± 200	0.42

4.3.3 Sampling

Samples were taken from three locations: 1) the sludge cake at the surface of the STW, 2) the saturated layer of the STW and 3) the system outlet (Figure 4.1). Core samples of the sludge cake were collected with a plastic cylinder (1.6 cm in diameter) one week after the last sludge application at two random points within the wetland, and sludge height was recorded. Part of the core sludge sample was analysed immediately for TS and TVS, while the rest was dried and stored at 4°C for subsequent TKN, TP and total carbon (TC) analysis. The saturated layer of the STWs was sampled at the end of the experiment by opening a valve located at the base of each mesocosm and collecting a volume of 300 mL. Water volume in the saturated layer was also estimated by using a calibrated water buoy installed in each STW; the volume was then used to calculate the mass of pollutants in the saturated layer at the end of the experiment. The outlet bucket was verified for water presence daily, and if water was present, the volume was measured and the water transferred to a container stored at -20°C. At the end of each week, the container was thawed and mixed, then analysed for TS and TVS. A subsample of the outlet water was refrozen (-20°C) and subsequently analysed for total phosphorus (TP), total Kjeldahl nitrogen (TKN) and total carbon (TC). Total volume discharged from the outlet for the week was used to calculate the mass of pollutants for this period.

4.3.4 Pollutants content of the sludge and masse balance analysis

Differences between plant species and the unplanted control were assessed with two different strategies 1) pollutant content of the sludge, determined by the ratio of pollutants per total sludge solids, and 2) mass balance analysis gives the distribution of water and pollutants in the mesocosms, which allowed us to evaluate the quantity remaining in the sludge cake, plants, saturated layer or discharged at the outlet. Furthermore, substances remaining unaccounted for were assumed to be an estimation of the percentage of pollutants trapped or transformed inside the STWs. A very low amount of pollutants (below sampling variation) appeared to be lost over the winter periods and was thus considered negligible.

4.3.5 Physical and chemical analyses of pollutants

Total solids (TS) and total volatile solids (TVS) were analysed according to Standard Methods (APHA, 1998). Total Kjeldahl nitrogen (TKN), total phosphorus (TP) and total carbon (TC) were measured using a Quikchem automated flow injection analyser according to the manufacturer's instructions (Quikchem 8500, Lachat). The percentage TS are reported per humid sludge cake, while TVS, TKN, TP and TC are presented by sludge cake dry total solid (Table 4.2). For the mass balance analysis, all concentrations of pollutants were multiplied by volume and divided by surface area of the mesocosms, which results are illustrated in Figure 4.2 as the percentage of pollutants per total pollutants added by the sludge per surface area of STW. The mass balance analysis of the sludge cake was calculated by dividing the mass of pollutants present in the core sludge sample by the sampling area (2 cm^2), which results are illustrated in Figure 4.2 as the percentage of pollutants per total pollutants added per surface area of STW. This extrapolation of the pollutant per surface area of STW was corrected by subtracting the surface area occupied by the plants and aeration pipe in the STWs.

4.3.6 Water balance analysis

Water balance was estimated for the summer of 2010 only, by calculating the amount of water in the sludge cake, the water lost by evapotranspiration, the volume present in the saturated layer and discharged at outlet. The amount of water present in the sludge layer of the STWs was calculated by extrapolating the water content of the core samples (2 cm^2) to the surface occupied by the sludge in the STWs. This was done at the beginning of the summer of 2010, to establish the initial water content of the sludge cake, and at the end of the summer. The difference between those values constitutes the amount of water retained in the sludge cake for this period, which was expressed as the percentage of water in the sludge cake per total water added (water in sludge + rain) during the summer of 2010 (Figure 4.2).

Water loss through evapotranspiration (ET) was calculated weekly by measuring total inlet volume, the variation of volume inside the mesocosms and total outlet volume (Equation 1).

$$ET = V_{In} - ((V_{d7}-V_{d1}) + V_{out}) \qquad (1)$$

V_{In} = Inlet volume (Sludge + rain volume for the week)

V_{d1} = Volume inside the mesocosm, day 1

V_{d7} = Volume inside the mesocosm, day 7

V_{out} = Volume collected from the outlet for the week

Water adhering to the sand and gravel media of the drained portion of the STWs was not included in the ET calculation, since it can be considered as negligible (Stefanakis and Tsihrintzis, 2011). The total volume lost by evapotranspiration was then divided by the surface of the mesocosm and by the 7 days of the week. Therefore, the average ET is expressed by L m^{-2} d^{-1} (Table 4.2) and presented as the percentage of water lost by evapotranspiration per total water added during the summer of 2010 in the mass balance analysis (Figure 4.2). The volume of water in the saturated layer at the end of the experiment and the total volume discharged at the outlet of the STWs are also presented as the percentage per total water added in the mass balance analysis (Figure 4.2). The water balance closed almost perfectly with less than 5% error, which was redistributed proportionally to avoid a total higher than 100%.

4.3.7 Plant density and nutrients content

At the end of each summer, the number of stems was counted and the above-ground portions were cut, dried and weighed. The measured weight of above-ground biomass was then divided by the surface area of the mesocosms and used to estimate the nutrient uptake by the plants. Below-ground biomass was assessed at the end of the third summer for only one replicate of each species. Half of the volume of each mesocosm was excavated and the rhizome and roots were collected, dried and weighed. The measured mass was then divided by the excavated surface of the mesocosms and used to estimate the nutrient uptake by the plants. Plant uptake of nitrogen and phosphorus was estimated by multiplying dry biomass (above- and below-ground) by the specific ratio of nutrients per dry biomass according to values determined by Tanner *et al.* (1995), Ennabili *et al.* (1998) and Smith *et al.* (2008). Since no phosphorus ratio was found in the literature for below-ground biomass of *Scirpus fluviatilis*, only above-ground phosphorus content is presented. The amount of nutrients present in plants tissues was expressed in percentage of nutrients per total nutrients added by the sludge (Figure 4.2). Plant density at the end of the summer of 2010 is presented in Table 4.2, which corresponds to the peak of plant establishment in the system.

4.4 Results

4.4.1 Plant parameters

The plants reached their maximum density and aboveground biomass at the end of the summer of 2010, with the highest value obtained by *Phragmites* (1 432 plants m^{-2} / 3087 g m^{-2}), followed by *Typha* (258 plants m^{-2} / 827 g m^{-2}) and then *Scirpus* (120 plants m^{-2} / 100 g m^{-2}) (Table 4.2). The average evapotranspiration rate for the summer of 2010 followed the same pattern, with the highest value for *Phragmites* (10.9 L m^{-2} d^{-1}) followed by *Typha* (5.3 L m^{-2} d^{-1}) and finally *Scirpus* (3.3 L m^{-2} d^{-1}), which had a similar value to the unplanted control (3.0 L m^{-2} d^{-1}) (Table 4.2).

4.4.2 Sludge volume reduction

A total of 0.59 m^3 m^{-2} of sludge was added to the STWs during the experiment. The highest reduction in sludge volume was measured in *Phragmites* STWs (89%), where the sludge cake was reduced to 0.07 m^3 m^{-2}, followed by the unplanted control with 0.09 m^3 m^{-2} (85%), *Scirpus* with 0.09 m^3 m^{-2} (84%), and *Typha* with 0.12 m^3 m^{-2} (80%) (Table 4.2)

Table 4.2. Plant density, evapotranspiration and characteristics of raw sludge and the sludge cake according to different plant species at the end of summer 2010 (Standard deviation is presented as ± beside de value)

	Plant density (Nb. m^{-2})	ET ($L\ m^{-2}\ d^{-1}$)	Volume reduction (%)	TS (%)	TVS (%)	TC (%)	TKN (%)	TP (%)
Sludge				4 ± 2	72 ± 14	39 ± 8	6.4 ± 1.6	2.4 ± 0.9
Phragmites	1 432 ± 165	10.9 ± 0.6	89 ± 1	31 ± 3	40 ± 6	32 ± 2	3.7 ± 0.3	2.3 ± 0.2
Typha	258 ± 55	5.3 ± 1.3	80 ± 10	28 ± 1	42 ± 6	31 ± 1	3.0 ± 0.4	2.2 ± 0.7
Scirpus	120 ± 50	3.3 ± 0.6	84 ± 1	33 ± 15	39 ± 2	31 ± 2	2.8 ± 0.4	1.9 ± 0.2
Unplanted		3.0 ± 1.1	85 ± 3	28 ± 5	34 ± 1	30 ± 7	2.1 ± 0.3	1.6 ± 0.2

4.4.3 Water fate in STWs

Approximately 762 L m^{-2} of water, including rain, was added to the STWs during the summer of 2010, where it was mainly evapotranspirated in *Phragmites* STWs (95%) and to a lesser extent in *Typha* (66%), while only 41% of water was lost by evapotranspiration in *Scirpus* and 39% for the unplanted control. The water discharged at the STWs outlet showed an inverse pattern, with the lowest value for *Phragmites* (1.6%), followed by *Typha* (25%), while *Scirpus* and the unplanted STWs had an outflow representing respectively 48% and 49% of the water input to the STWs. The water remaining in the saturated layer of the STWs represented a small fraction of the water input, with 1.6% for *Phragmites* followed by 6% for *Typha*, while *Scirpus* and the unplanted control had 8 and 9% respectively. The *Phragmites* STWs sludge cake had the lowest water level (1.4%), while the other plant species and the unplanted sludge cakes had levels about twice as high (2.9%, 2.8% and 2.6% for *Typha*, unplanted and *Scirpus* respectively).

4.4.4 Total solids

The ratio of TS ranged from 4% in the fresh sludge to 28% in the sludge cake of the *Typha* STWs and the unplanted, followed by 31% for *Phragmites* and 33% in *Scirpus* (Table 4.2).

In terms of mass balance, a total of 17.4 kg TS m^{-2} was added to the STWs during the experiment, and only half of the solids (52%) remained in the sludge cake of *Phragmites* STWs, while the unplanted, *Scirpus* and *Typha* retained 67, 70 and 74% respectively (Figure 4.2b). The amount of solids present in the saturated layer and at the outlet of the STWs was low (0.1% or less) for all the planted systems. However, a slightly higher portion of solids was measured in the saturated layer (0.1%) and at the outlet of the STWs (0.4%) of the unplanted control. For all treatments, the remaining solids unaccounted for were considered as trapped or transformed.

4.4.5 Total volatile solids

The ratio of TVS dropped from 72% in the fresh sludge to about 40% in the sludge cake of the planted systems, while the unplanted controls had the lowest (34%) volatile solids (Table 4.2).

In terms of mass balance, a total of 12.4 TVS m^{-2} was added to the STWs, and at the end of the experiment, most of the TVS was considered trapped or transformed within the STWs, with a higher fraction in *Phragmites* (71%), followed by the unplanted (68%), *Scirpus* (62%) and *Typha* (56%) (Figure 4.2c). The remaining volatile solids were retained in the sludge layer, with the lowest amount of volatile solids in the sludge cake of *Phragmites* STWs (29%), followed by the unplanted (32%), *Scirpus* (38%) and *Typha* (43%). The amount of volatile solids present in the saturated layer and at the outlet of the STWs was low (0.1% or less) for all planted units. Higher amounts of volatile solids were measured in the saturated layer (0.1%) and at the outlet of the STWs (0.2%) of unplanted STWs.

4.4.6 Total carbon

The ratio of TC per dry solids dropped from 39% in the fresh sludge to 32% in the sludge cake of *Phragmites* STWs, 31% for *Typha* and *Scirpus*, and 30% for the unplanted control (Table 4.2).

In term of mass balance, the carbon added by the sludge represented 6.9 kg TC m^{-2} and was mostly accumulated in the sludge cake, with a lower faction present in *Phragmites* (42%), followed by the unplanted (51%), *Scirpus* (54%) and *Typha* (59%) (Figure 4.2d). The rest of the carbon was generally trapped or transformed, with 58% for *Phragmites* STWs, followed by the unplanted (46%), *Scirpus* (45%) and *Typha* (41%). Only a small fraction of the carbon was present in the saturated layer of the STWs, with 0.2 to 0.4% for the planted systems and 0.8% for the unplanted controls. The carbon at the outflow followed a similar pattern, with the lowest amount of carbon for *Phragmites* (0.1%), followed by *Typha* (0.4%), *Scirpus* (0.8%) and the unplanted control (2.2%).

4.4.7 Total Kjeldahl nitrogen

The ratio of TKN per dry solids dropped from 6.4% in the fresh sludge to 3.7% in the sludge cake of *Phragmites* STWs, followed by *Typha* with 3.0% and *Scirpus* with 2.8%, while the control had the lowest percentage of nitrogen, with 2.1% (Table 4.2).

In terms of mass balance, the nitrogen added by the fresh sludge represented 0.91 kg TKN m^{-2} and most of it was trapped or transformed in the unplanted control (66%), followed by *Scirpus* (60%), *Typha* (50%) and *Phragmites* (41%) (Figure 4.2e). The remaining nitrogen was generally found in the sludge cake, with a lower portion in the unplanted control (26%), followed by *Phragmites* STWs (37%), *Scirpus* (38%) and *Typha* (43%). Part of the nitrogen was also sequestered in plant tissues, with 22 % of the total nitrogen in *Phragmites*, 6% for *Typha* and 0.8% for *Scirpus* STWs. The rest of the nitrogen was present in the saturated layer of the planted STWs (0.1% to 0.5%), with a higher amount in the unplanted control, which represented 2% of the total nitrogen input by the sludge. The fraction of nitrogen at the outflow was low in the planted STWs (0.2 to 1.0%), but significantly higher for the unplanted control (5.3%).

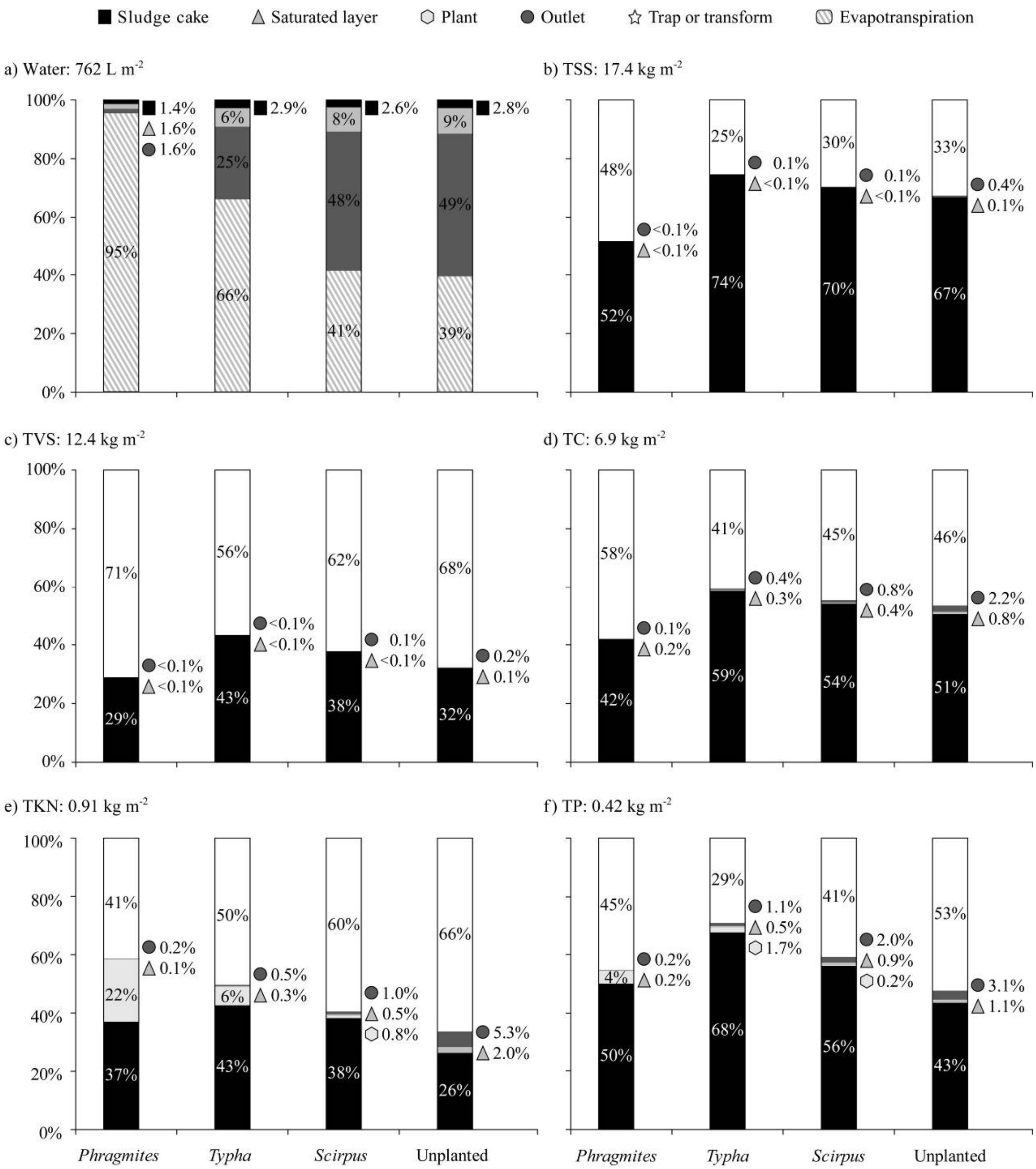

Figure 4.2. Mass balance of water and pollutants from summers 2008 through 2010 according to plant species: a) water balance*, b) total solids, c) total volatile solids, d) total carbon, e) total Kjeldahl nitrogen, f) total phosphorus. *Water balance only for summer 2010

4.4.7 Total phosphorus

The ratio of TP per dry solids diminished only slightly, from 2.4% in the fresh sludge to 2.3% in the *Phragmites* sludge cake, followed by *Typha* with 2.2% and *Scirpus* with 1.9%. The sludge cake of the unplanted control had a ratio of 1.6% of TP per dry solids (Table 4.2).

In terms of mass balance, phosphorus added by the fresh sludge (0.42 kg TP m^{-2}) was generally retained in the sludge cake, with 43% of the phosphorus for the unplanted control, followed by *Phragmites* (50%), *Scirpus* (56%) and *Typha* (68%) (Figure 4.2f). The rest was mainly considered as trapped or transformed in the STWs with 53% for the unplanted control, followed by *Phragmites* (45%), *Scirpus* (41%) and *Typha* (29%). A small fraction of the phosphorus was sequestered in plant tissues, with 4% for *Phragmites*, 1.7% for *Typha* and 0.2 for *Scirpus*. A minute amount of the sludge phosphorus was present in the saturated layer, with a value ranging from 0.2 to 0.9% in the planted systems and 1.1% in the unplanted. Very little phosphorus reached the outlet of the STWs, with a value between 0.2 to 2.0% in the planted systems and 3.1% at the outlet of the unplanted controls.

4.5 Discussion

Sludge pollutants were mainly retained within the sludge cake at the surface of the STWs, the remainder was generally considered as trapped inside the STWs or transformed into minerals, gas and water. In addition, a fraction of nitrogen and phosphorus of varying amounts, depending on the plant species, was sequestered in the plant biomass. Finally, only a very small percentage of the pollutants added by the sludge were present in the saturated layer or were discharged at the outlet of the STWs (Figure 4.2). The low percentage of pollutants discharged can be explained by the good physical filtration of the system and by the fact that the STWs were not completely drained, thus favouring evapotranspiration and a longer contact time between the pollutants and the plant rhizosphere. A detailed account of the effect of plant species on water quality at the outlet of the STWs is presented in Gagnon *et al.* (2012). The effect of plant species on the sludge cake is complex, since it acts on both the ratio and total amount of water and pollutants present. Thus the following section will examine the specific influence of plant species on dewatering and mineralisation.

4.5.1 Sludge volume reduction and dewatering

Sludge treatment wetlands planted with *Phragmites* had the highest sludge volume reduction and were the most efficient in sludge dewatering based on mass balance analysis (Table 4.2, Figure 4.2 a-b). Sludge volume reductions vary between 80 to 89% depending on plant species, which is in the range reported by the literature (81 to 98%) for STWs (Cooper *et al.*, 2004). The difference in reduction of sludge volume between *Phragmites* and the unplanted control (4%) is in the range of the 3-8% reported in the literature (Edwards *et al.*, 2001; Stefanakis and Tsihrintzis, 2011). The lower amount of water per surface in the sludge cake of *Phragmites* can be partly explained by the extremely high evapotranspiration rate of *Phragmites*, which was 2 to 3.5 times higher than for *Typha* and *Scirpus* respectively (Table 4.2). The percentage of water lost by evapotranspiration in *Phragmites* (95%) and *Typha* (66%) was higher than reported in the literature for similar sized systems, with a maximum of 84% in *Phragmites* and 42% in *Typha* (Panuvatvanich *et al.*, 2009; Stefanakis and Tsihrintzis, 2011). This could be explained by the fact that our STWs was not completely drained, thus enhancing evapotranspiration, but could also be due to the lower volume of sludge applied in our experiment (Gagnon *et al.*, 2012). Furthermore, *Phragmites* had by far the highest plant density, which riddled the sludge cake with tunnels created by the movement of the stems in the wind. These tunnels are thought to favour the drainage and aeration of the sludge cake, and consequently the dewatering and mineralisation processes (Nielsen, 2003).

The humidity of the sludge cake, as measured by the percentage of solids, did not differ between plant species and the unplanted control, with about 30% solids, which is in the range (20-30%) for STWs planted with *Phragmites* (Uggetti *et al.*, 2010). The absence of difference between plant species concurs with the study by Uggetti *et al.* (2012), in which no difference between *Phragmites* and *Typha* was found in terms of percentage of solids. At first glance, this seems to contradict the results of the difference between species measured in the mass balance analysis, but can be explained by the fact that the solid content of the sludge cake provides a ratio, and not the physical amount of solids or water present in the sludge layer. Thus the sludge cake of *Phragmites* had the same ratio of solids and water as the other STWs, but in terms of mass, had a lower absolute amount of solids and water per surface of STW. Therefore, our results indicate that use of a mass balance analysis in combination with the percentage of pollutants in the sludge cake provides a more accurate understanding when comparing the effect of plant species in STWs.

4.5.2 Sludge mineralisation

Higher sludge volume reduction in *Phragmites* can also be attributed to the enhanced mineralisation of the organic matter, where part of the solids is transformed into simpler compounds such as minerals, gas and water. Higher mineralisation in *Phragmites* STWs is shown in the mass balance by the lower amount of solids, volatile solids and carbon per surface compared to the other plant species and, to a lesser extent, to the unplanted control. This could be explained by the enhancement of microbial activity favoured by the better aeration of the sludge cake in *Phragmites* STWs. However, a slightly higher amount of organic matter per surface of wetland was measured in the *Typha* and *Scirpus* sludge cake when compared to the unplanted control. This could be due to the presence of plant litter within the sludge, fragments of which were clearly visible within the sludge samples, even though plants were harvested at the end of each summer. Thus, the addition of organic matter by the plant litter could have mitigated the mineralisation process for *Typha* and *Scirpus*, but had a lesser impact on *Phragmites* STWs, where the litter fell on a highly mineralised sludge cake. This idea is supported by the greater volatile solids ratio in the sludge cake of the planted STWs (39-42%), which was in the same range as reported in the literature (40-50%) (Uggetti *et al.*, 2010) for planted STWs, and slightly lower for the unplanted control (34%). We would have expected a higher amount of carbon in the sludge cake of the planted systems, due to the addition of carbon from the plant litter. If the percentage of total carbon per solids in the sludge cake did not vary according to plant species or in the unplanted control, the mass balance analysis shows that the total amount of carbon is greater in *Typha* and *Scirpus*, compared to control.

Planted systems tended to retain a higher mass of nutrients (TKN and TP), and at a higher ratio, in the sludge cake when compared to the unplanted control. This could be attributed to the added plant litter, which returned part of the nutrients back to the sludge cake. Nonetheless, the reduction of nitrogen content per solids was very similar to results obtained by Uggetti *et al.* (2012), who reported a nitrogen ratio of 3.9% and 3.4% in the sludge cake of *Phragmites* and *Typha* respectively, when loaded with fresh sludge containing about 6.7% of TKN/TS. However, in terms of percentage of phosphorus in the sludge cake, Uggetti *et al.* (2012) found a net decrease, the percentage present in fresh sludge dropping from about 2.5% TP/TS to 0.14% and 0.02% in *Phragmites* and *Typha*. In comparison, our study showed that the phosphorus in the sludge cake did not change, with 2.3% and 2.2% of TP/TS in *Phragmites* and *Typha* respectively when fed with fresh sludge at 2.4% TP/TS. This may be considered a positive outcome, since the higher percentage of phosphorus per solids adds fertilizing quality to the sludge residue and limits discharge into the environment.

Nitrogen mineralization was efficient in STWs, where 41% to 66% of the total nitrogen input by the sludge was considered as trapped or transformed in the planted and unplanted STWs. In planted STWs, nitrogen is thought to have been mainly transformed into nitrogen gas, with the sequential process of ammonification in the sludge layer, followed by nitrification in the aerated sludge and through the oxygenated root zone, and finally denitrification in the saturated part of the STWs (Faulwetter *et al.*, 2009). In addition, plants sequestered a fraction of nitrogen in their tissues, at a level particularly significant in *Phragmites*, with up to 22% of the total nitrogen input by sludge. Similar results were found by Korboulewsky *et al.* (2012), with a total of 23% of nitrogen input by sludge in *Phragmites* biomass. However, the unplanted STWs had limited nitrification due to the lack of available oxygen, which resulted in an accumulation of ammonia in saturated layers and prevented the removal of nitrogen through the denitrification process (Gagnon *et al.,* 2012). Nonetheless, mass balance analysis revealed that the unplanted control had the highest percentage of nitrogen unaccounted for, which was considered trapped or transformed in the STWs. This high reduction in nitrogen may be the result of the ammonia volatilisation in unplanted systems, in which the transformation of the ammonium ion to ammonia gas is favoured under a pH higher than 7, warm temperatures and high ammonium concentration (Jayaweera and Mikkelsen, 1991). Unplanted STWs had a high level of ammonium and a high concentration of inorganic carbon, which are presumably calcium carbonate (unpublished data), and would have increased pH and favoured ammonia volatilisation.

Phosphorus was mainly retained in the sludge in the planted system (50-68%) and, to a lesser extent, in the unplanted control (43%). The remaining phosphorus was considered trapped or transformed from organic into inorganic forms in the sludge layer and leached to the saturated layer of the STWs, where it was probably adsorbed or precipitated on calcium, aluminum or iron present in the gravel media. The higher amount of phosphorus trapped inside the unplanted STWs (53%) could be also explained by the possible presence of calcium carbonate and a higher pH.

4.6 Conclusion

The fate of pollutants in sludge treatment wetlands was mainly characterised by its retention within the sludge cake, with remaining pollutants generally considered as trapped inside the STWs or transformed into minerals, gas and water. A fraction of the nitrogen and phosphorus was sequestered in plant tissues, representing close to a quarter of the nitrogen input by the sludge in *Phragmites* STWs. Only a very small percentage of the pollutants was discharged at the outlet, due to the good physical filtration of the system and the fact that the STWs was not completely drained. Plants played an important role in these STWs, particularly *Phragmites australis*, which exhibited the highest sludge volume reduction and the best sludge dewatering and mineralisation, as measured by the mass balance analysis. This was explained by *Phragmites*' high evapotranspiration rate and plant density, which created tunnels in the sludge cake and favoured sludge drainage and oxygenation. However, in terms of sludge cake humidity, as measured by the percentage of solids, *Phragmites* did not differ from the other plant species or the unplanted control. This demonstrates that the sludge cake of *Phragmites* had the same ratio of solids and water as the other STWs, but in terms of mass had physically less solids and water per surface of STW. Therefore, combining measurement of the percentage of pollutants with a mass balance analysis could be a more accurate way of comparing the effect of plant species in

STWs. However, the sludge cake of the planted systems had a higher mass and percentage of nutrients than the unplanted STWs, possibly due to the presence of plant litter in the sludge cake, this is not necessarily a negative finding, since the nutrients retained in the sludge cake could be used as fertiliser.

Chapitre 5: Conclusion générale

Le but principal de cette étude était de déterminer l'influence de la présence ainsi que de l'espèce de plante sur le traitement des polluants en marais filtrant selon deux expériences ayant des conditions totalement différentes de climat, d'effluents ainsi que de types de marais filtrant. Mes expériences ont démontré que les plantes ont une influence significative sur l'épuration des eaux, même lorsque le traitement s'effectue dans des conditions qui sont extrêmement différentes. De plus, l'effet positif des plantes est généralement spécifique à l'espèce utilisée, ce qui indique un impact particulier de la physiologie de la plante sur l'enlèvement des polluants. L'effet de chaque espèce sur le traitement serait dû à un ou plusieurs facteurs présentés dans le tableau synthèse 5.1. Il est à noter que même sous des conditions extrêmement différentes, l'effet des plantes reste similaire à celui du traitement d'effluent municipal, montrant ainsi la versatilité de ces systèmes.

Tableau 5.1 Synthèse de l'effet des plantes sur les mécanismes d'enlèvement de polluants selon les deux expériences

Type d'effluent		**Rejet hydroponique**	**Boue piscicole**
Type de marais sous surfacique		**Flux horizontal**	**Flux vertical/horizontal**
Effets des plantes	**Saison**	**Hivernale**	**Estivale**
Oxygénation de la rhizosphère		Différente entre les espèces, inhibe la dénitrification, favorise la nitrification	Différente entre les espèces, inhibe la dénitrification, favorise la nitrification
Ajout de carbone par les exsudats racinaires		Différente entre les espèces, favorise la dénitrification	Négligeable, puisque la boue est très chargée en carbone
Évapotranspiration		Négligeable à basse température	Différente entre les espèces, réduit le volume et la masse de polluant en sortie
Séquestration du N et P dans les plantes		Nulle, car les plantes sont en sénescence	Différente entre les plantes, selon la biomasse de l'espèce
Filtration par le système racinaire		Nulle, car l'effluent ne contient pas de MES	Différente entre les systèmes plantés et non plantés
Prévention du colmatage		Nulle, car l'effluent ne contient pas de MES	Différente entre les espèces, permet le drainage et l'aération de la boue

L’effet positif des plantes et la distinction entre les espèces seraient expliquées par de multiples facteurs physiologiques lors du traitement des boues piscicoles, mais comparativement peu lors du traitement des rejets hydroponiques (Tableau 5.1). Ceci s’expliquerait par le fait que le traitement des rejets hydroponiques s’est fait en hiver, ce qui a limité l’effet des plantes sur le traitement. Le point commun entre ces expériences a été la diffusion d’oxygène dans la rhizosphère, ce qui a favorisé la nitrification de l’ammoniaque en nitrate, en particulier lorsque les systèmes étaient plantés de *Phragmites*. Ceci est clairement visible au cours de l’été dans la zone saturée des Lisam, où la masse d’ammoniaque est restée faible dans les systèmes plantés, mais a augmenté continuellement dans les contrôles non plantés. L’ammoniaque contenue dans les rejets hydroponiques a aussi été traitée de façon plus efficace dans les marais plantés de *Phragmites*, ce qui peut être expliqué par la diffusion passive d’oxygène au système racinaire. Ceci montre que les plantes peuvent avoir une influence durant la période hivernale, même lorsqu’elles sont en dormance. Par contre, puisque l’effluent hydroponique est principalement composé de nitrate, la présence d’oxygène dans la rhizosphère de *Phragmites* inhibe en partie la dénitrification et conséquemment l’espèce fût moins efficace pour l’enlèvement des nitrates.

La présence d’exsudats racinaires riches en carbone organique aurait stimulé la dénitrification lors du traitement des rejets hydroponiques et cela de façon significativement plus élevée pour les marais filtrant plantés de *Phalaris* comparativement aux autres espèces de plante. Toutefois, même si des différences significatives ont été mesurées, l’apport de carbone via les exsudats était insuffisant pour enlever de façon adéquate les nitrates des rejets hydroponiques, montrant ainsi les limites physiologiques des plantes lorsque l’effluent est très concentré. L’ajout d’une source externe de carbone (sucre) au marais filtrant a été démontré comme un moyen efficace pour stimuler la dénitrification avec de meilleurs résultats lorsque les marais étaient plantés de *Phalaris*. Les exsudats racinaires n'ont pas eu d'effet détectable lors du traitement des boues, puisque le carbone n'était pas limitant dans ce type d'effluent.

L'évapotranspiration fût un des facteurs les plus importants pour expliquer la différence entre les espèces de plantes lors du traitement des boues, mais fût négligeable lors du traitement des rejets hydroponiques, puisque l'expérience se déroulait en condition hivernale. L'évapotranspiration a favorisé l'épuration des eaux en augmentant le temps de rétention hydraulique, permettant ainsi un plus long contact entre les polluants et la rhizosphère des plantes. Une corrélation linéaire a été trouvée entre la biomasse aérienne et l'évapotranspiration. Toutefois le volume d'eau en sortie des Lisam a diminué de façon exponentielle et non de façon linéaire comme l'aurait prévu la perte d'eau par l'évapotranspiration. Ceci s'explique par le fait que les Lisam ne sont pas drainés, mais ont une surverse à 25 cm de la base du marais. Cette rétention de l'eau dans les Lisam permettrait de favoriser l'évapotranspiration et limiterait la quantité d'eau à la sortie pour les espèces ayant un fort taux de transpiration. Le bilan en eau des Lisam montre clairement que la majorité de l'eau est perdue lors de l'évapotranspiration dans les systèmes plantés de *Phragmites*, suivis de ceux plantés de *Typha*.

Une partie de l'épuration s'est faite par la séquestration de l'azote et du phosphore dans les tissus végétaux lors du traitement des boues, ce qui était impossible pour le traitement des rejets hydroponiques, puisque les mesures ont été faites en condition hivernale. L'enlèvement de l'azote a été très efficace dans les Lisam plantés de *Phragmites* avec presque le quart de l'azote ajouté par la boue qui a été séquestré dans les tissus végétaux. Les plantes ont aussi un effet mécanique par la filtration des particules via le système racinaire, ce qui a limité la quantité de MES à la sortie des Lisam plantés comparativement aux contrôles non plantés. Les tiges des plantes ont aussi joué un rôle en formant des canaux dans la boue, ce qui a favorisé le drainage et l'aération de la boue accumulée en surface des marais. La forte densité de tiges dans les Lisam plantés de *Phragmites* a résulté en une boue contenant un faible pourcentage d'eau et fortement minéralisée.

Cette étude démontre que le choix de l'espèce de plantes en marais filtrant est important et dépend de la composition de l'effluent à traiter. Dans le cas du traitement des rejets hydroponiques, le meilleur choix serait d'avoir un premier marais planté de *Phragmites* afin de nitrifier l'ammoniaque suivi d'un marais planté de *Phalaris* avec l'ajout d'une source externe carbone organique afin d'optimiser la dénitrification. Dans le cas des lits de séchage plantés de macrophyte, les meilleurs résultats seraient un marais planté de *Phragmites*, puisque cette espèce est efficace tant au point de vue de la déshydratation et de la minéralisation des boues que du traitement du lixiviat de boue. Toutefois, puisque *Phragmites* est une plante considérée invasive en Amérique du Nord, l'utilisation de *Typha* pourrait être une option puisque cette espèce a une efficacité comparable à *Phragmites* ou bien arrive en deuxième position.

Il est présumé que le rôle bénéfique des plantes en marais filtrant est issu en grande partie de l'influence de la rhizosphère sur les microorganismes responsables de la biodégradation des polluants. Il serait donc intéressant que des recherches futures s'orientent sur la microbiologie des marais filtrants artificiels afin de mieux comprendre les mécanismes d'épuration engendrés par cette interaction entre l'écosystème microbien et les macrophytes. Il serait tout aussi intéressant de faire ces mesures microbiennes en été comme en hiver et avec différents polluants afin de voir les limitations de ces systèmes biologiques.

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Annexes

Annexe 1 : Revue de la littérature sur la caractéristique des rejets hydroponiques

	Références					
Nutriments	**Grasselly *et al.* 2005**	**Koide *et al.* 2004**	**Prystay et Lo 2001**	**Park et al., 2008**	**Communication personnelle**	**Moyenne**
pH	7,3	7,3			6,6	**7,1**
DCO	37	12				**25**
MES	8	3				**6**
$N-NO_3$	283	868	223	325		**425**
$N-NH_4$		4	18			**11**
N					343	**343**
P			99	38	53	**63**
PO_4-P			99	30		**65**
$S-SO_4$	89	321			581	**330**
K^+		290		459	607	**452**
Ca^{2+}				296	381	**339**
Cl^-				80	303	**192**
Na^+				108	56	**82**
Mg^{2+}				85	143	**114**
HCO_3					79	**79**
Zn				0,5	0,8	**0,7**
Mn				0,8	0,8	**0,8**
Cu				0,1	0,1	**0,1**
Fe				1,5	2,6	**2,0**
B				0,7	1,2	**1,0**
MO					0,1	**0,1**
Si					11	**11**

Annexe 2 : Données sur le traitement des rejets hydroponiques

				Removal rate ($g/m^2/d$)			
Date	**Replicate**	**Species**	**Carbon**	**TP**	**NH_3-N**	**NO_3-N**	**COD**
2007-11-02	1	*Phalaris*	No	0,16	0,22	0,89	0,74
2007-11-02	2	*Phalaris*	No	0,05	0,26	1,12	0,81
2007-11-23	1	*Phalaris*	No	0,22	0,19	0,94	0,59
2007-11-23	2	*Phalaris*	No	0,27	0,20	0,66	0,05
2007-12-21	1	*Phalaris*	No	0,28	0,15	0,03	-0,02
2007-12-21	2	*Phalaris*	No	0,34	0,20	0,07	-0,52
2008-01-30	1	*Phalaris*	No	0,37	-	1,41	0,67
2008-01-30	2	*Phalaris*	No	0,14	-	1,10	0,06
2008-02-06	1	*Phalaris*	No	0,15	0,09	-0,27	-
2008-02-06	2	*Phalaris*	No	0,02	0,14	-0,36	-
2008-02-13	1	*Phalaris*	No	0,54	0,25	1,62	-
2008-02-13	2	*Phalaris*	No	0,64	0,26	1,38	-
2008-02-27	1	*Phalaris*	No	0,05	0,22	0,75	0,35
2008-02-27	2	*Phalaris*	No	-0,34	0,24	-0,59	0,21
2008-03-05	1	*Phalaris*	No	0,18	0,21	0,80	0,23
2008-03-05	2	*Phalaris*	No	0,34	0,33	0,58	0,14
2008-03-12	1	*Phalaris*	No	0,38	0,26	1,01	0,21
2008-03-12	2	*Phalaris*	No	0,40	0,32	0,79	0,12
2008-03-19	1	*Phalaris*	No	0,22	0,23	0,93	0,12
2008-03-19	2	*Phalaris*	No	0,12	0,35	1,13	-0,02
2008-04-02	1	*Phalaris*	No	-0,03	0,21	0,32	0,35
2008-04-02	2	*Phalaris*	No	0,01	0,24	0,49	0,24
2008-04-10	1	*Phalaris*	No	0,09	0,32	0,26	0,13
2008-04-10	2	*Phalaris*	No	0,30	0,44	0,87	0,37
2008-04-16	1	*Phalaris*	No	0,36	0,31	1,24	0,16
2007-11-02	1	*Phalaris*	Yes	0,83	0,23	4,61	22,74
2007-11-02	2	*Phalaris*	Yes	0,75	0,31	4,02	22,50
2007-11-23	1	*Phalaris*	Yes	1,85	0,42	6,87	23,78
2007-11-23	2	*Phalaris*	Yes	0,66	0,25	3,71	22,08
2007-12-21	1	*Phalaris*	Yes	0,43	0,25	4,26	22,10
2007-12-21	2	*Phalaris*	Yes	0,32	0,32	4,32	22,50
2008-01-30	1	*Phalaris*	Yes	0,45	-	5,43	22,70
2008-01-30	2	*Phalaris*	Yes	0,43	-	5,28	22,71
2008-02-06	1	*Phalaris*	Yes	0,37	0,35	5,52	-
2008-02-06	2	*Phalaris*	Yes	0,26	0,34	5,43	-
2008-02-13	1	*Phalaris*	Yes	0,38	0,36	4,83	-
2008-02-13	2	*Phalaris*	Yes	1,70	0,57	7,02	-
2008-02-27	1	*Phalaris*	Yes	-0,16	0,37	5,52	24,66
2008-02-27	2	*Phalaris*	Yes	-0,17	0,30	5,58	25,10
2008-03-05	1	*Phalaris*	Yes	0,31	0,42	6,03	22,25
2008-03-05	2	*Phalaris*	Yes	0,25	0,34	6,06	22,94
2008-03-12	1	*Phalaris*	Yes	0,27	0,38	5,54	28,64
2008-03-12	2	*Phalaris*	Yes	0,21	0,30	5,61	28,96
2008-03-19	1	*Phalaris*	Yes	0,20	0,28	5,83	22,50
2008-03-19	2	*Phalaris*	Yes	0,87	0,34	6,53	23,62
2008-04-02	1	*Phalaris*	Yes	0,40	0,32	5,44	23,17
2008-04-02	2	*Phalaris*	Yes	0,44	0,24	5,31	23,97
2008-04-10	1	*Phalaris*	Yes	0,44	0,25	6,32	22,83
2008-04-10	2	*Phalaris*	Yes	0,13	0,26	6,06	23,13
2008-04-16	1	*Phalaris*	Yes	0,74	0,25	5,93	23,78

				Removal rate (g/m²/d)			
Date	**Replicate**	**Species**	**Carbon**	**TP**	**NH_3-N**	**NO_3-N**	**COD**
2007-11-02	1	*Phragmites*	No	0,35	0,44	1,02	1,08
2007-11-02	2	*Phragmites*	No	0,24	0,41	-0,39	0,81
2007-11-23	1	*Phragmites*	No	0,20	0,34	0,60	0,57
2007-11-23	2	*Phragmites*	No	0,10	0,32	0,21	0,07
2007-12-21	1	*Phragmites*	No	0,09	0,31	0,43	0,32
2007-12-21	2	*Phragmites*	No	0,06	0,29	0,75	-0,27
2008-01-30	1	*Phragmites*	No	0,19	-	1,89	0,38
2008-01-30	2	*Phragmites*	No	-0,04	-	1,20	0,64
2008-02-06	1	*Phragmites*	No	0,25	0,26	-0,45	-
2008-02-06	2	*Phragmites*	No	0,01	0,23	0,18	-
2008-02-13	1	*Phragmites*	No	0,57	0,51	2,07	-
2008-02-13	2	*Phragmites*	No	0,63	0,43	1,98	-
2008-02-27	1	*Phragmites*	No	-0,16	0,45	-0,54	0,26
2008-02-27	2	*Phragmites*	No	-0,29	0,35	-0,66	0,23
2008-03-05	1	*Phragmites*	No	0,11	0,47	-0,06	0,12
2008-03-05	2	*Phragmites*	No	0,02	0,38	-0,48	0,30
2008-03-12	1	*Phragmites*	No	0,38	0,46	0,21	-0,68
2008-03-12	2	*Phragmites*	No	0,19	0,36	0,33	0,02
2008-03-19	1	*Phragmites*	No	0,25	0,50	-0,09	-0,03
2008-03-19	2	*Phragmites*	No	0,14	0,39	0,18	0,13
2008-04-02	1	*Phragmites*	No	-0,08	0,50	-0,03	0,33
2008-04-02	2	*Phragmites*	No	-0,18	0,33	0,51	0,36
2008-04-10	1	*Phragmites*	No	0,22	0,56	-0,42	-0,69
2008-04-10	2	*Phragmites*	No	-0,01	0,40	-0,04	0,28
2008-04-16	1	*Phragmites*	No	0,56	0,52	0,35	0,04
2007-11-02	1	*Phragmites*	Yes	1,04	0,41	4,30	22,75
2007-11-02	2	*Phragmites*	Yes	0,83	0,39	3,62	23,26
2007-11-23	1	*Phragmites*	Yes	0,76	0,27	4,01	23,65
2007-11-23	2	*Phragmites*	Yes	0,65	0,28	3,60	23,49
2007-12-21	1	*Phragmites*	Yes	0,51	0,29	5,07	22,94
2007-12-21	2	*Phragmites*	Yes	0,47	0,31	4,94	22,52
2008-01-30	1	*Phragmites*	Yes	0,76	-	4,20	23,75
2008-01-30	2	*Phragmites*	Yes	0,64	-	4,59	23,36
2008-02-06	1	*Phragmites*	Yes	0,41	0,33	4,56	-
2008-02-06	2	*Phragmites*	Yes	0,61	0,38	4,92	-
2008-02-13	1	*Phragmites*	Yes	0,67	0,41	4,17	-
2008-02-13	2	*Phragmites*	Yes	0,67	0,40	4,56	-
2008-02-27	1	*Phragmites*	Yes	0,03	0,38	4,71	25,76
2008-02-27	2	*Phragmites*	Yes	-0,10	0,33	5,31	25,54
2008-03-05	1	*Phragmites*	Yes	0,61	0,47	4,95	23,49
2008-03-05	2	*Phragmites*	Yes	0,32	0,41	5,07	23,20
2008-03-12	1	*Phragmites*	Yes	0,53	0,45	4,84	29,35
2008-03-12	2	*Phragmites*	Yes	0,35	0,38	5,10	29,23
2008-03-19	1	*Phragmites*	Yes	0,28	0,40	4,73	23,53
2008-03-19	2	*Phragmites*	Yes	-0,15	0,26	5,11	23,08
2008-04-02	1	*Phragmites*	Yes	0,69	0,50	4,52	23,54
2008-04-02	2	*Phragmites*	Yes	0,49	0,40	4,95	23,57
2008-04-10	1	*Phragmites*	Yes	0,50	0,53	4,93	23,88
2008-04-10	2	*Phragmites*	Yes	0,21	0,45	5,59	23,13
2008-04-16	1	*Phragmites*	Yes	0,82	0,48	4,85	24,22

				Removal rate ($g/m^2/d$)			
Date	**Replicate**	**Species**	**Carbon**	**TP**	**NH_3-N**	**NO_3-N**	**COD**
2007-11-02	1	*Typha*	No	0,35	0,14	1,69	0,64
2007-11-02	2	*Typha*	No	0,35	0,44	0,34	0,65
2007-11-23	1	*Typha*	No	0,47	0,14	0,94	0,27
2007-11-23	2	*Typha*	No	0,06	0,38	0,42	0,29
2007-12-21	1	*Typha*	No	0,24	0,09	0,54	-0,19
2007-12-21	2	*Typha*	No	0,06	0,34	-0,06	0,17
2008-01-30	1	*Typha*	No	-0,15	-	0,98	0,62
2008-01-30	2	*Typha*	No	0,00	-	2,28	0,53
2008-02-06	1	*Typha*	No	0,14	-0,03	0,24	-
2008-02-06	2	*Typha*	No	0,06	0,30	0,21	-
2008-02-13	1	*Typha*	No	0,53	0,25	1,44	-
2008-02-13	2	*Typha*	No	0,52	0,53	2,04	-
2008-02-27	1	*Typha*	No	0,00	0,17	-0,23	0,17
2008-02-27	2	*Typha*	No	0,00	0,46	-0,36	0,40
2008-03-05	1	*Typha*	No	0,29	0,21	0,09	0,30
2008-03-05	2	*Typha*	No	0,04	0,49	-1,35	0,26
2008-03-12	1	*Typha*	No	0,39	0,22	0,89	0,06
2008-03-12	2	*Typha*	No	0,27	0,52	0,53	0,15
2008-03-19	1	*Typha*	No	-0,06	0,23	0,48	0,13
2008-03-19	2	*Typha*	No	-0,05	0,52	-0,27	0,15
2008-04-02	1	*Typha*	No	0,07	0,15	0,58	0,67
2008-04-02	2	*Typha*	No	-0,16	0,49	0,14	0,45
2008-04-10	1	*Typha*	No	-0,13	0,18	-0,04	0,03
2008-04-10	2	*Typha*	No	0,24	0,55	0,29	0,49
2008-04-16	1	*Typha*	No	0,23	0,17	-0,21	0,42
2007-11-02	1	*Typha*	Yes	0,66	0,40	4,21	23,09
2007-11-02	2	*Typha*	Yes	1,04	0,41	4,80	23,05
2007-11-23	1	*Typha*	Yes	0,57	0,37	3,69	23,35
2007-11-23	2	*Typha*	Yes	0,74	0,34	4,31	23,14
2007-12-21	1	*Typha*	Yes	0,54	0,32	4,85	22,89
2007-12-21	2	*Typha*	Yes	0,46	0,32	5,30	23,85
2008-01-30	1	*Typha*	Yes	0,50	-	4,44	23,52
2008-01-30	2	*Typha*	Yes	0,67	-	5,33	23,89
2008-02-06	1	*Typha*	Yes	0,42	0,43	4,44	-
2008-02-06	2	*Typha*	Yes	0,47	0,29	4,95	-
2008-02-13	1	*Typha*	Yes	0,53	0,44	4,26	-
2008-02-13	2	*Typha*	Yes	0,53	0,27	3,84	-
2008-02-27	1	*Typha*	Yes	-0,23	0,38	4,68	25,43
2008-02-27	2	*Typha*	Yes	-0,42	0,23	5,34	25,28
2008-03-05	1	*Typha*	Yes	0,55	0,42	4,77	23,22
2008-03-05	2	*Typha*	Yes	0,51	0,28	5,41	23,03
2008-03-12	1	*Typha*	Yes	0,39	0,34	4,03	29,39
2008-03-12	2	*Typha*	Yes	0,39	0,29	5,07	29,15
2008-03-19	1	*Typha*	Yes	0,36	0,31	4,58	23,33
2008-03-19	2	*Typha*	Yes	0,25	0,23	5,58	23,19
2008-04-02	1	*Typha*	Yes	0,61	0,37	3,82	23,42
2008-04-02	2	*Typha*	Yes	0,56	0,23	4,80	23,22
2008-04-10	1	*Typha*	Yes	0,47	0,36	4,91	23,49
2008-04-10	2	*Typha*	Yes	0,26	0,22	5,73	23,04
2008-04-16	1	*Typha*	Yes	0,55	0,32	4,29	24,08

				Removal rate (g/m²/d)			
Date	Replicate	Species	Carbon	TP	NH_3-N	NO_3-N	COD
2007-11-02	1	Control	No	0,10	0,14	0,33	0,61
2007-11-02	2	Control	No	-0,03	0,40	0,24	0,95
2007-11-23	1	Control	No	0,27	0,15	0,57	0,32
2007-11-23	2	Control	No	0,15	0,31	0,54	-0,18
2007-12-21	1	Control	No	0,10	0,11	-0,93	-0,07
2007-12-21	2	Control	No	0,18	0,28	1,02	-0,17
2008-01-30	1	Control	No	0,13	-	0,69	0,67
2008-01-30	2	Control	No	0,12	-	1,65	0,51
2008-02-06	1	Control	No	0,06	-0,05	0,09	-
2008-02-06	2	Control	No	0,02	0,26	-0,33	-
2008-02-13	1	Control	No	0,45	0,22	1,56	-
2008-02-13	2	Control	No	0,46	0,48	1,74	-
2008-02-27	1	Control	No	-0,20	0,17	-0,69	0,39
2008-02-27	2	Control	No	-0,34	0,44	-0,30	-0,40
2008-03-05	1	Control	No	0,10	0,19	0,55	-0,07
2008-03-05	2	Control	No	0,42	0,49	1,79	0,09
2008-03-12	1	Control	No	0,28	0,15	0,69	-0,22
2008-03-12	2	Control	No	0,68	0,50	2,49	0,39
2008-03-19	1	Control	No	-0,03	0,23	0,45	0,01
2008-03-19	2	Control	No	-0,04	0,50	0,06	0,12
2008-04-02	1	Control	No	-0,03	0,17	0,54	0,54
2008-04-02	2	Control	No	-0,23	0,45	-0,60	0,20
2008-04-10	1	Control	No	-0,05	0,17	-0,55	0,34
2008-04-10	2	Control	No	0,03	0,40	-0,39	0,13
2008-04-16	1	Control	No	0,22	0,13	-0,19	-0,21
2007-11-02	1	Control	Yes	0,52	0,36	3,57	22,44
2007-11-02	2	Control	Yes	0,62	0,32	2,61	21,36
2007-11-23	1	Control	Yes	0,75	0,30	3,29	22,21
2007-11-23	2	Control	Yes	0,55	0,30	3,12	21,75
2007-12-21	1	Control	Yes	0,39	0,30	3,36	20,98
2007-12-21	2	Control	Yes	0,37	0,19	4,29	21,39
2008-01-30	1	Control	Yes	0,53	-	4,50	23,01
2008-01-30	2	Control	Yes	0,36	-	5,13	22,54
2008-02-06	1	Control	Yes	0,46	0,30	5,58	-
2008-02-06	2	Control	Yes	0,34	0,23	5,58	-
2008-02-13	1	Control	Yes	0,47	0,33	4,38	-
2008-02-13	2	Control	Yes	0,49	0,23	4,47	-
2008-02-27	1	Control	Yes	-0,56	0,31	5,19	24,51
2008-02-27	2	Control	Yes	-0,17	0,16	5,04	24,53
2008-03-05	1	Control	Yes	0,26	0,35	6,06	22,91
2008-03-05	2	Control	Yes	0,28	0,16	5,80	22,30
2008-03-12	1	Control	Yes	0,21	0,30	4,41	28,40
2008-03-12	2	Control	Yes	0,24	0,13	5,13	28,19
2008-03-19	1	Control	Yes	0,21	0,25	5,61	22,87
2008-03-19	2	Control	Yes	0,22	0,02	5,85	22,37
2008-04-02	1	Control	Yes	0,79	0,28	5,34	23,36
2008-04-02	2	Control	Yes	0,44	-0,01	3,93	22,28
2008-04-10	1	Control	Yes	0,28	0,20	5,29	22,93
2008-04-10	2	Control	Yes	0,24	-0,01	5,68	22,31
2008-04-16	1	Control	Yes	0,49	0,22	5,13	23,82

Annexe 3 : Revue de la littérature sur la caractéristique des boues

Type of wastewater	TS	TVS	TS	TVS	COD	BOD	TKN	NH_4-N	NO_3-N	TP	PO_4-P	Reference
	(%)	(%)	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	
Raw wastewater			220	165	500	200	40	25	0,01	8		Kadlec and Wallace, 2009
Activated sludge		72%	2 750	1 994			193			46		Liénard *et al.*, 1995
Activated sludge		71%	2 400	1 416	2 400		1 900	117	1	45	13	Vincent *et al.*, 2011
Thickened aerated sludge		80%	9 493	7 594	11 840		739	15	0	229	38	Troesch *et al.*, 2009
Activated sludge	1,3%		14 133		4 584		424			219		Gustavsson and Engwall, 2012
Thickened activated sludge plus settling sludge	3,0%	35%	22 340	7 760								Cui *et al.*, 2008
Surplus activated sludge	3,1%	72%	22 500	16 600								Stefanakis *et al.*, 2009
Sludge from BAF for pig manure	4,1%	74%	31 364	23 272	22 600	7 000		659			5 650	Edwards *et al.*, 2001
Septage sludge	2,0%	71%	19 000	13 500	17 000	2 800	1 000	350	5			Koottatep *et al.*, 2001
Fecal sludge			22 420			2 225	950	320	5			Panuvatvanich *et al.*, 2009
Fecal sludge	3,7%	65%		27 600	18 050	31 100		1 100	600			Kengne *et al.*, 2009
Septage sludge		71%	30 000	21 300	42 000		1 423	287		517	49	Vincent *et al.*, 2011
Fecal sludge		71%	30 450	21 620	38 200	10 000		1 500				Cofie *et al.*, 2006
Septage sludge		68%	35 185	23 926	47 051		1 555	302		699	46	Troesch *et al.*, 2009
Food industry sludge			7 830		8 330	3 280	200	15	1,3	33	0,2	Wang *et al.*, 2009
Diluted Fish sludge		82%	7 860	6 204	6 855							Summerfelt *et al.*, 1999
Decanted Fish sludge	2,9%	70%	28 500	20 500	68 000		1 800	470	44	820	120	Gagnon *et al.*, 2012

Annexe 4 : Données sur la masse de polluant à la sortie des Lisam

Date	Rep.	Species	Clay	Outlet (L/m^2)	Mass of pollutant at the outlet (g/m^2)							
					$P\text{-}PO_4$	TP	$N\text{-}NH_3$	$N\text{-}NO_3$	NTK	COD	TSS	TVSS
2010-06-02	1	Control	Yes	35,71	0,08	0,28	0,13	0,02	0,28	1,52	1,20	0,82
2010-06-02	2	Control	Yes	26,00	0,06	0,28	0,36	0,02	0,70	2,50	3,04	2,42
2010-06-09	1	Control	Yes	25,71	0,02	0,23	0,23	0,01	0,49	13,79	2,62	1,05
2010-06-16	1	Control	Yes	10,07	0,03	0,14	0,08	0,01	0,18	5,60	1,02	0,71
2010-06-23	2	Control	Yes	4,89	0,00	0,02	0,11	0,00	0,11	1,89	0,30	0,16
2010-06-30	1	Control	Yes	75,36	0,32	1,30	7,93	0,10	4,76	42,38	10,47	3,99
2010-06-30	2	Control	Yes	53,43	0,18	0,30	4,55	0,05	2,65	30,09	5,82	3,15
2010-07-07	1	Control	Yes	13,43	0,03	0,16	0,72	0,01	0,94	12,65	2,74	0,87
2010-07-07	2	Control	Yes	18,29	0,01	0,13	0,65	0,02	1,24	20,04	3,46	1,54
2010-07-14	1	Control	Yes	54,82	0,34	0,93	5,76	0,77	4,11	35,98	4,53	1,53
2010-07-14	2	Control	Yes	23,61	0,02	0,20	1,22	0,03	1,65	14,26	6,09	2,50
2010-07-21	1	Control	Yes	82,71	0,43	1,16	4,70	0,08	5,87	35,29	4,62	1,95
2010-07-21	2	Control	Yes	66,71	0,10	0,41	1,55	0,05	2,28	18,16	2,44	1,33
2010-07-28	1	Control	Yes	11,71	0,11	0,11	1,19	0,01	0,68	3,28	1,24	0,35
2010-07-28	2	Control	Yes	13,14	0,04	0,07	0,72	0,02	0,52	4,61	2,36	0,75
2010-08-04	1	Control	Yes	45,93	0,15	0,59	4,26	0,34	3,53	14,11	3,35	1,19
2010-08-04	2	Control	Yes	43,43	0,03	0,22	2,17	0,10	2,18	15,24	3,43	2,04
2010-08-11	1	Control	Yes	43,82	0,13	0,59	2,86	0,12	2,96	17,71	3,21	0,61
2010-08-11	2	Control	Yes	35,68	0,07	0,21	2,19	0,10	1,68	10,26	1,77	0,21
2010-08-18	1	Control	Yes	54,54	0,24	0,94	3,58	0,14	3,71	16,45	3,00	1,25
2010-08-18	2	Control	Yes	35,43	0,12	0,28	2,57	0,09	2,31	21,72	2,34	1,12
2010-08-25	1	Control	Yes	11,14	0,02	0,11	0,49	0,02	0,68	2,89	0,60	0,29
2010-08-25	2	Control	Yes	8,71	-0,01	0,04	0,38	0,01	0,49	4,14	0,65	0,40
2010-09-01	1	Control	Yes	1,89	0,00	0,01	0,05	0,00	0,10	0,43	0,16	0,06
2010-09-01	2	Control	Yes	6,00	0,00	0,02	0,16	0,01	0,30	2,28	0,75	0,24
2010-09-08	1	Control	Yes	2,46	0,01	0,04	0,29	0,00	0,21	0,67	0,23	0,09
2010-09-08	2	Control	Yes	4,43	0,01	0,05	0,70	0,00	0,58	4,52	0,58	0,34
2010-06-02	1	Control	No	34,61	0,10	0,44	0,25	0,02	0,40	1,28	1,09	0,64
2010-06-02	2	Control	No	29,07	0,04	0,13	0,06	0,02	0,24	1,17	7,40	0,65
2010-06-09	1	Control	No	54,32	0,28	0,32	0,19	0,03	0,62	29,52	8,47	3,22
2010-06-16	1	Control	No	1,14	0,00	0,02	0,03	0,00	0,03	0,22	0,07	0,03
2010-06-23	1	Control	No	6,71	0,03	0,14	0,31	0,00	0,27	3,58	1,01	0,44
2010-06-30	1	Control	No	74,86	0,54	2,04	5,51	0,07	4,15	42,15	7,64	4,19
2010-06-30	2	Control	No	18,36	0,03	0,19	0,96	0,01	0,59	9,77	2,02	0,95
2010-07-07	1	Control	No	15,29	0,03	0,28	0,44	0,01	0,80	5,12	2,57	0,92
2010-07-07	2	Control	No	35,79	0,18	1,06	0,95	0,03	3,44	37,60	8,16	4,12
2010-07-14	1	Control	No	54,61	0,40	1,63	4,18	0,07	4,24	23,18	7,48	2,13
2010-07-14	2	Control	No	41,57	0,11	0,55	5,23	0,07	2,87	36,92	5,19	2,74
2010-07-21	1	Control	No	59,61	0,49	1,48	3,29	0,16	2,84	16,40	3,13	1,37
2010-07-21	2	Control	No	63,18	0,34	1,01	4,20	0,07	4,28	37,49	4,57	2,42
2010-07-28	1	Control	No	19,36	0,06	0,39	1,62	0,02	1,16	5,21	2,57	0,73
2010-07-28	2	Control	No	12,39	0,01	0,15	1,54	0,02	0,85	5,01	1,92	0,90
2010-08-04	1	Control	No	49,07	0,25	1,23	5,62	0,22	3,88	20,58	3,53	1,03
2010-08-04	2	Control	No	38,79	0,11	0,56	4,17	0,16	3,43	18,33	4,88	3,10
2010-08-11	1	Control	No	38,21	0,17	0,79	1,72	0,11	2,38	8,90	3,27	0,42
2010-08-11	2	Control	No	2,39	0,00	0,02	0,17	0,01	0,17	1,32	0,18	0,03

2010-08-18	1	Control	No	56,79	0,29	1,55	3,88	0,13	3,73	15,54	2,81	0,99
2010-08-18	2	Control	No	29,29	0,12	0,39	0,00	0,07	1,94	14,87	1,32	0,66
2010-08-25	1	Control	No	11,14	0,03	0,16	0,35	0,01	0,58	2,76	0,67	0,24
2010-08-25	2	Control	No	11,07	0,02	0,16	0,63	0,01	0,81	7,64	0,64	0,30
2010-09-01	1	Control	No	5,54	0,01	0,07	0,17	0,01	0,31	1,29	0,66	0,20
2010-09-01	2	Control	No	8,00	0,01	0,06	0,26	0,01	0,36	2,77	0,77	0,38
2010-09-08	1	Control	No	4,57	0,03	0,12	0,55	0,00	0,41	1,08	0,25	0,09
2010-06-02	1	*Phrag.*	Yes	10,79	0,02	0,07	0,00	0,00	0,07	0,33	0,50	0,39
2010-06-02	2	*Phrag.*	Yes	10,57	0,03	0,13	0,07	0,00	0,23	0,90	1,60	1,16
2010-07-21	2	*Phrag.*	Yes	16,86	0,03	0,13	0,01	0,39	0,19	2,79	0,64	0,29
2010-08-04	2	*Phrag.*	Yes	6,43	0,01	0,05	0,02	0,51	0,07	1,84	-	-
2010-08-11	2	*Phrag.*	Yes	2,89	0,00	0,01	0,00	0,08	0,03	1,12	0,15	-
2010-06-02	1	*Phrag.*	No	13,57	0,02	0,07	0,00	0,01	0,05	0,30	0,32	0,22
2010-06-02	2	*Phrag.*	No	12,93	0,01	0,04	0,00	0,01	0,02	0,20	0,11	0,06
2010-08-04	1	*Phrag.*	No	12,64	0,04	0,14	0,01	0,32	0,13	3,41	0,81	0,21
2010-06-02	1	*Scirpus*	Yes	23,14	0,07	0,24	0,01	0,66	0,10	0,51	0,36	0,24
2010-06-02	2	*Scirpus*	Yes	15,21	0,02	0,07	0,00	0,24	0,06	0,33	0,12	0,08
2010-06-09	1	*Scirpus*	Yes	20,57	0,03	0,24	0,02	0,61	0,10	2,57	1,59	0,38
2010-06-30	1	*Scirpus*	Yes	29,25	0,11	0,32	0,03	0,91	0,18	4,11	0,57	0,18
2010-06-30	2	*Scirpus*	Yes	31,89	0,15	0,56	0,19	0,03	0,40	9,12	2,22	0,97
2010-07-07	1	*Scirpus*	Yes	31,50	0,13	0,54	0,14	0,03	0,57	5,37	2,60	0,95
2010-07-07	2	*Scirpus*	Yes	16,93	0,03	0,21	0,00	0,01	0,18	2,82	2,12	0,95
2010-07-14	1	*Scirpus*	Yes	48,50	0,25	0,84	0,27	0,11	0,56	8,36	1,67	0,95
2010-07-14	2	*Scirpus*	Yes	46,29	0,16	0,55	0,30	0,05	0,56	9,28	2,01	0,88
2010-07-21	1	*Scirpus*	Yes	70,39	0,28	0,93	0,25	1,86	0,73	7,78	0,54	0,28
2010-07-21	2	*Scirpus*	Yes	60,82	0,16	0,51	0,20	0,14	0,70	7,45	1,91	0,59
2010-07-28	1	*Scirpus*	Yes	13,79	0,00	0,07	0,03	0,35	0,10	1,61	0,98	0,12
2010-07-28	2	*Scirpus*	Yes	18,36	0,00	0,13	0,09	0,03	0,17	2,14	3,12	0,25
2010-08-04	1	*Scirpus*	Yes	44,14	0,09	0,37	0,20	1,15	0,49	12,51	2,79	0,55
2010-08-04	2	*Scirpus*	Yes	38,32	0,05	0,27	0,29	0,60	0,59	6,89	2,18	0,67
2010-08-11	1	*Scirpus*	Yes	30,29	0,04	0,20	0,06	0,83	0,27	3,70	1,67	-
2010-08-11	2	*Scirpus*	Yes	30,54	0,01	0,11	0,12	0,26	0,31	4,16	1,48	0,12
2010-08-18	1	*Scirpus*	Yes	45,57	0,14	0,39	0,15	1,32	0,42	6,91	0,49	0,20
2010-08-18	2	*Scirpus*	Yes	35,29	0,06	0,21	0,13	0,11	0,30	4,56	0,47	0,12
2010-08-25	1	*Scirpus*	Yes	7,54	-0,01	0,02	0,00	0,08	0,05	1,15	0,32	0,07
2010-08-25	2	*Scirpus*	Yes	11,50	0,01	0,07	0,02	0,02	0,09	1,83	0,38	0,09
2010-09-01	1	*Scirpus*	Yes	0,29	0,00	0,00	0,00	0,00	0,00	0,03	0,01	0,00
2010-09-01	2	*Scirpus*	Yes	5,86	0,01	0,05	0,00	0,01	0,04	0,63	0,45	0,07
2010-09-08	2	*Scirpus*	Yes	5,86	0,01	0,03	0,00	0,00	0,04	0,95	-	-
2010-06-02	1	*Scirpus*	No	26,71	0,06	0,30	0,01	0,43	0,07	0,46	0,29	0,17
2010-06-02	2	*Scirpus*	No	25,29	0,07	0,28	0,01	0,16	0,06	0,44	0,20	0,11
2010-06-09	1	*Scirpus*	No	18,86	0,17	0,45	0,02	0,01	0,13	3,46	3,32	1,11
2010-06-09	2	*Scirpus*	No	8,57	0,02	0,13	0,01	0,01	0,05	1,49	1,62	0,45
2010-06-23	1	*Scirpus*	No	8,93	0,01	0,09	0,02	0,00	0,04	1,06	0,90	0,25
2010-06-30	1	*Scirpus*	No	58,79	0,43	1,67	0,30	0,04	0,54	10,65	3,41	1,00
2010-06-30	2	*Scirpus*	No	50,82	0,42	1,28	0,28	0,04	0,51	14,57	3,35	1,74
2010-07-07	1	*Scirpus*	No	10,43	0,03	0,19	0,01	0,01	0,07	1,24	1,66	0,56
2010-07-07	2	*Scirpus*	No	10,25	0,03	0,18	0,00	0,01	0,10	1,77	1,35	0,52

2010-07-14	1	*Scirpus*	No	43,07	0,24	0,80	0,40	0,27	0,45	5,35	0,99	0,34
2010-07-14	2	*Scirpus*	No	45,68	0,24	0,83	0,35	0,04	0,58	7,45	2,23	0,78
2010-07-21	1	*Scirpus*	No	74,07	0,42	1,19	0,40	0,45	0,90	8,81	0,86	0,40
2010-07-21	2	*Scirpus*	No	82,39	0,32	0,91	0,54	1,96	0,95	8,33	0,54	0,35
2010-07-28	1	*Scirpus*	No	17,75	0,03	0,19	0,17	0,02	0,21	2,72	1,95	0,36
2010-07-28	2	*Scirpus*	No	16,25	0,01	0,13	0,14	0,50	0,20	1,92	1,08	0,16
2010-08-04	1	*Scirpus*	No	41,86	0,15	0,66	0,63	0,69	0,59	6,40	2,18	0,65
2010-08-04	2	*Scirpus*	No	43,04	0,15	0,59	0,28	1,60	0,65	8,52	2,52	0,83
2010-08-11	1	*Scirpus*	No	27,50	0,05	0,24	0,10	0,55	0,28	3,48	1,43	0,22
2010-08-11	2	*Scirpus*	No	32,50	0,08	0,38	0,11	1,23	0,35	4,52	2,08	0,15
2010-08-18	1	*Scirpus*	No	43,39	0,17	0,48	0,26	0,37	0,46	6,58	0,23	0,12
2010-08-18	2	*Scirpus*	No	52,71	0,21	0,63	0,20	1,24	0,42	8,21	0,41	0,16
2010-08-25	1	*Scirpus*	No	8,11	0,02	0,07	0,01	0,02	0,05	1,09	0,29	0,07
2010-08-25	2	*Scirpus*	No	15,39	0,03	0,14	0,02	0,09	0,11	2,22	0,73	0,18
2010-09-01	1	*Scirpus*	No	25,93	0,03	0,19	0,01	0,03	0,13	2,48	1,46	0,33
2010-09-01	2	*Scirpus*	No	7,79	0,00	0,06	0,00	0,01	0,04	0,81	0,74	0,16
2010-09-08	2	*Scirpus*	No	2,21	0,00	0,02	0,00	0,00	0,01	0,34	0,17	0,04
2010-06-02	1	*Typha*	Yes	8,93	0,02	0,08	0,01	0,02	0,06	0,23	0,32	0,18
2010-06-02	2	*Typha*	Yes	13,29	0,02	0,06	0,00	0,09	0,04	0,22	0,09	0,07
2010-06-30	1	*Typha*	Yes	8,43	0,03	0,14	0,00	0,01	0,06	2,22	1,26	0,70
2010-06-30	2	*Typha*	Yes	30,75	0,18	0,57	0,11	0,03	0,24	4,56	1,09	0,42
2010-07-07	1	*Typha*	Yes	2,46	0,01	0,04	0,00	0,00	0,02	0,42	0,34	0,13
2010-07-07	2	*Typha*	Yes	4,14	0,01	0,06	0,00	0,00	0,03	0,49	0,40	0,13
2010-07-14	2	*Typha*	Yes	4,86	0,00	0,03	0,03	0,02	0,03	0,64	0,16	0,03
2010-07-21	1	*Typha*	Yes	20,79	0,08	0,25	0,06	0,20	0,24	3,24	0,44	0,19
2010-07-21	2	*Typha*	Yes	42,75	0,08	0,24	0,04	0,86	0,32	5,54	0,28	0,19
2010-08-18	2	*Typha*	Yes	4,50	0,00	0,02	0,00	0,12	0,05	0,91	0,14	0,05
2010-06-02	1	*Typha*	No	47,21	0,14	0,91	0,27	0,03	0,48	2,65	2,27	1,63
2010-06-02	2	*Typha*	No	13,86	0,03	0,13	0,00	0,08	0,05	0,26	0,17	0,12
2010-06-09	2	*Typha*	No	19,00	0,07	0,47	0,05	0,01	0,18	4,19	2,87	1,09
2010-06-30	1	*Typha*	No	46,79	0,26	0,80	0,05	0,52	0,27	5,45	0,39	0,22
2010-06-30	2	*Typha*	No	44,04	0,33	1,09	0,18	0,19	0,43	6,88	2,17	0,94
2010-07-07	1	*Typha*	No	6,29	0,01	0,06	0,00	0,01	0,03	0,69	0,41	0,13
2010-07-07	2	*Typha*	No	8,21	0,05	0,18	0,01	0,00	0,07	1,22	0,90	0,29
2010-07-14	1	*Typha*	No	20,00	0,08	0,38	0,15	0,40	0,25	3,23	1,34	0,51
2010-07-14	2	*Typha*	No	8,21	0,02	0,12	0,02	0,07	0,09	1,74	0,56	0,18
2010-07-21	1	*Typha*	No	72,04	0,45	1,11	0,08	1,03	0,55	9,59	0,41	0,20
2010-07-21	2	*Typha*	No	44,43	0,27	0,71	0,07	0,42	0,45	6,92	0,59	0,33
2010-07-28	1	*Typha*	No	13,61	0,02	0,17	0,01	0,18	0,08	1,74	1,29	0,22
2010-07-28	2	*Typha*	No	5,57	0,00	0,05	0,01	0,01	0,05	1,13	0,53	0,13
2010-08-04	1	*Typha*	No	12,14	0,05	0,27	0,02	0,40	0,13	2,42	1,34	0,30
2010-08-04	2	*Typha*	No	4,14	0,01	0,06	0,03	0,17	0,08	1,37	0,45	0,18
2010-08-11	1	*Typha*	No	20,25	0,05	0,32	0,02	0,60	0,17	7,10	1,83	0,30
2010-08-11	2	*Typha*	No	6,07	0,01	0,05	0,01	0,15	0,07	1,25	0,40	0,11
2010-08-18	1	*Typha*	No	24,21	0,12	0,35	0,02	0,20	0,18	4,38	0,40	0,12
2010-08-18	2	*Typha*	No	8,29	0,02	0,07	0,01	0,07	0,08	2,22	0,46	0,15
2010-08-25	1	*Typha*	No	11,18	0,02	0,16	0,00	0,02	0,06	1,87	0,95	0,21
2010-09-01	1	*Typha*	No	1,07	0,00	0,01	0,00	0,00	0,01	0,10	0,03	0,01

Annexe 5 : Bilan de masse des polluants dans les Lisam de 2008-2010

		TS (g/m²)					TS distribution (%)			
Species*	**Rep.**	**Inlet**	**Cake**	**Saturated**	**Outlet**	**Trap or trans.**	**Cake**	**Saturated**	**Outlet**	**Trap or trans.**
Phrag.	**1**	17 381	9 494	0,7	0,9	7 885	54,6%	0,00%	0,01%	45%
Phrag.	**2**	17 381	8 477	1,5	2,2	8 901	48,8%	0,01%	0,01%	51%
Typha	**1**	17 381	12 932	0,8	11,3	4 437	74,4%	0,00%	0,07%	26%
Typha	**2**	17 381	12 963	1,2	9,8	4 407	74,6%	0,01%	0,06%	25%
Scirpus	**1**	17 381	11 617	2,3	20,9	5 740	66,8%	0,01%	0,12%	33%
Scirpus	**2**	17 381	12 768	3,0	18,6	4 592	73,5%	0,02%	0,11%	26%
Control	**1**	16 571	11 549	17,4	87,2	4 918	69,7%	0,10%	0,53%	30%
Control	**2**	16 571	10 593	9,3	40,7	5 928	63,9%	0,06%	0,25%	36%

		TVS (g/m²)					TVS distribution (%)			
Species*	**Rep.**	**Inlet**	**Cake**	**Saturated**	**Outlet**	**Trap or trans.**	**Cake**	**Saturated**	**Outlet**	**Trap or trans.**
Phrag.	**1**	12 352	4 168	0,4	0,4	8 183	33,7%	0,003%	0,004%	66%
Phrag.	**2**	12 352	3 006	0,7	1,1	9 344	24,3%	0,006%	0,009%	76%
Typha	**1**	12 352	4 744	0,3	4,2	7 603	38,4%	0,003%	0,034%	62%
Typha	**2**	12 352	5 995	0,7	3,8	6 352	48,5%	0,005%	0,031%	51%
Scirpus	**1**	12 352	4 600	0,5	6,5	7 745	37,2%	0,004%	0,053%	63%
Scirpus	**2**	12 352	4 765	0,9	6,3	7 580	38,6%	0,007%	0,051%	61%
Control	**1**	11 779	4 104	8,7	38,9	7 628	34,8%	0,073%	0,330%	65%
Control	**2**	11 779	3 466	3,2	18,1	8 292	29,4%	0,027%	0,153%	70%

		TC (g/m²)					TC distribution (%)			
Species*	**Rep.**	**Inlet**	**Cake**	**Saturated**	**Outlet**	**Trap or trans.**	**Cake**	**Saturated**	**Outlet**	**Trap or trans.**
Phrag.	**1**	6 880	3 186	9,7	0,6	3 684	46,3%	0,14%	0,01%	54%
Phrag.	**2**	6 880	2 601	18,0	11,2	4 249	37,8%	0,26%	0,16%	62%
Typha	**1**	6 880	3 948	24,1	33,7	2 874	57,4%	0,35%	0,49%	42%
Typha	**2**	6 880	4 115	21,9	26,5	2 716	59,8%	0,32%	0,38%	39%
Scirpus	**1**	6 880	3 724	25,7	54,2	3 076	54,1%	0,37%	0,79%	45%
Scirpus	**2**	6 880	3 732	29,9	54,2	3 064	54,2%	0,44%	0,79%	45%
Control	**1**	6 586	2 959	66,9	184,6	3 375	44,9%	1,02%	2,80%	51%
Control	**2**	6 586	3 710	35,8	101,5	2 739	56,3%	0,54%	1,54%	42%

		TKN (g/m²)						TKN distribution (%)				
Species*	**Rep.**	**Inlet**	**Cake**	**Sat.**	**Plant**	**Outlet**	**Trap or trans.**	**Cake**	**Sat.**	**Plant**	**Outlet**	**Trap or trans.**
Phrag.	**1**	911	372	0,5	198,4	1,3	339	41%	0,1%	21,8%	0,1%	37%
Phrag.	**2**	911	300	1,2	194,0	2,0	414	33%	0,1%	21,3%	0,2%	45%
Typha	**1**	911	367	2,1	54,8	4,2	483	40%	0,2%	6,0%	0,5%	53%
Typha	**2**	911	410	2,9	59,7	4,2	434	45%	0,3%	6,5%	0,5%	48%
Scirpus	**1**	911	376	5,2	7,5	9,2	514	41%	0,6%	0,8%	1,0%	56%
Scirpus	**2**	911	322	4,3	6,6	8,3	571	35%	0,5%	0,7%	0,9%	63%
Control	**1**	865	261	20,9	0,0	68,7	515	30%	2,4%	-	7,9%	59%
Control	**2**	865	193	13,8	0,0	23,6	635	22%	1,6%	-	2,7%	73%

		TP (g/m²)						TP distribution (%)				
Species*	**Rep.**	**Inlet**	**Cake**	**Sat.**	**Plant**	**Outlet**	**Trap or trans.**	**Cake**	**Sat.**	**Plant**	**Outlet**	**Trap or trans.**
Phrag.	**1**	419	232	0,3	18,3	0,5	168	55,3%	0,1%	4,4%	0,1%	40%
Phrag.	**2**	419	188	0,9	17,9	0,8	211	44,9%	0,2%	4,3%	0,2%	50%
Typha	**1**	419	227	2,2	6,8	5,0	178	54,3%	0,5%	1,6%	1,2%	42%
Typha	**2**	419	340	2,0	7,3	3,9	66	81,0%	0,5%	1,7%	0,9%	16%
Scirpus	**1**	419	250	3,5	1,0	8,9	155	59,7%	0,8%	0,2%	2,1%	37%
Scirpus	**2**	419	220	3,9	0,7	7,5	186	52,6%	0,9%	0,2%	1,8%	44%
Control	**1**	406	200	6,8	0,0	20,1	179	49,3%	1,7%	-	5,0%	44%
Control	**2**	406	151	1,7	0,0	5,1	247	37,3%	0,4%	-	1,2%	61%

* *Phrag. = Phragmites*

Annexe 6 : Comparaison entre l’efficacité épuratoire retrouvée dans la littérature et les valeurs de cette étude selon les deux types de marais filtrants (flux horizontal et vertical)

	Review of the treatment efficiency of horizontal subsurface flow CWs (Vymazal et al., 2006)						Treatment of a hydroponics wastewater using constructed wetlands in winter conditions(Gagnon et al., 2010)					
	Loading*	Inflow	Outflow	Inflow	Outflow	Efficiency	Loading	Inflow	Outflow	Inflow	Outflow	Efficiency
Pollutants	L/m^2/d	(mg/l)	(mg/l)	(g/m^2/d)	(g/m^2/d)	(%)	L/m^2/d	(mg/l)	(mg/l)	(g/m^2/d)	(g/m^2/d)	(%)
COD	42	284,0	72,0	12,0	3,5	71%	30	814,0	28,0	24,7	0,8	97%
TSS	50	107,0	18,1	5,4	1,2	78%	-	-	-	-	-	-
TP	44	8,7	5,2	0,4	0,3	32%	30	57,0	40,0	1,7	1,2	30%
TN	38	46,6	26,9	1,8	1,1	39%	30	245,0	71,0	7,4	2,1	72%
NH_4-N	27	38,9	20,1	1,1	0,7	34%	30	17,0	6,0	0,5	0,2	68%
NO_3-N	61	4,4	2,9	0,3	0,2	32%	30	228,0	65,0	6,9	1,9	72%

	Review of the treatment efficiency of vertical subsurface flow CWs (Vymazal et al., 2006)						Effect of plant species on water quality at the outlet of a sludge treatment wetland (Gagnon et al., 2012)					
	Loading*	Inflow	Outflow	Inflow	Outflow	Efficiency	Loading	Inflow	Outflow	Inflow	Outflow	Efficiency
Pollutants	L/m^2/d	(mg/l)	(mg/l)	(g/m^2/d)	(g/m^2/d)	(%)	L/m^2/d	(mg/l)	(mg/l)	(g/m^2/d)	(g/m^2/d)	(%)
COD	52	303,0	75,0	15,8	4,6	71%	3,5	68 000	231,3	184,1	0,025	99,99%
TSS	95	97,0	18,4	9,2	1,3	86%	3,5	28 500	15,2	75,4	0,009	99,99%
TP	44	8,6	4,4	0,4	0,2	50%	3,5	820	8,2	2,1	0,001	99,96%
TN	49	61,0	35,0	3,0	1,5	51%	3,5	1 800	10,3	4,7	0,002	99,96%
NH_4-N	51	45,6	15,8	2,3	0,6	74%	3,5	470	0,3	1,3	0,0003	99,98%
NO_3-N	46	1,6	15,1	0,1	0,9	-1 204%	3,5	44	1,2	0,1	0,003	98,00%