Departament d'Enginyeria
Agroalimentària i Biotecnologia

Departament d'Enginyeria
Hidràulica, Marítima i Ambiental

PhD Thesis

CLOGGING IN HORIZONTAL SUBSURFACE FLOW CONSTRUCTED WETLANDS

Measures, design factors and prevention strategies

Tesis Doctoral

COLMATACIÓN EN HUMEDALES CONSTRUIDOS DE FLUJO SUBSUPERFICIAL HORIZONTAL

Medidas, factores de diseño y estrategias de prevención

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El President	El Secretari
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Vi el poder en el sol, y a él reverencié
Vi el amor en la luna, y a ella adoré
Vi la sabiduría en las estrellas, y a ellas respeté
Vi la perfección en el cosmos, y a él admiré
Mas un día necesité ayuda, cariño y guía
Tendí al cielo mis manos implorantes
¡Cuán lejos moraban aquellos mis dioses!
Y lloré. Fue así que por primera vez vi,
próximo a mí, el lamento de otro hombre
llorando, también él, su estúpida soledad.

Anónimo

(escrito en la fuente de la Plaza Mayor de Cinco Olivas)

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Abbreviations and Symbols / Abreviaciones y símbolos

	English	Castellano
BOD/DBO	Biochemical Oxygen Demand	Demanda Bioquímica de Oxígeno
CHM	Constant Head Method	Permeámetro de carga constante
COD/DQO	Chemical Oxygen Demand	Demanda Química de Oxígeno
C_u	Uniformity coefficient	Coeficiente de uniformidad
D_{10}	Size of the sieve in which a 10% of a gravel sample pass through	Tamaño del tamiz por el que pasa el 10% de una muestra de árido
D_{50}	Size of the sieve in which a 50% of a gravel sample pass through	Tamaño del tamiz por el que pasa el 50% de una muestra de árido
D_{60}	Size of the sieve in which a 60% of a gravel sample pass through	Tamaño del tamiz por el que pasa el 60% de una muestra de árido
DM/MS	Dry Matter	Materia Seca
DPT	Differential Pressure Transductor	Transductor de presión diferencial
E_H	Redox Potential	Potencial redox
FHM	Falling Head Method	Permeámetro de carga variable
FWS CW	Free Water Surface Constructed Wetland	Humedal construido de flujo superficial
HRT	Hydraulic Retention Time	Tiempo de retención hidráulico
HSSF CW	Horizontal Subsurface Flow Constructed Wetland	Humedal construido de flujo subsuperficial horizontal
HUSB	Hydrolytic Up-flow Sludge Blanket reactor	Reactor hidrolítico de flujo ascendente
K	Hydraulic conductivity	Conductividad hidráulica
LAA	Los Angeles Abrasion test	Test de abrasión de Los Ángeles
ORL	Organic Loading Rate	Carga orgánica
P	Porosity	Porosidad
PE/hab-eq	Population equivalent	Habitante Equivalente
SRL	Solids Loading Rate	Carga de sólidos
SRT	Solids Retention Time	Tiempo de retención de sólidos

	English	Castellano
SSF CW	Subsurface Flow Constructed Wetland	Humedal construido de flujo subsuperficial
TN/NT	Total Nitrogen	Nitrógeno Total
TP/PT	Total Phosphorus	Fósforo Total
TS/ST	Total Solids	Sólidos Totales
TSS/SST	Total Suspended Solids	Sólidos en Suspensión Totales
UASB	Up-flow Anaerobic Sludge Blanket reactor	Reactor anaeróbico de flujo ascendente
VS/VS	Volatile Solids	Sólidos Volátiles
VSS/SSV	Volatile Suspended Solids	Sólidos en Suspensión Volátiles
WWTP/EDAR	Wastewater Treatment Plant	Estación Depuradora de Agua Residual

Abstract

Constructed wetlands are alternative systems to conventional wastewater treatment for small communities (up to 2000 PE). This is mainly due to that operation and maintenance costs are reduced. Despite the advantages, the clogging of the bed is the most important trouble encountered by the managers of subsurface flow wetlands treatment systems. The solids accumulation in the interstitial spaces of the gravel over time causes the decrease of both, hydraulic conductivity and porosity. This leads in the development of preferential paths and short-circuiting of the water through the wetland, which converge in overland flow and contaminant removal efficiency decrease. Therefore, clogging is the main limiting factor of the life span of a subsurface flow constructed wetland.

The strategies to reverse clogging imply important economic investments. Generally, the most widespread option is the replacement of gravel. Hence the need to measure realibly the degree of clogging of a system in order to identify the factors that influence this phenomenon and to search new configurations and strategies to delay its progress, thus operations needed to reverse it.

Therefore, the aims of this thesis must contribute to establish new design criteria and operation of horizontal subsurface flow constructed wetlands to minimize clogging processes without reducing the contaminant removal efficiency. Then, the objectives were to compare a method to measure hydraulic conductivity *in situ*, based on falling head permeameter, in order to determine the state of clogging of wetlands and its horizontal distribution; to determine the most suitable indicator to assess clogging in horizontal subsurface flow wetlands; to identifie new design and operation parameters likely to cause or encourage the clogging of the filter medium; and to analyse two new configurations in a pilot scale system in terms of contaminant removal and clogging development.

Among the different clogging indicators analysed (drenable porosity, hydraulic conductivity, accumulated solids and effective volume from a tracer test), hydraulic conductivity seems to be the best one. Firstly, because hydraulic conductivity provides information about the hydraulic behaviour inside the bed, and linked appropriately with the accumulated solids (a correlation of 74.5% was found between both indicators). Secondly, because its application in full-scale wetlands is more straightforward than other techniques.

The falling head permeameter for hydraulic conductivity measure is a simple method to measure reliably in the range of values that can be found in a wetland (0-500 m/d). It is a repeatable method although the discrepancies with a constant head permeameter (both in lab measurements and *in situ* constant head method). However, these discrepancies are acceptable in the range of conductivities measured. A potential relationship was found between the two methods ($\text{FallingHead} = \text{ConstantHead}^{0.7821}$). This indicates that the falling head method is less sensitive to small changes of hydraulic conductivity. Although, its straightforward applicability makes it suitable for practical purposes.

Hydraulic conductivity measurements by falling head permeameter lead to obtain reliable results of the degree of clogging and its horizontal distribution in subsurface flow constructed wetlands. The analysis of horizontal distribution of hydraulic conductivity allows identify aspects of the design that enhances preferential flow paths. Thus, wastewater tends to flow through the less resistance pathways. Wetlands with length/width ratio <1 could easily develop preferential paths and dead zones, while the design of wetlands with this ratio >1 force the water to flow occupying the entire width of

the bed. Therefore, the hydraulic behaviour is improved and consequently, clogging develops more uniformly. Water distributors also affect the appearance of preferential paths, favoured by a heterogeneous distribution at the inlet. Channels distributors are recommended instead of perforated pipes since they perform better and their maintenance is more comfortable.

The type of gravel used as filter media can also be a factor involved in the clogging development, especially if breakable gravels are used. Calcareous materials (limestone) are less resistant to abrasion than granitic ones. However, being composed by uniform and rounded aggregates, are preferable to granite, which its composition leads in angular and breakable particles. It is recommended to use gravels composed basically by quartz, due to their mechanical resistance properties, or gravels with values below 25% of Los Angeles abrasion test.

It is strongly recommended the use of primary treatments in order to reduce the solids loading rate at the inlet of the wetlands (limiting this parameter to 10 g/m^2d), which can greatly reduce the development of clog matter. In this sense, the use of a hydrolytic up-flow sludge blanket (HUSB) reactor can effectively remove particulate matter from wastewater than conventional primary treatments (as settlers). The HUSB reactor produced effluents clearly more reduced, which can imply a slight loss in contaminant removal efficiencies. However, this could be compensated by a lower accumulation of solids in the inlet of the wetlands (up to 30% less), then delaying the clogging symptoms.

Alternating saturated and unsaturated phases in the operation of horizontal wetlands allow increase contaminant removal efficiencies. This is especially favoured in cold moths, when the ammonium removal efficiency can achieve a 50% higher than wetlands conventionally operated. This operation strategy produced slightly higher accumulations of sludge (10% more accumulated sludge) instead, but more uniformly distributed within the media. However, the higher sludge accumulation was compensated by lower plant growth (and therefore roots), then keeping hydraulic conductivity values in the same range than conventional operation.

Resumen

Los humedales construidos son una alternativa al tratamiento convencional de agua residual para pequeños municipios (hasta 2000 hab-eq) ya que son fáciles de operar y mantener y tienen un coste de explotación bajo. El mayor inconveniente a que se enfrentan los explotadores de sistemas de tratamiento con humedales subsuperficiales es la colmatación del lecho. Con el tiempo, la acumulación de sólidos de diversa naturaleza en los espacios intersticiales del medio filtrante, provoca la disminución de la conductividad hidráulica y la porosidad iniciales de la grava. Esto conduce al desarrollo de caminos preferenciales y cortocircuitos en el curso del agua que convergen en la aparición de agua en superficie. A la larga, esto puede comprometer la capacidad depurativa del sistema. Por ello, la colmatación supone el factor limitante de la vida útil de un humedal construido.

Las estrategias para solventar la colmatación, una vez se ha producido, son costosas y pasan por realizar inversiones no despreciables. Generalmente la opción más extendida es el cambio del material granular. De ahí la necesidad de medir, de manera fiable, en qué grado un sistema está colmatado, identificar los factores que influyen en el fenómeno e indagar en nuevas configuraciones y estrategias que permitan retrasar el avance de la colmatación y consigo, aplazar las intervenciones necesarias para devolver al sistema un estado óptimo de funcionamiento.

Los objetivos de esta tesis doctoral han de contribuir a establecer nuevos criterios de diseño y operación de humedales construidos de flujo subsuperficial horizontal para minimizar, o cuanto menos retrasar, la colmatación de estos sistemas, sin mermar la eficiencia de eliminación de contaminantes del agua residual. Por ello, los objetivos son cuantificar la precisión y exactitud de un método de medición *in situ* de la conductividad hidráulica, basado en el permeámetro de carga variable, para la determinación del grado de colmatación de un lecho y la distribución horizontal de la misma; estudiar la idoneidad de diferentes indicadores de la colmatación de un humedal de flujo subsuperficial horizontal; evaluar la incidencia de diferentes factores de diseño y operación de humedales de flujo subsuperficial horizontal en el proceso de la colmatación; y caracterizar (en términos de eficiencia de eliminación de contaminantes y de evolución de la colmatación) dos nuevas configuraciones de humedales construidos a escala piloto.

De los distintos indicadores estudiados para evaluar el grado de colmatación de un humedal (porosidad drenable, conductividad hidráulica, acumulación de sólidos, volumen efectivo mediante ensayos de trazadores), la conductividad hidráulica es el que ofrece más ventajas. En primer lugar porque aporta información acerca del comportamiento hidráulico en el interior del lecho y se correlaciona bien con la acumulación de sólidos (se halló una correlación negativa entre conductividad hidráulica y sólidos acumulados del 74.5%). En segundo lugar, porque su aplicación en humedales a gran escala es mucho más sencilla que otros indicadores.

El permeámetro de carga variable para medir la conductividad hidráulica de un humedal de flujo subsuperficial horizontal es un método de sencilla aplicación que permite medir de manera fiable en el rango de valores que podemos encontrar en un humedal (0-500 m/d). Se trata de un método preciso y aunque presenta variaciones en los valores respecto al permeámetro de carga constante (tanto en el laboratorio como un método de medida *in situ*), son aceptables en el rango de conductividades a medir. La comparación del permeámetro de carga variable con uno de carga constante ha permitido establecer una relación potencial entre ambos ($\text{CargaVariable} = \text{CargaConstante}^{0.7821}$), la cual indica que el permeámetro de carga variable es un método menos sensible a variaciones de

conductividad. Sin embargo la facilidad de aplicación lo hace adecuado para fines prácticos.

La medida de la conductividad hidráulica mediante el permeámetro de carga variable permite, en consecuencia, obtener resultados fiables acerca del grado de colmatación en que se encuentra un humedal de flujo subsuperficial y de la distribución espacial de la misma. Esto permite identificar aspectos del diseño del humedal susceptibles de producir caminos preferenciales del agua residual en el lecho. El agua residual tiende a tomar el camino que menor resistencia ofrece al paso. Humedales con relación largo/ancho menores a 1 pueden desarrollar con facilidad caminos preferenciales y pueden presentar zonas muertas. El diseño de lechos con relaciones mayores a 1 fuerzan a que el agua circule por el humedal ocupando todo el ancho, por lo que se consigue una mejoría en el comportamiento hidráulico y consecuentemente una distribución más uniforme de la colmatación. Del mismo modo afecta el diseño el sistema de distribución del agua en la entrada del humedal. Una distribución heterogénea del agua en la entrada favorece la aparición de caminos preferenciales. Los canales de distribución dan mejores resultados que tuberías perforadas, teniendo además un mantenimiento más sencillo.

El tipo de grava utilizada en los humedales también puede ser un factor relacionado con el desarrollo de colmatación, sobre todo si se usan gravas fácilmente disgregables. Los áridos de naturaleza calcárea resultan menos resistentes a la abrasión y al desmoronamiento que aquellos de naturaleza granítica. Sin embargo, siendo el grano uniforme y redondeado, son preferibles al granito, cuya composición forma granos angulosos y fácilmente disgregables. La grava más recomendable es la compuesta eminentemente de cuarzo por sus cualidades de resistencia mecánica o bien gravas con valores por debajo del 25% en el ensayo de abrasión de Los Ángeles.

Es muy recomendable el uso de tratamientos primarios del agua residual para disminuir la carga de sólidos aplicada al humedal, lo que puede retrasar en gran medida el desarrollo de la colmatación. En este sentido, el uso de un reactor hidrolítico de flujo ascendente (HUSB) permite eliminar más eficazmente la materia particulada del agua residual que tratamientos primarios convencionales (basados en la decantación). El uso de un reactor HUSB como tratamiento primario, no obstante, produce efluentes claramente más reducidos y puede suponer una ligera pérdida en la eficiencia de eliminación de contaminantes en el sistema. En cambio, los humedales alimentados con el efluente de este reactor presentaron una menor acumulación de sólidos en la entrada del humedal (hasta un 30% menos), retrasando la colmatación.

La alternancia de fases saturadas e insaturadas en la operación de un humedal permiten obtener mayores eficiencias de eliminación de contaminantes. Esto se ve favorecido especialmente en los meses fríos, cuando la eficiencia de eliminación de materia orgánica y amonio puede llegar a ser un 50% mayor que en los humedales operados convencionalmente. Esta estrategia de operación produce una mayor acumulación de lodo en el lecho aunque más uniforme. Sin embargo la mayor acumulación de lodo se ve compensada con un menor crecimiento de plantas, y por lo tanto raíces pudiendo mantener conductividades hidráulicas mayores en el lecho.

Resum

Els aiguamolls construïts són una alternativa al tractament convencional d'aigua residual per a petits municipis (fins 2000 hab-eq) degut, principalment, a la facilitat en llur operació i manteniment y a les reduïdes despeses d'explotació. L'inconvenient més important amb què es troben els explotadors de sistemes de tractament amb aiguamolls subsuperficials és la colmatació del llit. Al llarg del temps, l'acumulació de sòlids de diversa natura en els espais intersticials de la grava, provoca la disminució de la conductivitat hidràulica i la porositat inicials. Això condueix al desenvolupament de camins preferencials i curtcircuits en el curs de l'aigua a través de l'aiguamoll, que convergeixen en l'aparició d'aigua en superfície, i que, a llarg termini pot comprometre la capacitat depurativa del sistema. Per això, la colmatació suposa el factor limitant de la vida útil d'un aiguamoll construït.

Les estratègies per a fer front a la colmatació, un cop s'ha produït, passen per realitzar inversions econòmiques gens menyspreables. Generalment, l'opció més extesa és la reposició del material granular. D'aquí neix la necessitat de mesurar, de manera fiable, el grau de colmatació d'un sistema, d'identificar aquells factors que influeixen en el fenomen y qüestionar noves configuracions y estratègies que permetin retardar l'avenç de la colmatació, i per tant ajornar les intervencions necessàries per tornar al sistema a l'estat òptim de funcionament.

Per tot això els objectius d'aquesta tesi doctoral han de contribuir a establir nous criteris de disseny i operació d'aiguamolls construïts de flux subsuperficial horitzontal per tal de minimitzar la colmatació d'aquests sistemes, sense minvar l'eficiència d'eliminació de contaminants de l'aigua residual. Els objectius específics són quantificar la precisió i exactitud d'un mètode de mesura *in situ* de la conductivitat hidràulica, basat en el permeàmetre de càrrega variable, per a la determinació de l'estat de colmatació d'un llit així com la distribució horitzontal d'aquesta; determinar l'indicador més adecuat per avaluar la colmatació d'un aiguamoll construït de flux subsuperficial horitzontal; identificar nous paràmetres de disseny y operación susceptibles de causar o afavorir la colmatación del medi filtrant; i caracteritzar (en termes d'eliminació de contaminants i d'evolució de la colmatación) dues noves configuracions d'aiguamolls construïts a escala pilot.

Dels diferents indicadors estudiats per avaluar el grau de colmatació d'un aiguamoll (porositat drenable, conductivitat hidràulica, acumulació de sòlids i volum efectiu), la conductivitat hidràulica és el més avantatjós. En primer lloc, perquè aporta informació sobre el comportament hidràulic a l'interior del llit i es relaciona adequadament amb els sòlids acumulats (amb una correlació negativa del 74.5%). En segon lloc, perquè la seva aplicació en aiguamolls a gran escala és molt més senzilla que altres tècniques.

El permeàmetre de càrrega variable per mesurar la conductivitat hidràulica és un mètode d'aplicació senzilla que permet mesurar de manera fiable dins el rang de valors que podem trobar en un aiguamoll (0-500 m/d). Es tracta d'un mètode precís i, tot i que presenta variacions en les mesures en relació al permeàmetre de càrrega constant (tant en mesures al laboratori com comparant-lo amb un mètode de mesura *in situ*), són acceptables en el rang de conductivitats mesurades. La relació potencial entre ambdós mètodes de mesura ($\text{CàrregaVariable} = \text{CàrregaConstant}^{0.7821}$) indica que el permeàmetre de càrrega variable és un mètode menys sensible a petites variacions de conductivitat. Malgrat tot la facilitat en la seva aplicació el fa adequat per a finalitats pràctiques.

La mesura de la conductivitat hidràulica mitjançant el permeàmetre de càrrega variable permet ob-

tenir resultats fiables de l'estat de colmatació d'un aiguamoll i de la distribució espacial d'aquesta. Cosa que permet identificar aspecte del disseny susceptibles de produir camins preferencials de l'aigua en el llit. L'aigua residual tendeix a prendre el camí que menys resistència ofereix al seu pas. Aiguamolls amb relacions llarg/ample inferiors a 1 desenvolupen amb facilitat camins preferencials i poden presentar zones mortes. El disseny d'aiguamolls amb relacions superiors a 1 forcen l'aigua a circular ocupant la totalitat de l'ample del llit, de manera que s'aconsegueix una millora en el comportament hidràulic i conseqüentment una distribució més uniforme de la colmatació. Igualment afecta a l'aparició de camins preferencials la distribució de l'aigua a l'entrada de l'aiguamoll. Una distribució heterogènia afavoreix l'aparició de camins preferencials. Els canals de distribució ofereixen millors resultats que canonades perforades i tenen un manteniment més senzill.

El tipus de grava utilitzat en els aiguamolls també pot ésser un factor relacionat amb el desenvolupament de la colmatació, sobretot si s'utilitzen graves fàcilment disgregables. Els àrids de naturalesa calcària són menys resistents a l'abrasió i esmicolament que el de naturalesa granítica. Tot i què, estant constituïts per agregats uniformes i arrodonides, són preferibles al granit, la composició del qual forma partícules anguloses i fàcilment disgregables. La grava més recomanable és aquella composta eminentment per quars, per llurs qualitats de resistència mecànica, o bé graves amb valors per sota del 25% en l'assaig d'abrasió de Los Ángeles.

És molt recomanable l'ús de tractaments primaris de l'aigua per disminuir la càrrega de sòlids aplicada a l'aiguamoll, cosa que pot retardar en gran mesura el desenvolupament de la colmatació. En aquest sentit, l'ús d'un reactor hidrolític de flux ascendent (HUSB) permet eliminar més eficaçment la matèria particulada de l'aigua residual que tractaments primaris convencionals (decantació). El reactor HUSB produeix efluents clarament més reduïts, cosa que pot suposar una lleugera pèrdua en l'eficiència d'eliminació de contaminants en el sistema. D'altra banda, els aiguamolls alimentats amb l'efluent d'aquest reactor van presentar una menor acumulació de sòlids a l'entrada de la'iguamoll (fins un 30% menys), retardant la colmatació.

L'alternança de fases saturades i insaturades en la operació d'un aiguamoll horitzontal permet obtenir millors eficiències d'eliminació de contaminants. Això es veu afavorit especialment durant els mesos freds, en què l'eficiència d'eliminació d'amoni pot arribar a ser un 50% superior que en aiguamolls operats de manera convencional. Aquesta estratègia d'operació produeix una major acumulació de fang en el llit, tot i què distribuïda més uniformement. Malgrat tot, aquesta major acumulació de fang es veu compensada per un menor creixement de plantes (i per tant d'arrels), de manera que es manté la conductivitat en valors similars a l'estratègia convencional d'operació.

1

Introducción

El saneamiento del agua residual constituye una de las actuaciones fundamentales para garantizar el buen estado de las masas de agua de un territorio. Desde la aprobación de la Directiva 2000/60/CE, conocida como Directiva Marco del Agua (DMA) el agua ha dejado de ser considerada exclusivamente como un recurso para ser calificada como parte estructural y funcional indispensable del medio natural. Se establece así un marco comunitario de actuación en la política de aguas, desarrollando medidas de protección, mejora y regeneración de las masas de agua superficiales. El objetivo de la DMA es garantizar el buen estado de los sistemas acuáticos, tanto en calidad como en cantidad, mediante la protección a largo plazo de los recursos hídricos gracias a un uso sostenible de los mismos.

Para dar respuesta a las exigencias de la DMA en cuanto a calidad, el Ministerio de Medio Ambiente, Medio Rural y Marino (MARM) puso en marcha el Plan Nacional de Calidad de las Aguas (PNCA): Saneamiento y Depuración 2007 – 2015 (MARM, 2007). En él se prevé dotar de sistemas eficaces de saneamiento a aquellas poblaciones y aglomeraciones de más de 2000 habitantes equivalentes (hab-eq) que en 2005 aún no gozaban de ellos, dando así total cumplimiento a la Directiva 91/271/CEE sobre el tratamiento de aguas residuales urbanas. Esto supone garantizar el saneamiento de 800 poblaciones entre 2000 y 4000 hab-eq en el plazo establecido, correspondiendo al 9% de la carga contaminante total.

Del mismo modo, la Agència Catalana de l'Aigua (ACA), empresa pública adscrita al Departament de Medi Ambient i Habitatge de la Generalitat de Catalunya y encargada de la política de aguas en Catalunya, puso en marcha el Programa de Saneamiento de Aguas Residuales Urbanas (PSARU 2005). Las líneas de actuación, planificadas en dos escenarios (2006-2008 y 2009-2014), pretenden alcanzar los objetivos fijados en la DMA. La planificación hidrológica en Catalunya prevé construir más de 1500 estaciones depuradoras de agua residual (EDAR) en el horizonte de 2014 para tratar también los núcleos de menos de 2000 hab-eq. En cuanto a urbanizaciones de menos de 2000 hab-eq, el PSARU 2005 detalla que al no ser consideradas "núcleos convencionales", la ACA no subvencionará obras de saneamiento en alta a menos que hayan adoptado mecanismos económicos a cargo de los usuarios que los dote de recursos necesarios, tal y como se especifica en el Apartado 6.4. del PSARU 2005. En este sentido el plazo límite de acreditación para ser objeto de financiamiento finaliza el 31 de diciembre de 2010. Por otro lado, las comunidades rurales remotas no siempre disponen de los recursos necesarios para hacer frente por ellos mismos a la inversión que supone la construcción y explotación de una EDAR convencional, amén de las dificultades que la orografía de la zona pueda suponer.

En Cataluña, la ACA cuenta en la actualidad con 377 EDAR, de las cuales 340 operativas (junio de 2010) que dan servicio al 90% de la población catalana. Aproximadamente el 35% del caudal tratado se concentra en dos EDAR, que corresponden a las depuradoras del Prat del Llobregat y del Besós. Si bien, debido a diversos factores como la distribución de los núcleos urbanos, la orografía del terreno, etc. esto siempre no es factible. La tendencia, en cuanto a la depuración del agua residual, es a centralizar los tratamientos en grandes EDAR que puedan dar servicio a varios municipios para minimizar los costes de construcción y explotación (Figura 1.1).

En España, fuera del ámbito de actuación del PNCA restan unos 13 millones de hab-eq agrupados en núcleos de menos de 2000 habitantes. Para poblaciones de menos de 2000 hab-eq la Directiva 91/271 no regula ni el tipo ni el nivel de tratamiento y se limita a instar el tratamiento más adecuado. De ahí la necesidad de investigar en sistemas de tratamiento alternativos que puedan facilitar el saneamiento de estas pequeñas aglomeraciones.

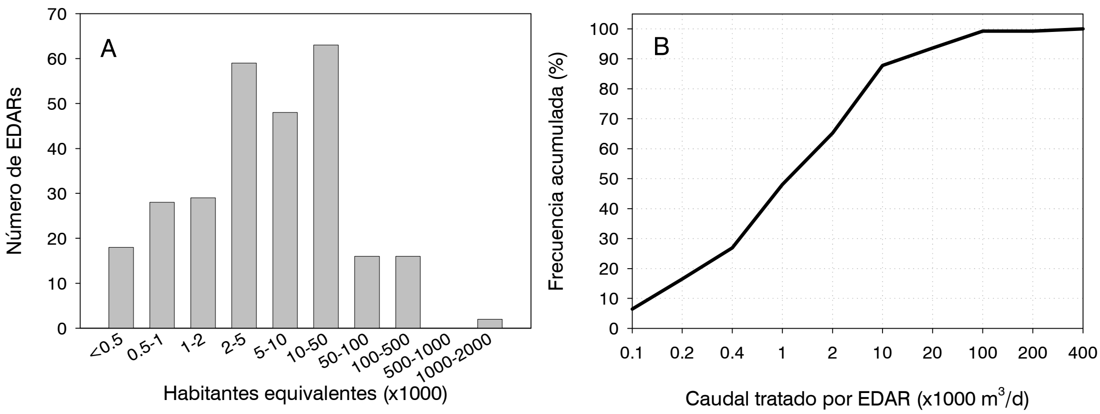

Figura 1.1 A) Distribución de EDAR en Catalunya (año 2008) según rangos de población servida y B) distribución de frecuencia acumulada de caudal tratado por EDAR. De las EDAR en servicio (340) sólo se ha contemplado aquellas de las que se conoce el caudal tratado (279). Fuente: ACA.

El PSARU 2005 también ahonda en la regulación de los tratamientos, ya que a menudo las EDAR disponen de la capacidad para alcanzar niveles de tratamiento por encima de lo exigido por la ley, lo que deriva en unos costes económicos difícilmente asumibles, ya sean debidos a incremento de consumo energético, inversión, reactivos, etc. Puede llegarse a dar situaciones en que un sobretratamiento sea perjudicial para el medio ambiente en su conjunto. Hay que valorar, por lo tanto, la eliminación de tratamientos considerados “excesivos”, siempre y cuando la reducción del nivel de tratamiento no suponga un empeoramiento del nivel de calidad del medio receptor por debajo del límite objetivo ni un incumplimiento de las obligaciones. En conclusión, el PSARU 2005 insta a valorar el equilibrio “razonable” entre nivel de depuración, siempre por encima del nivel exigido, y el coste que éste tiene asignado.

En este sentido, los sistemas naturales de tratamiento de agua residual constituyen una alternativa en auge desde la década de los años 70 en Europa (Rousseau et al., 2004; Vymazal, 2002) y desde los 90 en España (Puigagut et al., 2007) para el saneamiento de pequeñas comunidades. En Catalunya estas tecnologías representan casi el 10% del número total de sistemas de saneamiento del territorio (Figura 1.2).

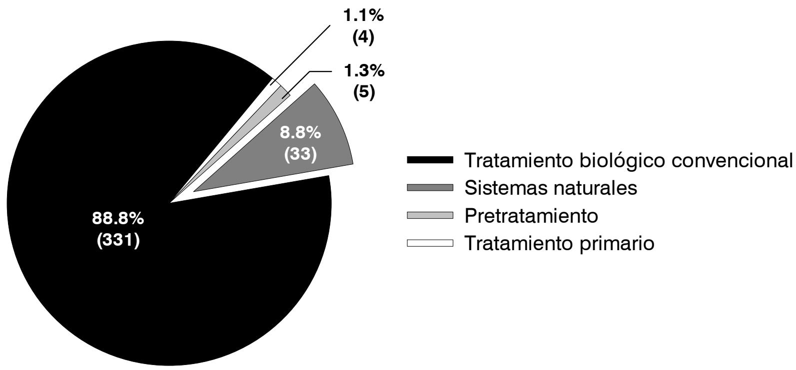

Figura 1.2 Distribución del saneamiento en Catalunya en 2009 (ACA, 2010). Entre paréntesis se indica el número de instalaciones.

1.1 Los sistemas naturales de tratamiento de agua residual

Por sistemas naturales o extensivos de tratamiento de agua residual se entiende aquellas tecnologías no convencionales que imitan ecosistemas naturales para depurar el agua residual. Entre estos, destacan el lagunaje y los humedales construidos.

Todos ellos son sistemas biológicos de tratamiento de agua en que la depuración se lleva a cabo sin aporte energético y por lo tanto las reacciones en estos sistemas se producen a velocidades ambientales (Salas et al., 2007). Por esta razón el tiempo de retención del agua residual en este tipo de tratamientos puede llegar a ser incluso 100 veces mayor que en sistemas convencionales de tratamiento. Es por ello que se requieren mayores superficies para tratar el mismo caudal de agua que en los sistemas convencionales (Tabla 1.1). Estos sistemas comportan un menor coste de explotación (operación y mantenimiento) ya que se reducen los costes energéticos y de personal. De hecho, en estos sistemas los requerimientos de energía suelen ser entre 5-10 veces menores que en tratamientos convencionales (Kadlec and Wallace, 2009). Además, siendo sencillos de operar, no requieren de personal especializado para su mantenimiento (García, 2004).

Los humedales construidos constituyen una de las alternativas no convencionales más usadas en España para el tratamiento del agua residual de pequeñas comunidades (Puigagut et al., 2007), si bien en muchos casos se recurre a la combinación de diversas tecnologías. La construcción de sistemas híbridos, que combinan humedales construidos, lagunaje y/o filtros de arena, por ejemplo, permite alcanzar de forma fiable el nivel de calidad en los efluentes exigido por la ley (Rousseau et al., 2004).

Tabla 1.1 Requerimiento de superficie por habitante-equivalente para distintos tratamientos de agua residual urbana.

Tipo de tratamiento	Superficie requerida (m^2/hab-eq)
Fangos activos[1]	0.06
Filtros percoladores[1]	0.40
Laguna facultativa[2]	9 – 27
Laguna aeróbica[2]	3.5 – 7
Humedal de flujo superficial[3]	5.6
Humedal de flujo subsuperficial vertical[1]	2.0
Humedal de flujo subsuperficial horizontal[4]	4.5

Área requerida por el reactor biológico para la eliminación de DBO de un afluente de 60 g DBO/hab-eq·d, teniendo en cuenta que el caudal generado por un hab-eq es de 200 L/d.
[1] Cooper, 2005; [2] USEPA, 1983; [3] Kadlec, 2009; [4] Vymazal, 2005

Un humedal construido es un sistema pasivo de depuración constituido por una laguna o canal poco profundo (generalmente de 0,4 a 0,9 m) excavado en el terreno, impermeabilizado y plantado con vegetales propios de zonas húmedas en el que los procesos de descontaminación son ejecutados simultáneamente por componentes físicos, químicos y biológicos a medida que el agua circula a través del mismo.

Existen dos tipos de humedales construidos, los humedales de flujo subsuperficial (SSF CWs) y los de flujo superficial (FWS CWs). La diferencia primordial entre ambos se basa en que en estos últimos el agua está en lámina libre mientras que en los SSF CWs el nivel de agua se encuentra bajo un sustrato (Kadlec et al. 2000).

Los humedales construidos se diseñan para una amplia gama de aplicaciones. Así, se pueden utilizar para tratar aguas residuales domésticas e industriales, lixiviados de vertederos, aguas de escorrentía agrícola y urbana y para el tratamiento de lodos de depuradora (García, 2004; Kadlec and Wallace, 2009).

Los humedales de flujo superficial (Figura 1.3) son sistemas someros (<0.4 m) generalmente utilizados para el tratamiento terciario del agua residual. La configuración típica de este tipo de humedales los hace especialmente apropiados para proyectos de restauración ambiental (Kadlec and Knight, 1996). Ejemplos de este tipo de actuación en Catalunya los encontramos en Granollers (García and Domingo, 2006) y en la EDAR de Empúriabrava (Seguí et al., 2009), en la que el tratamiento terciario del agua se lleva a cabo por humedales construidos. En el caso de Empúriabrava estos humedales se encuentran situados dentro del Parque Natural de els Aiguamolls de l'Empordà.

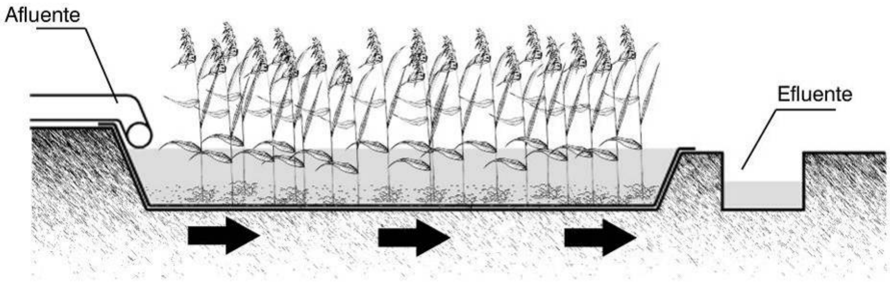

Figura 1.3 Esquema de un humedal de flujo superficial. Las flechas indican el sentido del flujo.

1.2 Humedales construidos de flujo subsuperficial

Los humedales de flujo subsuperficial se construyen tanto para el tratamiento secundario como terciario del agua residual. El agua circula subsuperficialmente por un medio filtrante que ofrece una superficie considerable para el crecimiento de los microorganismos encargados de la depuración. Por ello este tipo de sistemas necesitan áreas menores que los humedales superficiales para tratar el mismo caudal de agua (Tabla1.1). Atendiendo a la dirección del flujo de agua en los humedales subsuperficiales, hablamos de humedales de flujo vertical u horizontal. Cada una de estas tipologías está esquematizada en la Figura 1.4.

Los humedales de flujo vertical (Figura 1.4A) se componen de un lecho impermeabilizado y relleno de varias capas de grava, con un tamaño de grano generalmente creciente con la profundidad del lecho. En las configuraciones habituales, la capa más profunda está compuesta de gravas gruesas (20-40 mm) para facilitar el drenaje, una capa intermedia de grava fina (3-10 mm) que constituye el grueso del humedal y finalmente la capa superficial compuesta de arena (0.25-0.40 mm). El agua se distribuye superficial y uniformemente sobre la capa de arena mediante tuberías formando una red y circula verticalmente por gravedad hasta el fondo, donde otra red de tuberías recoge el agua tratada. Estos humedales se alimentan a pulsos de modo que la operación en *batch* permite la aireación del medio granular (García and Corzo, 2008).

Los SSF CWs horizontales son el objeto de estudio de esta tesis. El esquema típico de un humedal de flujo horizontal (Figura 1.4B) es un lecho impermeabilizado y relleno de un material granular donde el agua circula horizontalmente de un extremo al otro del sistema. Una tubería perforada o un canal de distribución que ocupa la anchura del lecho distribuye el afluente en la entrada del mismo. El agua avanza por gravedad hasta el otro extremo. Ahí, una tubería perforada colocada en el fondo del lecho recoge el agua tratada. El colector de salida es regulable en altura, lo que permite ajustar el nivel del agua dentro del humedal. Al contrario que los sistemas verticales este tipo de humedales operan inundados permanentemente, por lo que el ambiente para la depuración es mucho más reductor que en los sistemas verticales.

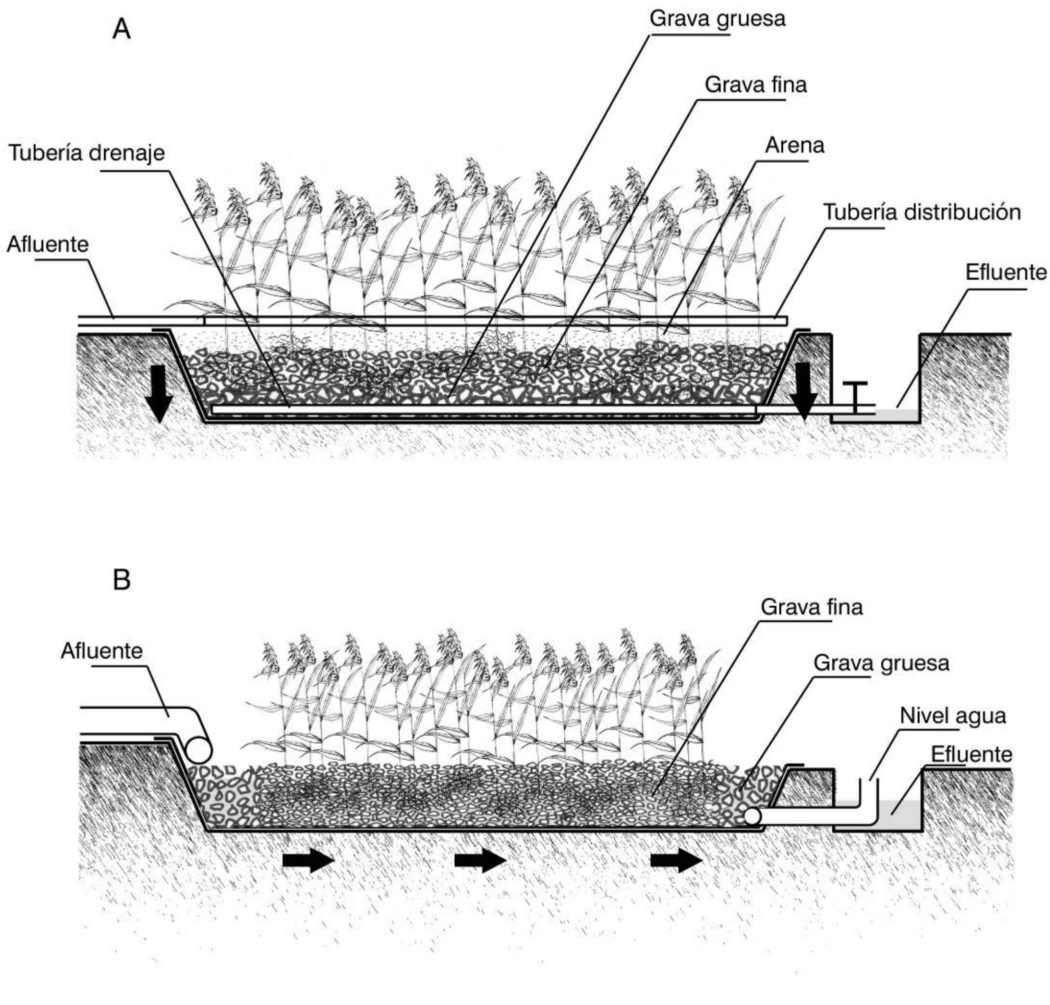

Figura 1.4 Esquemas de humedales subsuperficiales. A) de flujo vertical y B) de flujo horizontal horizontal. Las flechas indican el sentido del flujo.

1.2.1 Mecanismos de eliminación de contaminantes convencionales en humedales de flujo subsuperficial

Los SSF CWs son esencialmente reactores biológicos de tratamiento de agua residual. No obstante, en ellos la eliminación de contaminantes se lleva a cabo por mecanismos tanto biológicos como físicos y químicos. Los rendimientos de eliminación de contaminantes son del mismo orden que el de las tecnologías convencionales (Tabla 1.2).

Tabla 1.2 Rendimientos de eliminación (%) de distintos parámetros de calidad del agua residual en SSF CWs. Las eficiencias de eliminación mostradas están basadas en concentración.

Parámetro	SSF CWs verticales[1]	SSF CWs horizontales[2]
SST	91 – 97	77 – 86
DBO_5	89 – 99	80 – 93
DQO	–	64 – 82
NT	23 – 63	24 – 56
Amonio	76 – 98	25 – 73
PT	25 – 64	27 – 58

[1] Brix and Arias, 2005; [2] Vymazal, 2002

Sólidos en suspensión

La eliminación de sólidos en suspensión del agua se produce básicamente por mecanismos físicos. Esto es, sedimentación de la materia particulada, floculación por agregación de pequeñas partículas que acaban sedimentando y filtración a través de la grava y raíces y rizomas de las plantas. Las elevadas eficiencias en la eliminación de sólidos en suspensión se deben fundamentalmente a que el agua circula a baja velocidad a través del lecho (USEPA, 2000). Una importante fracción de estos sólidos (más del 50%) se eliminan en el primer tercio del humedal (Caselles-Osorio et al., 2007; Trang et al., 2010).

Los sólidos retenidos en el medio filtrante mineralizan con el tiempo por procesos de biodegradación que llevan a cabo los microorganismos que crecen en el lecho (Tanner et al, 1998). Del balance entre la carga de sólidos externa (presente en el afluente) e interna (proveniente del crecimiento bacteriano y la deposición de detritos de plantas entre otros) y las pérdidas por exportación y descomposición, se puede calcular teóricamente la acumulación neta de materia orgánica en un humedal (Tanner et al., 1998), factor que contribuye a la colmatación del lecho. Esta cuestión se desarrollará más adelante.

Materia orgánica

La materia orgánica es la principal fuente de carbono para el desarrollo de comunidades microbianas, y por lo tanto se elimina en definitiva por procesos biológicos. La eliminación ocurre debido a procesos tanto aeróbicos como anaeróbicos, si bien estos últimos parecen tener mayor importancia en los humedales de flujo horizontal (Calheiros et al., 2009; Kadlec and Wallace, 2009). Dado que éstos trabajan generalmente con un aporte continuo de afluente y están permanentemente inundados, la capacidad para oxigenar el lecho se ve muy limitada en comparación a los humedales de flujo vertical, cuya alimentación combina fases saturadas e insaturadas del lecho, lo que proporciona una mayor aireación del lecho.

Procesos anaeróbicos envueltos en la eliminación de materia orgánica, como la metanogénesis, la sulfato-reducción o la desnitrificación dependen de la temperatura, por lo que las eficiencias de eliminación de materia orgánica pueden, en teoría, verse afectadas en periodos fríos (USEPA, 2000). Aunque generalmente esta ralentización en el metabolismo no se ha reflejado en las eficiencias de eliminación de DQO o DBO (García et al., 2005; Ruíz et al., 2008; Serrano et al., 2009; Vymazal and Kröpfelová, 2009). Huang et al. (2005) encontraron diferencias estacionales en la eliminación de los ácidos acético e isovalérico no relacionadas con la carga afluente. Los autores atribuyeron esta estacionalidad a la descomposición de sólidos en suspensión retenidos en los espacios intersticiales de la grava en periodos cálidos, lo que aumentaría la generación interna de materia orgánica. De este modo el efecto de la temperatura en la eliminación de DBO queda enmascarado.

Nitrógeno

El nitrógeno orgánico se elimina básicamente por procesos de amonificación (desaminación de proteínas e hidrólisis de urea), mientras que el amonio por nitrificación−desnitrificación (Figura 1.5). No obstante, una pequeña fracción está envuelta em otros procesos como la asimilación por parte de las plantas y microorganismos, volatilización de amonio o procesos alternativos como la oxidación anaeróbica del amonio (ANAMMOX) (García et al., 2010).

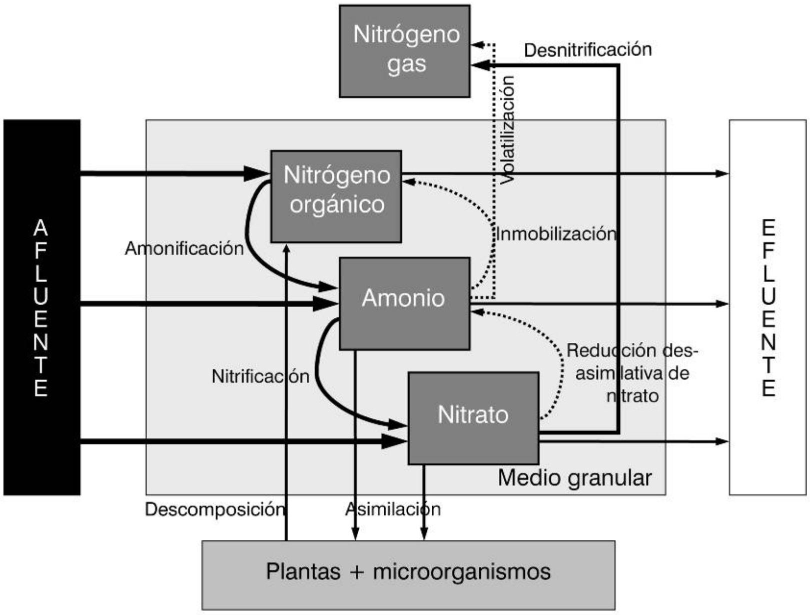

Figura 1.5 Modelo esquemático de los procesos que envuelven la eliminación del nitrógeno en humedales de flujo subsuperficial. Adaptado de Tanner et al. (2002).

La nitrificación (oxidación del amonio a nitrato) es un proceso oxígeno-dependiente que llevan a cabo bacterias nitrificantes, de los géneros *Nitrosomonas* y *Nitrobacter*, en dos etapas (Ecuaciones 1.1 y 1.2). Generalmente se supone que concentraciones de oxígeno mayores de 2 mg/L son las mínimas para no limitar el proceso (Metcalf and Eddy, 2003). No obstante, nótese que en humedales subsuperficiales la concentración suele estar muy por debajo de este valor (García et al., 2004).

$$2NH_4^+ + 3O_2 \xrightarrow{Nitrosomonas} 2NO_2^- + 4H^+ + 2H_2O \quad (1.1)$$

$$2NO_2^- + O_2 \xrightarrow{Nitrobacter} 2NO_3^- \quad (1.2)$$

Por otro lado, la desnitrificación (conversión de nitrato a nitrógeno gas) la llevan a cabo bacterias desnitrificantes (que incluyen varios géneros de bacterias heterótrofas) y en ausencia de oxígeno (en condiciones anóxicas o anaeróbicas). En los humedales encontramos diferentes fuentes de carbono para la desnitrificación, por que la ecuación 1.3 describe el proceso en estos sistemas (Kadlec and Wallace, 2009). En aguas residuales urbanas esta reacción se ve limitada por la nitrificación y por la cantidad de materia orgánica presente (se necesitan 3.02 g de materia orgánica expresada como DQO para reducir 1 g NO_3^--N).

$$84.8NO_3^- + (CH_2O)_{106}(NH_3)_{16}(H_3PO_4) \xrightarrow{Pseudomonas} 42.4N_2 + 106CO_2 + 16NH_3 + 148.8H_2O + H_3PO_4 \tag{1.3}$$

Fósforo

El fósforo es un nutriente de difícil eliminación en tratamientos biológicos por lo que las depuradoras convencionales suelen recurrir a tratamientos químicos para aumentar la eficiencia del tratamiento. Los mecanismos principales de eliminación del fósforo en humedales construidos son la adsorción y la precipitación química por combinación con hierro, aluminio y calcio, elementos generalmente procedentes del material granular del lecho. Así pues, la eliminación de este elemento depende en gran medida de la composición del material filtrante del humedal y su capacidad para acumular fósforo (Arias and Brix, 2005). Por esta razón algunos estudios han evaluado el uso potencial de materiales alternativos, como conchas de mar, grava calcárea, o Filtralite®, que puedan incrementar las eficiencias de eliminación del fósforo en sistemas naturales de tratamiento (Zhu et al., 1997; Brix et al., 2001). En otros estudios se estima la posible utilización de residuos industriales como cenizas bituminosas, que han resultado ser eficientes en la eliminación de fósforo (Cheug and Venkitachalam, 2000; Grubb et al., 2000).

Sin embargo la eliminación de fósforo por adsorción al medio filtrante está limitada, no sólo por la capacidad del material para eliminarlo, sino también por otros condicionantes como el crecimiento de biofilm alrededor de la grava, el tiempo de residencia del agua en el lecho, la concentración afluente o el régimen hidráulico (Arias and Brix, 2005). La liberación de fósforo en el humedal depende del potencial redox del sistema, afectado en gran medida por el régimen hidráulico (Caselles-Osorio and García, 2007a). En condiciones anaeróbicas el fósforo ligado a hierro puede liberarse en forma de fosfatos por la reducción de Fe^{3+} a Fe^{2+} (Volha et al., 2007).

1.2.2 Factores de diseño y operación que influyen en la eliminación de contaminantes

La concentración de contaminantes en el efluente de un humedal depende en gran medida de la concentración afluente y la carga hidráulica aplicada al sistema (Trang et al., 2010). No en vano la carga orgánica a es uno de los factores más importantes a tener en cuenta en el dimensionamiento de un humedal (García and corzo, 2008). Por ello se recomienda diseñar el humedal para una carga orgánica superficial inferior a 6 g DBO/m^2d para los de flujo horizontal (García et al., 2005; USEPA, 2000) y entre 20-40 g DBO/m^2d para los de flujo vertical (Cooper, 2003), con el objeto de obtener efluentes con concentración máxima de DBO de 30 mg/L. Los humedales construidos se consideran idealmente reactores de flujo en pistón en los que la eliminación de contaminantes sigue un modelo cinético de primer orden (Kadlec and Knight, 1996), por lo que para su dimensionamiento se suele usar ecuaciones como la 1.4:

$$A = \frac{Q}{K}\ln\left(\frac{C_e - C^*}{C_i - C^*}\right) \tag{1.4}$$

donde,

A es la superficie del humedal, en m^2

C_e es la concentración en el efluente, en mg/L

C_i es la concentración en el afluente, en mg/L

C^* es la concentración de fondo, en mg/L

Q es el caudal, en m^3/d

K es la constante cinética de primer orden, en m/d.

El oxígeno es un elemento de crucial importancia en los procesos biológicos de descontaminación. En algunos casos el oxígeno constituye un factor limitante para las reacciones que se llevan a cabo. La transferencia de oxígeno de la atmósfera al interior del lecho se produce por tres vías principales: 1) transferencia directa en la superficie del agua por difusión, 2) transferencia mediada por las plantas y 3) oxígeno disuelto en el agua residual (Kadlec and Wallace, 2009). La última es despreciable y las otras dos, en sistemas horizontales, presentan tasas de transferencia comprendidas entre 0.3-3 g O_2/m^2d (Tyroller et al., 2010), muy por debajo de las tasas necesarias para oxidar la materia orgánica de un afluente estándar. Por lo tanto, aquellos parámetros de diseño y operación que repercutan en un aumento o disminución del oxígeno disponible o en el estado de óxido-reducción del sistema promoverán o limitarán el nivel de depuración en el mismo. En la Figura 1.6 se esquematiza las condiciones ambientales y los procesos que se dan en una escala de potencial redox.

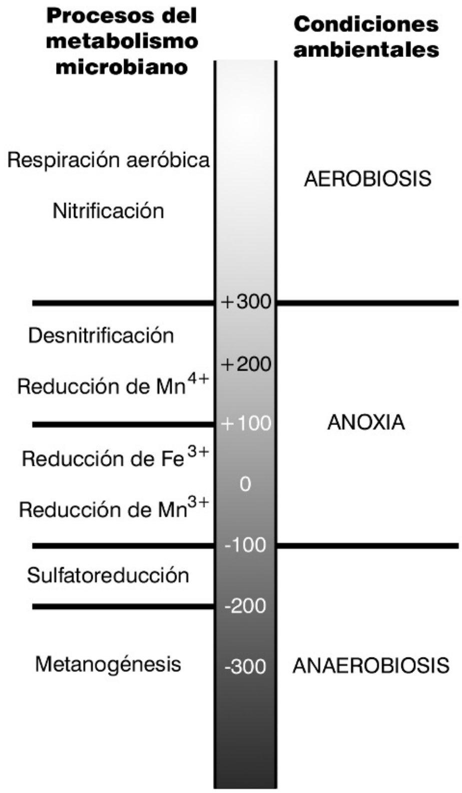

Figura 1.6 Procesos metabólicos típicos de humedales construidos según el potencial redox del medio (adaptado de Kadlec and Wallace, 2009).

La profundidad del lecho es uno de los factores que afectan a la eliminación de contaminantes en sistemas horizontales (Faulwetter et al., 2009). Humedales someros (nivel de agua entre 0.2-0.35 m) presentan condiciones más oxidadas que los lechos comúnmente utilizados (nivel de agua entre 0.5-0.7), lo que permite una mayor eficiencia de eliminación de contaminantes (García et al., 2004, 2005; Matamoros et al., 2005). El potencial redox necesario para la nitrificación (el potencial redox se encuentra en el rango de estados anóxicos y anaeróbicos en humedales profundos) puede darse en sistemas someros llegando a aumentar las típicas eficiencias de eliminación de amonio de un 40-55% a más del 80% (Austin and Nivala, 2009; Vymazal and Kröpfelová, 2009; Caselles-Osorio and García, 2006, 2007a, 2007b).

Otra cuestión a tener en cuenta es el modo de alimentación del humedal. Caselles-Osorio and García (2007a) demostraron que alimentar un humedal de flujo horizontal de manera intermitente, aún cuando el humedal esté permanentemente inundado, permite alcanzar mayores eficiencias de eliminación que el modo continuo en que generalmente se operan estos sistemas. Los humedales de flujo vertical no sólo se alimentan a pulsos sino que además alternan fases saturadas e insaturadas del lecho. Las fluctuaciones en el nivel de agua en sistemas operados de este modo repercuten en estados de oxidación más favorables para la eliminación de la materia orgánica por vías aeróbicas que las conseguidas con una alimentación continua (Breen, 1997; Tanner et al., 1999; Stein et al., 2003; Vymazal and Masa, 2003).

También la granulometría del material filtrante puede tener consecuencias en el nivel de depuración. Puesto que los sólidos en suspensión se eliminan especialmente por filtración, cuanto menor es el tamaño de partículas del medio granular tanto mejor será la eliminación de materia en suspensión del agua residual. Además, una grava más fina facilita el enraizamiento de las plantas, potenciando los efectos de éstas en la eficiencia de eliminación (García et al., 2005). Sin embargo, el uso de materiales finos como medio filtrante del lecho conduce a una prematura colmatación del sistema. Este tema se tratará más ampliamente en las subsecuentes secciones.

1.3 La colmatación en humedales construidos de flujo subsuperficial horizontal

La colmatación es el fenómeno por el cual el medio filtrante del humedal va perdiendo progresivamente las características hidráulicas iniciales, esto es, la disminución de porosidad y la conductividad hidráulica. Cuando la colmatación es muy severa, el lecho no permite al agua infiltrarse, por lo que el sistema disminuye su capacidad depurativa. Luego la colmatación es el principal factor limitante de la vida útil de un humedal. Aunque un humedal se diseña para operar durante décadas, un mal diseño u operación del sistema puede reducir la vida útil a pocos años.

La zona de entrada del agua es la primera en sufrir esta colmatación ya que es donde se produce principalmente la retención de los sólidos en suspensión el agua (Figura 1.7). No obstante, la acumulación de sólidos provenientes del agua residual no es la única causa de colmatación de un lecho. El crecimiento del biofilm, la deposición de detritos de plantas, el crecimiento de raíces y rizomas de éstas y la precipitación química de compuestos son otros motivos del progreso de la colmatación (Brix, 1997; Tanner et al., 1998; Nguyen, 2000; García et al, 2007).

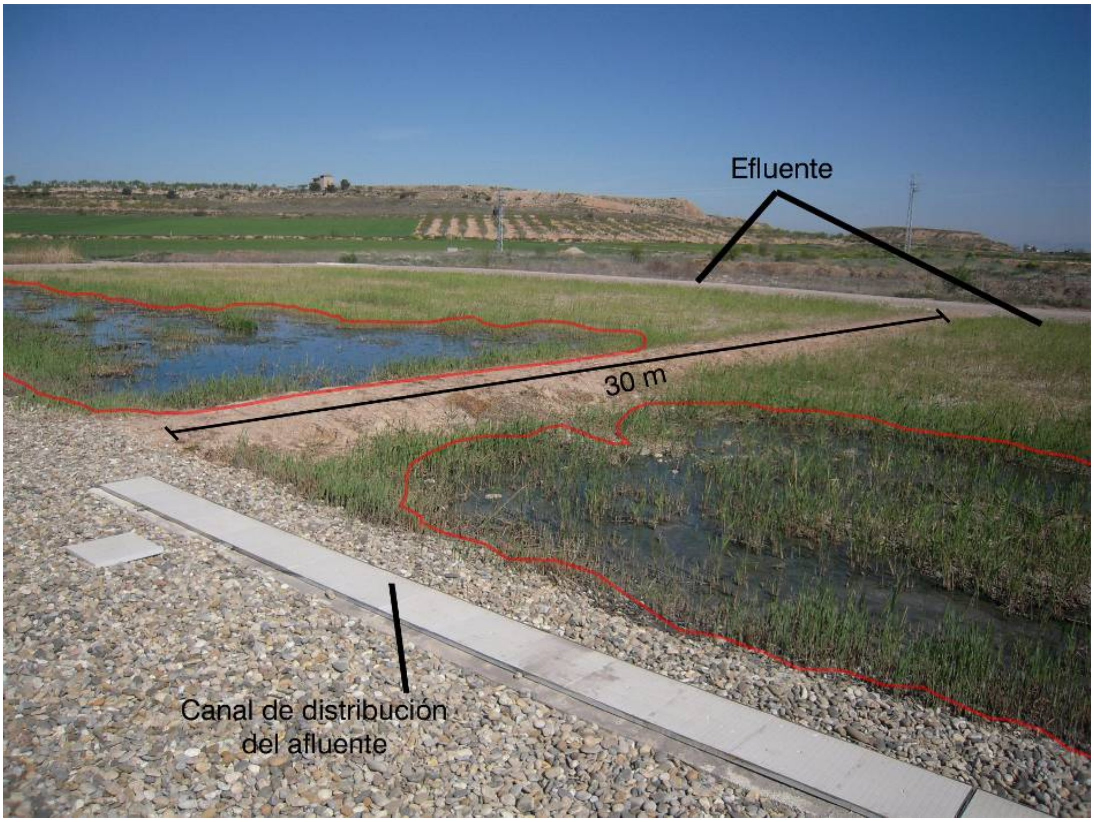

Figura 1.7 Imagen de dos humedales horizontales en que se observa encharcamiento en la zona de entrada (marcada con el trazo rojo).

1.3.1 Agentes de la colmatación

Como ya se ha comentado, los sólidos (lodo) que se acumulan en los espacios intersticiales de la grava y causantes de la colmatación de ésta son de diversa naturaleza. Éstos se pueden clasificar en dos grandes grupos, aquellos de contribución externa y los de producción endógena en el humedal (Tanner et al., 1998). La acumulación neta de lodo en un sistema resulta del balance entra la contribución externa de materia orgánica e inorgánica, la contribución interna (por crecimiento de biofilm y deposición de detritos de plantas) y las pérdidas debidas a la descomposición y exportación de sólidos en el lecho por lavado (Figura 1.8).

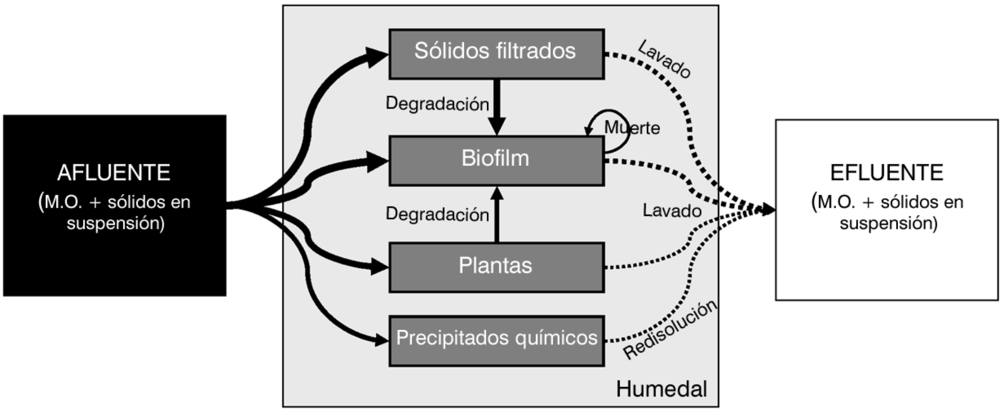

Figura 1.8 Modelo esquemático de procesos que conducen a la acumulación neta de sólidos en un humedal subsuperficial.

Agentes de contribución externa

Los sólidos en suspensión contenidos en el agua residual a tratar es el principal factor de contribución externa a la colmatación. Estos sólidos se acumulan en el lecho por filtración a través de la grava y de las raíces y rizomas de las plantas, entendiendo por filtración, de forma práctica, todos aquellos procesos que conducen a la retención de estos sólidos, esto es el impacto por la inercia del flujo con el medio, la adhesión a la grava y partes subterráneas de las plantas y la sedimentación o decantación.

El agua de entrada a un humedal contiene sólidos fundamentalmente orgánicos pero también inorgánicos (Tabla 1.3), lo que determina la biodegradabilidad de la fracción particulada del agua. La importancia de la biodegradabilidad de estos sólidos radica en que cuanto más biodegradables sean más fácilmente se podrán eliminar del lecho.

Tabla 1.3 Caracterización típica de un afluente primario.

Parámetro	Unidades	EUA[1]	España[2]
DBO	mg/L	129 – 147	23 – 323
DQO	mg/L	310 – 344	92 – 602
SST	mg/L	44 – 54	133 – 213
SSV	mg/L	32 – 39	-
NT	mg/L	41 – 49	25 – 93
Amonio	mg/L	28 – 34	35 – 67
Nitrato	mg/L	0 – 0.9	-
PT	mg/L	12 – 14	-
Coliformes fecales	log/100 mL	5.4 – 6.0	-

[1] USEPA, 2000; [2] Puigagut et al., 2007.

Esta contribución externa de sólidos es inevitable puesto que va asociada a la función principal del humedal, la depuración del agua y por tanto la eliminación de sólidos en suspensión y de materia orgánica. No obstante se puede limitar la entrada de sólidos en suspensión mediante el dimensionamiento del lecho. Generalmente, se recomienda una carga orgánica límite de 6 g DBO/m^{2}d (USEPA, 2000; García et al., 2005) y una carga de sólidos de 20 g SST/m^{2}d como máximo a fin de conseguir las eficiencias de eliminación esperadas. Sin embargo, cargas superficiales de sólidos inferiores a ese valor han demostrado evidenciar síntomas de colmatación temprana (Caselles-Osorio et al., 2007). Por lo tanto es necesario diseñar los humedales estableciendo también la carga de sólidos a aplicar en la entrada para, por lo menos, impedir la colmatación prematura del lecho debido a una sobrecarga.

Agentes de contribución interna

Dentro de los agentes internos que contribuyen a la colmatación de un humedal de flujo subsuperficial destacan el crecimiento de las plantas, el desarrollo del biofilm y la deposición de precipitados químicos.

Las plantas constituyen el soporte físico, además de la grava, para el crecimiento de los microorganismos y por lo tanto el desarrollo de una biopelícula o biofilm (el principal encargado de la depuración del agua). La contribución de las plantas al funcionamiento de los humedales construidos no es despreciable (Brix, 1997), como tampoco lo es su aportación a la colmatación del lecho. Y es que el crecimiento de las plantas implica tanto el crecimiento de la parte aérea como de la subterránea. La senescencia de las plantas produce la deposición de detritos en el interior del lecho (por acumulación de raíces y rizomas marchitos) así como la deposición de materia orgánica en superficie.

Todo esto conduce al desarrollo de una capa impermeable de sólidos en la superficie que impide la infiltración del agua en el lecho y que puede llegar a ser muy importante, sobre todo en la entrada del humedal (Figura 1.9). En algunos casos la contribución de estas deposiciones de materia orgánica puede llegar a ser más importante que la contribución externa por parte de los sólidos en suspensión del agua residual (Tanner and Sukias, 1995; Tanner et al., 1998).

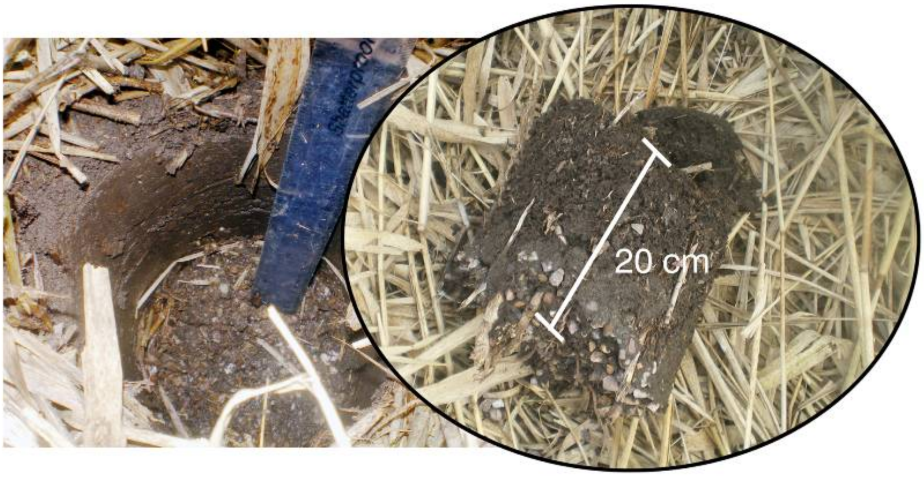

Figura 1.9 Testigo extraído de los 20 cm superficiales a la entrada de un humedal de flujo subsuperficial horizontal. Nótese la capa superficial de lodo que impide la infiltración.

Sumado a este efecto, el desarrollo de la parte subterránea de las plantas puede tener implicaciones en el funcionamiento hidráulico del humedal, afectando al flujo del agua en su interior. En primer lugar porque el desarrollo de la parte subterránea implica una pérdida de volumen efectivo (ocupado ahora por las raíces). La conductividad hidráulica del medio en sistemas de características similares es mayor en lechos no plantados que en humedales plantados (Sanford et al., 1995). Edwards (1992) estimó el volumen ocupado por las raíces en un 5% del volumen de sustrato total de un humedal de flujo subsuperficial (45 cm de profundidad) plantado con juncos (*Scirpus validus*), teniendo en cuenta que la penetración de las raíces fue entre 12 – 15 cm de profundidad. Sin embargo, la disminución del tiempo de retención está muy ligada a la fracción ocupada por raíces (USEPA, 2000).

En segundo lugar, el crecimiento de raíces y rizomas, por un lado, crea una zona de mayor resistencia al paso del agua (principalmente allí donde se concentra el crecimiento de plantas, generalmente en los 20 cm superficiales del humedal) que acarrea el establecimiento de caminos preferenciales de agua. Diversos autores han comprobado, mediante estudios de trazadores, un mayor flujo de agua concentrado en el fondo del humedal (Bowmer, 1987; Fisher, 1990; Drizo et al., 2000). Por otro lado, Chazarenc et al. (2007) encontraron tiempos de retención hidráulica mayores en humedales de flujo vertical plantados que en lechos sin plantar y que atribuían a la retención de agua en la parte más superficial (donde hay una mayor concentración de lodo y detritos de vegetales) y a la evapotranspiración mediada por las plantas. Así pues, el efecto de las plantas en la colmatación no es nada despreciable.

Otro factor interno de colmatación es el desarrollo del biofilm. Los microorganismos que crecen a expensas de la materia orgánica del agua residual se asocian formando una biopelícula que suele ser más importante en la zona de entrada del humedal (Ragusa et al., 2004; Tietz et al., 2007) debido a una mayor carga orgánica disponible y por lo tanto mayor fuente de carbono para su crecimiento. El desarrollo de esta biopelícula depende en gran medida de la calidad de la materia orgánica del agua. Así, Caselles-Osorio and García (2006) detectaron una conductividad hidráulica menor en un lecho alimentado con agua sintética a base de glucosa que en su homólogo alimentado con agua a base de almidón. Atribuyeron la mayor pérdida de conductividad en el lecho alimentado con glucosa a un mayor crecimiento del biofilm, que se desarrollaría mejor con una fuente de materia orgánica más fácilmente biodegradable (glucosa). Por otro lado el crecimiento de la biopelícula se ve favorecido en ambientes aeróbicos (Chazarenc et al., 2009), factor a tener en cuenta en humedales horizontales muy someros, en los que el potencial redox es claramente mayor que en aquellos más profundos (García et al., 2003).

Finalmente, la precipitación química de compuestos en el interior del humedal puede contribuir de forma significativa al desarrollo de la colmatación. Dependiendo de la composición del agua residual y de las condiciones ambientales del lecho se puede producir con cierta facilidad la combinación de sustancias que generen compuestos insolubles y que precipitan y pasan a formar parte del lodo del humedal. En este sentido, Nivala (2005) encontró importantes acumulaciones de precipitados de hierro y causantes de una mayor colmatación en el primer cuarto de un humedal. Ambientes anaeróbicos (con potenciales redox por debajo de -100 mV) puede favorecer la combinación de compuestos del agua residual con el calcio, el aluminio y/o silicatos presentes en el medio granular que pueden ocasionar formaciones espesas que enquistadas en los espacios intersticiales de la grava pueden llegar a soldarla.

1.3.2 Colmatación diferencial según la configuración del humedal

Aunque el concepto de colmatación es sencillo, se trata de un proceso en el que intervienen numerosos factores, en gran medida relacionados con el diseño y operación del sistema. La retención de sólidos en la entrada del humedal está muy relacionada con la configuración en la entrada de éste. Otros factores, concernientes al diseño, construcción y operación de los humedales se discutirán más detalladamente en secciones posteriores.

En los humedales de flujo horizontal podemos diferenciar dos avances característicos de la colmatación según el tipo de alimentación. Así, en los humedales alimentados superficialmente (esta es la configuración más típica en sistemas españoles y del Reino Unido) la colmatación se desarrolla inicialmente en la parte superior de la entrada del lecho. Esto resulta en una capa colmatada en la entrada que impide la infiltración del agua, provocando el flujo superficial del agua hasta alcanzar zonas del lecho menos colmatadas (Figura 1.10). En estados severos de colmatación se puede observar un deficiente crecimiento del carrizo,

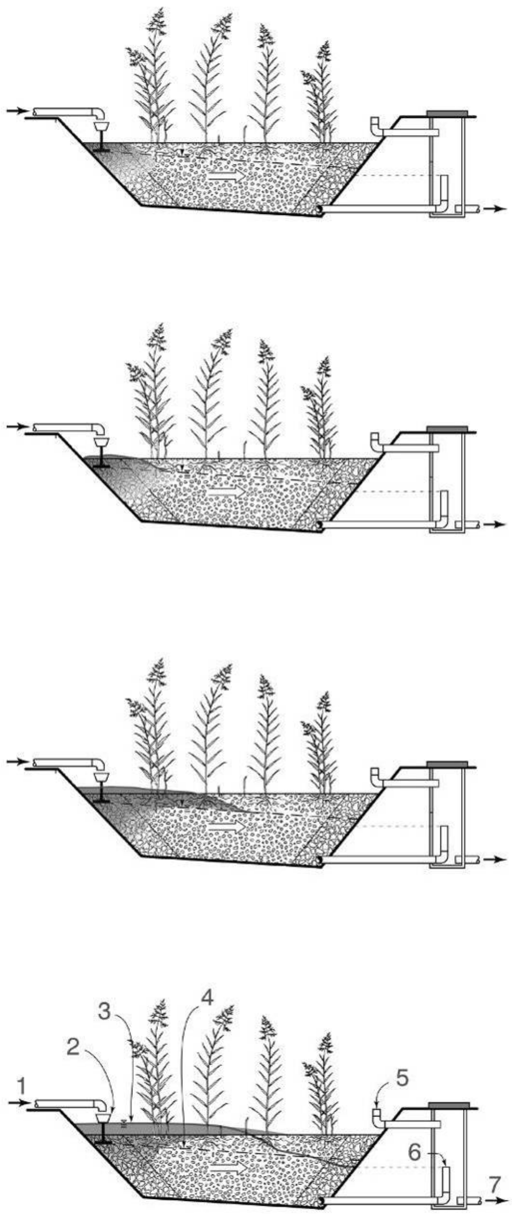

Figura 1.10 Desarrollo de la colmatación en un humedal de flujo horizontal con alimentación superficial (adaptado de Knowles et al., 2010). 1 Afluente; 2 Tubería de distribución superficial; 3 Nivel del agua en respuesta a la colmatación; 4 Nivel de agua teórico (subsuperficial); 5 Aliviadero; 6 Tubería ajustable; 7 Efluente.

infestación por malas hierbas y un encharcamiento global en el lecho (Cooper et al., 2005; Rousseau et al., 2005; Cooper et al., 2008). En algunos casos esta acumulación de sólidos puede llegar a formar capas de lodo tan espesas que asemejan a un suelo (Figura 1.9).

Por otro lado, los humedales de flujo horizontal con alimentación subsuperficial, más comunes en Estados Unidos y Australia, reciben el agua por tuberías subterráneas (unos centímetros por debajo de la superficie) por lo que el patrón de colmatación es distinto, al menos en estadios iniciales del tratamiento (Figura 1.11). En este caso la colmatación se desarrolla subsuperficialmente, y como es lógico alrededor de la tubería de distribución (Wallace and Knight, 2006; Chazarenc et al., 2007). La obstrucción de esta zona obliga al agua a emerger a la superficie para volverse a infiltrar más adelante como ocurría con los humedales con carga superficial. Aunque a la larga ambos sistemas presenten el mismo aspecto la carga subsuperficial parece retrasar la indeseable aparición de charcos.

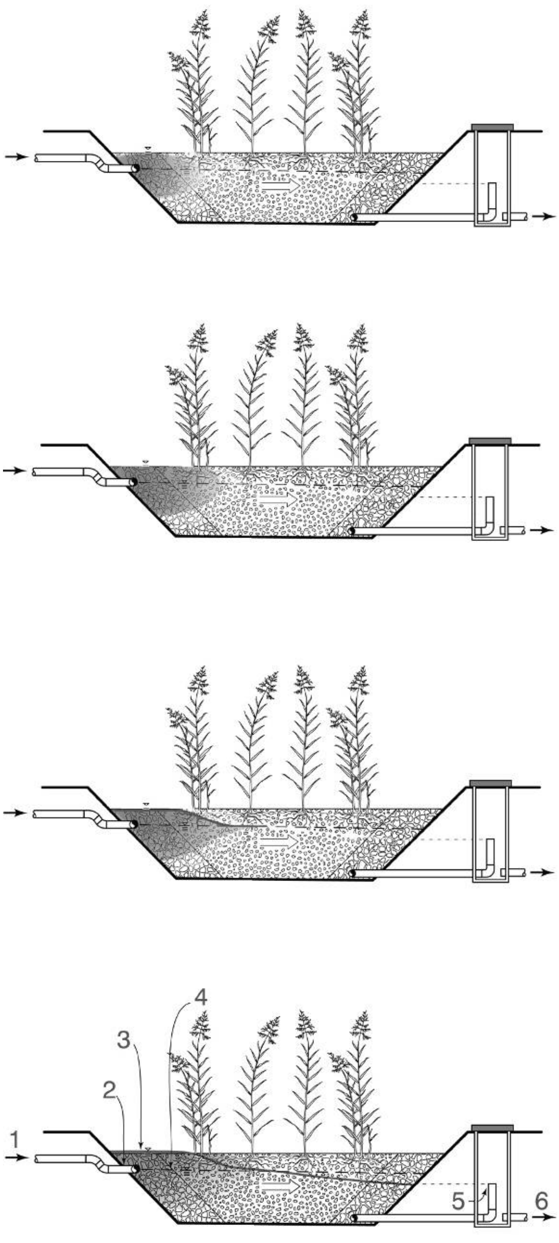

Figura 1.11 Desarrollo de la colmatación en un humedal de flujo horizontal con alimentación subsuperficial (adaptado de Knowles et al., 2010). 1 Afluente; 2 Tubería de distribución subsuperficial; 3 Nivel de agua en respuesta a la colmatación; 4 Nivel de agua teórico (subsuperficial); 5 Tubería ajustable; 6 Efluente.

1.3.3 Indicadores de la colmatación

Existen diversos parámetros que permiten evaluar el grado de colmatación de un humedal. Aunque cada uno de ellos tiene sus propias limitaciones, en general permiten obtener información acerca del estado hidráulico del sistema.

Acumulación de sólidos

Puesto que la colmatación se produce por la acumulación de sólidos de diversa naturaleza, la cuantificación de éstos constituye una medida indirecta de cuán colmatado está el lecho. Se trata de una aproximación porque la naturaleza de los sólidos acumulados parece estar relacionada con las características del lodo que causa la colmatación. Y es que la cantidad de sólidos de una muestra no es directamente proporcional a otras medidas de carácter hidráulico, como la porosidad o la conductividad hidráulica (Tanner et al., 1998; Caselles-Osorio et al., 2007).

Generalmente la cantidad de sólidos (expresada en kg de MS/m^2) es mayor en la zona de entrada que en la salida del humedal. Sin embargo, el crecimiento del biofilm da origen a la formación de un lodo gelatinoso, y por tanto capaz de retener agua, que disminuye considerablemente la porosidad drenable y la conductividad hidráulica (Kadlec and Watson, 1993; Platzer and Mauch, 1997; Suliman et al., 2006a). Del mismo modo, la fracción volátil de los sólidos (comparable a la cantidad de materia orgánica) tiende a disminuir de la entrada a la salida del lecho (Tanner and Sukias, 1995; Tanner et al., 1998; Caselles-Osorio et al., 2007). Con la disminución de sólidos volátiles aumentan la densidad y decantabilidad del lodo (Llorens et al., 2009).

La distribución vertical de los sólidos también presenta un gradiente, siendo generalmente mayor la acumulación en las capas superficiales que en el fondo del lecho. Esto parece motivado por el crecimiento de la parte subterránea de las plantas (generalmente limitado a los 20 cm superiores), que además de ocupar este espacio soporta el crecimiento de microorganismos (Bavor and Shulz, 1993; Chazarenc and Merlin, 2005).

Conductividad hidráulica

La conductividad hidráulica es una medida indirecta de la colmatación que informa acerca de las propiedades hidráulicas del sistema. Se trata de la velocidad de infiltración que permite el medio (expresada en unidades de espacio/tiempo). Así, a menor conductividad hidráulica mayor es la resistencia que ofrece el medio granular al paso del agua y por lo tanto mayor es el grado de colmatación.

Tradicionalmente, la conductividad hidráulica (o permeabilidad) se ha estimado a partir de la medida del nivel del agua en diferentes puntos a lo largo de un humedal. La pérdida de carga entre estos puntos se relaciona con la conductividad por la Ley de Darcy. Sin embargo, este tipo de medidas revelan importantes diferencias entre estudios (Kadlec and Watson, 1993; Sanford et al., 1995;Watson and Choate, 2001). Además, ofrece una conductividad global del lecho y no permite conocer a fondo la distribución de la colmatación.

Idealmente, el mejor modo de medir la conductividad hidráulica es extrayendo un testigo para analizarlo con permeámetros estandarizados. No obstante, el carácter no cohesivo de la grava frustra la extracción de una muestra inalterada por lo que se ha tratado de aplicar métodos de medida *in situ*. Así, se ha utilizado el permeámetro de Guelph (Mastrorilli et al., 2001; Langergraber et al., 2003), un método de permeabilidad de carga variable (NAVFAC, 1986) o de carga constante (Knowles and Davies, 2009). Todos estos métodos usan la ley de Darcy para calcular la conductividad hidráulica. El permeámetro de Guelph y el permeámetro de carga constante se basan en medir el caudal necesario para mantener un nivel constante de agua en una zona de sondeo, aunque el primer método se ha usado generalmente para la medición en suelos. El segundo, en cambio, se diseñó específicamente para medir conductividades en el rango que podemos encontrar en un humedal subsuperficial (entre 0-500 m/d en función del tipo de grava y su estado de colmatación). El permeámetro de carga variable se basa en medir el tiempo que tarda una columna de agua en descender en una zona de sondeo. En todas estas técnicas expuestas la zona de sondeo la constituye una celda que atrapa una porción de medio. Por lo tanto, es inevitable una cierta perturbación del medio, aunque es despreciable comparada con la alteración producida al intentar extraer un testigo.

Porosidad drenable

La porosidad drenable es un parámetro íntimamente ligado al volumen efectivo de un humedal. Es la medida del volumen de agua obtenido al vaciar un lecho referido al volumen de los espacios intersticiales del medio granular. La deposición de sólidos en suspensión junto con el crecimiento del biofilm puede reducir la porosidad inicial de un lecho en más del 50% (Suliman et al., 2006a; Kadlec and Watson, 1993).

El crecimiento de las plantas también resulta importante en la reducción de la porosidad, no sólo porque la parte subterránea ocupa un espacio sino también porque la senescencia de las raíces y rizomas genera un depósito extra de sólidos (Brix, 1997). Estos sólidos junto al lodo acumulado pueden formar mantos consistentes que den cohesión al medio granular, como se ha apreciado anteriormente en la Figura 1.9.

Estudios del comportamiento hidráulico con sustancias trazadoras

Una sustancia trazadora es un compuesto inerte, que no participa de las reacciones que tienen lugar en un humedal. Así la cuantificación de la concentración de un trazador inyectado en un humedal permite conocer el flujo de agua en el mismo. Los trazadores más comúnmente usados son el cloruro de sodio, el bromuro, el litio, la rodamina y la fluoresceína (Bowmer, 1987; Batchelor and Ioots, 1997; García et al., 2003; Knowles et al., 2009).

Mediante la evaluación de la curva del trazador en el efluente se puede determinar el comportamiento hidráulico del sistema. Teóricamente los humedales se diseñan como reactores ideales de flujo en pistón, aunque en la práctica presentan desviaciones de este tipo de flujo. En experimentos de inyección puntual del trazador, la aparición de varios picos de concentración en la curva del efluente permite determinar la presencia de cortocircuitos y zonas muertas así como el tiempo de retención hidráulico real del lecho. La medida de la concentración del trazador en diferentes puntos del humedal (y no sólo en el efluente) permite establecer zonas de flujo preferencial del agua (Knowles et al., 2009). Sin embargo este tipo de experimentos proporciona información acerca de un momento en concreto ya que los resultados pueden diferir enormemente según la época del año en que se ha realizado el ensayo, según el caudal y el trazador usado (Kadlec and Wallace, 2009).

El progreso de la colmatación provoca, por un lado, una disminución del tiempo de retención del agua en el humedal debido a la pérdida de volumen efectivo (Tanner et al., 1999). Además, puesto que la acumulación de sólidos es heterogénea y depende de muchos factores, tanto de diseño como de operación (que se discutirán a continuación) aparecen con facilidad caminos preferenciales de flujo de agua en los que acusa aún más la colmatación.

1.3.4 Factores de diseño y operación que afectan a la colmatación

La colmatación es un fenómeno inevitable en un humedal de flujo subsuperficial ya que la función de éste es tratar un agua cargada en sólidos y materia orgánica. Sin embargo, existen elementos clave en el diseño de estos sistemas que pueden exacerbar el proceso. Por lo tanto es necesario, en primer lugar conocer cuáles son estos factores y en segundo lugar proponer mejoras para retrasar en lo posible la colmatación. Entre los factores más importantes que afectan la colmatación se encuentran la carga orgánica y de sólidos en el afluente, la distribución del agua en la entrada del sistema y las características del medio granular.

Carga afluente

Puesto que la colmatación se da por una acumulación de sólidos en los espacios intersticiales de la grava, es obvio que la carga orgánica y especialmente la carga de sólidos con que se alimenta un humedal es de capital importancia en el diseño de éste. En los primeros estadios de la colmatación, ésta es mucho más importante en la entrada que a medida que se avanza hacia la salida. De hecho, es en la entrada del humedal donde se filtra y retiene la mayor parte de sólidos en suspensión del agua. Por lo tanto, una sobrecarga de sólidos en la entrada promueve el aumento de lodo en esta zona, luego la rápida pérdida de la velocidad de infiltración del agua y la aparición de charcos (Figura 1.7).

En general, la carga de sólidos aplicada en el humedal se relaciona directamente con las tasas de acumulación de sólidos, expresado en kg de materia seca/m^2año (Caselles-Osorio et al., 2007). De este modo, humedales alimentados con elevada carga de sólidos presentan mayores acumulaciones de lodo (Figura 1.12). Sin embargo, así como se consideran 6 g DBO/m^2d la carga orgánica límite para obtener buenos rendimientos de eliminación de contaminantes, no hay un consenso en el límite de carga de sólidos a aplicar para reducir el riesgo de una colmatación prematura del lecho.

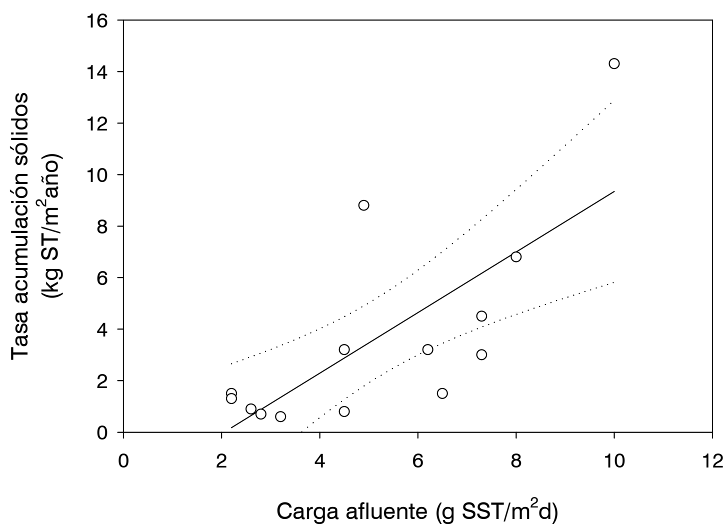

Figura 1.12 Relación entre la carga de sólidos superficial en el afluente y la acumulación de sólidos en el humedal (datos extraídos de Caselles-Osorio et al., 2007; Tanner et al., 1998; Tanner and Sukias, 1995).

Tratamiento primario

En relación con lo expuesto en el punto anterior, y para disminuir la carga de sólidos aplicada a un humedal, los tratamientos previos juegan un papel primordial. Generalmente, en sistemas de humedales como tratamiento secundario del agua, el tratamiento primario lo suelen desempeñar decantadores, fosas sépticas y tanques Imhoff (Puigagut et al., 2007). Estos tratamientos, fundamentalmente físicos, tienen eficiencias de eliminación del 30-40% en DBO y entre 50-60% en sólidos en suspensión (Metcalf and Eddy, 2003). Sin embargo, si están mal diseñados o deficientemente mantenidos pueden contribuir en un indeseable aporte extra de sólidos en la entrada del humedal.

Recientemente se empiezan a considerar otro tipo de tecnologías aplicables como tratamiento primario. Es el caso de los reactores anaeróbicos. Los reactores anaeróbicos de flujo ascendente (UASB) y reactores hidrolíticos de flujo ascendente (HUSB) constituyen buenas alternativas a los tratamientos primarios convencionales ya que producen efluentes con bajas concentraciones de materia orgánica y tienen buenos rendimientos de eliminación de sólidos en suspensión (Barros et al., 2008; Díaz et al., 2008; Dornelas et al., 2008; Álvarez et al., 2008a). Se trata de reactores compactos donde el agua circula de forma ascendente a través de un lecho de lodo mantenido en condiciones anaeróbicas. Las bacterias del interior eliminan materia orgánica soluble y particulada por vías anaeróbicas. En los UASB se lleva a cabo la metanogénesis, mientras que en los reactores HUSB se limita el metabolismo a únicamente la primera fase (hidrolítica) mediante la disminución del tiempo de retención del agua en el reactor. Así, los UASB suelen tener tiempos de retención hidráulicos de entre 8 y 14 horas mientras en los HUSB este tiempo es de 2-5 horas (Álvarez et al., 2008b; Ruíz et al., 2008).

Distribución afluente

Si la acumulación de sólidos en el humedal se produce de manera heterogénea, fácilmente se forman canales preferenciales del paso del agua a través del lecho. Uno de las causas que pueden empeorar el establecimiento de estos flujos preferenciales es una distribución no uniforme del agua en la entrada del humedal. En los humedales horizontales con carga superficial los sistemas de distribución más comúnmente utilizados son tuberías perforadas, tuberías verticales y canales con vertederos en forma de V (Figura 1.13). Teóricamente, cualquiera de estos sistemas permite una buena distribución del agua en la entrada. Sin embargo, las tuberías perforadas se asientan diferencialmente en el terreno provocando zonas con mayor afluencia de agua y las tuberías verticales tienden a obstruirse por acumulación de lodo. Por eso cada vez más se instalan canales. Hasta el momento se juzgan más adecuados puesto que su mantenimiento es más sencillo (Kadlec and Wallace, 2009; Griffin et al., 2008).

Figura 1.13 Sistemas de distribución del afluente. 1 tubería perforada; 2 tubería perforada; 3 tubería vertical; 4 tubería vertical obstruida; 5 canal de distribución; 6 detalle; 7 canal de distribución.

Alternancia o intermitencia en la operación

La manera de operar los humedales puede tener consecuencias en el desarrollo de la colmatación. Así lo sugieren algunos estudios realizados en humedales de flujo vertical. Periodos de reposo en el funcionamiento del humedal permiten la recuperación de parte de la porosidad del medio granular (Platzer and Mauch, 1997; Admon et al., 2002; Langergraber et al., 2003). Esto implicaría disponer de varias celdas de operación para poder aplicar periodos de barbecho.

Alternativas a estos periodos de reposo serían la fluctuación del nivel de agua dentro del humedal o la alimentación intermitente. Estas estrategias permitirían promover los procesos aeróbicos en el lecho, combatiendo así los efectos negativos de la colmatación (Batchelor and Loots, 1997; Nguyen, 2000; Behrends et al., 2001; Zhao et al., 2006). Aunque esto supone cambiar el funcionamiento habitual en continuo de los humedales de flujo horizontal.

Características del medio filtrante

La conductividad hidráulica del medio filtrante está directamente relacionada con el tamaño del árido (Tabla 1.4). Cuanto más fina sea la grava utilizada en el humedal, menor es su conductividad hidráulica inicial y también su porosidad. Luego, a menor tamaño de partícula mayor es la tendencia del lecho a colmatarse.

Tabla 1.4 Conductividad hidráulica (en m/d) de distintos tipos de áridos según su tamaño (Crites and Tchobanoglous, 1998).

Material	Tamaño efectivo D_{10} (mm)*	Porosidad (%)	Conductividad hidráulica (m/d)
Arena mediana	1	30	500
Arena gruesa	2	32	1000
Grava fina	8	35	5000
Grava mediana	32	40	10000
Grava gruesa	128	45	100000

* D_{10} es el tamaño del tamiz por el que pasa el 10% de una muestra del árido

La grava en los humedales de flujo subsuperficial constituye una parte importante del tratamiento ya que a través de ella se produce la filtración y por lo tanto la retención de sólidos en suspensión. También sirve de soporte para el enraizamiento de las plantas y el crecimiento del biofilm encargado de la depuración del agua.

La recomendación en el uso de tamaños de grava ha ido modificándose en los últimos veinte años. El tamaño de grava recomendado por la Agencia de Protección Medioambiental de los Estados Unidos (USEPA) ha ido incrementando de 1-8 mm a 20-30 mm en este tiempo (USEPA, 1993, 2000). Precisamente el uso de arenas o gravas finas limita aún más la vida útil de un humedal por lo que es más recomendable utilizar gravas más gruesas, que no comprometan la eficiencia de eliminación de contaminantes. Actualmente se ha generalizado el uso de gravas con tamaños comprendidos entre 6-11 mm en sistemas de tratamiento terciario en Gran Bretaña (Griffin et al., 2008).

Sin embargo otras consideraciones importantes en elación a las características del medio granular son la distribución granulométrica y la forma de las partículas. Altamente recomendable es el uso de gravas lo más uniforme posible en su distribución de tamaños. Los áridos con grandes proporciones de finos reducen ya inicialmente la conductividad hidráulica, agravando la colmatación. Por otro lado, aquellas gravas cuya composición produce agregados irregulares presentan también porosidades más bajas que las gravas de formas redondeadas. Por todo esto, se recomienda utilizar gravas redondeadas y lavadas previamente para eliminar en la medida de lo posible grandes cantidades de finos.

1.3.5 Estrategias para aliviar la colmatación

Cuando un humedal disminuye su capacidad depurativa debido a la colmatación del lecho es necesario restaurar las condiciones, sino iniciales de operación, sí aceptables para garantizar el tratamiento del agua residual dentro de los límites que establece la ley. En ese sentido, existen diversas alternativas disponibles y que podrían clasificarse en dos grupos, a saber, métodos invasivos y métodos no invasivos.

Los métodos invasivos para remediar la colmatación incluyen la extracción y reposición de la grava (total o parcialmente), y la extracción y lavado de la grava (Cooper et al., 2005; Wallace and Knight, 2006; Murphy and Cooper, 2009). De éstas, la primera alternativa resulta la más costosa económicamente (Tabla 1.5) puesto que implica un importe que puede suponer una fracción importante de la inversión inicial para la construcción del sistema (entre un 10 y un 19%, Kadlec and Wallace, 2009). Esto es debido a que la adquisición y transporte de la grava es uno de los aspectos más caros del presupuesto constructivo de un humedal. En cualquier caso, todas ellas requieren una interrupción en la operación del sistema. En cambio, la aplicación de métodos no invasivos no tiene por qué implicar tal interrupción. Entre ellos cabe destacar la fluidización de lodos y extracción por bombeo, la adición de agentes microbianos comerciales o la aplicación de agentes oxidantes fuertes, como el peróxido de hidrógeno (Behrends et al., 2006; Nivala and Rousseau, 2009).

Tabla 1.5 Coste aproximado (en €/m^2) de diferentes estrategias para remediar la colmatación en humedales construidos de flujo subsuperficial.

Estrategia	Coste (€/m^2)	País	Referencia
Extracción y reposición de la grava	80	Reino Unido	Griffin et al., 2008
Extracción y lavado de la grava	35	Reino Unido	Murphy et al., 2009
Aplicación *in situ* de peróxido de hidrógeno	6	EUA y Bélgica	Nivala and Rousseau, 2009

Extracción y reposición de la grava

La extracción de la grava colmatada de un humedal y reposición con grava nueva es la estrategia más ampliamente utilizada hasta ahora a pesar de ser la más costosa. El presupuesto para esta operación (aproximadamente 80 €/m^2 en Reino Unido) depende de las características del humedal y de las de la nueva grava a utilizar (calidad, cantidad y transporte). La extracción debe hacerse cuidadosamente para evitar el deterioro de la membrana impermeabilizante. En el presupuesto hay que tener en cuenta pues, los trabajos de extracción y la adquisición del nuevo material, pero también la replantación del humedal y disposición de la antigua grava como residuo.

Generalmente se renueva el material filtrante con gravas de tamaño superior a la anterior para evitar que se colmaten tan rápidamente. Sin embargo, hay que tener en cuenta que si la sustitución es parcial se pueden producir con cierta facilidad caminos preferenciales y cortocircuitos en el flujo del agua (Knowles and Davies, 2009).

Extracción y lavado de la grava

La extracción y lavado de la grava es una técnica de reciente aplicación en Reino Unido. La empresa ARM Ltd. (Staffordshire, UK) ha desarrollado una máquina que permite lavar la grava colmatada para luego devolverla al lecho (Figura 1.14). La máquina consta de un tanque de lavado provisto de una trampa de arena, un tambor giratorio y un decantador lamelar. El agua de lavado se devuelve a cabecera de planta para tratarla o bien se transporta a una depuradora mayor.

Esto permite la reutilización de la grava y se evita el coste de adquisición y transporte de la nueva grava. De este modo se consigue un ahorro de hasta un 55% respecto al coste de la extracción y reposición con un material granular distinto (Murphy et al., 2009).

Figura 1.14 Máquina lavadora de grava (ARM Ltd.).

Aplicación in situ de oxidantes químicos

Los métodos no invasivos para aliviar la colmatación resultan una alternativa prometedora puesto que evitan la excavación y extracción del medio granular. Aunque se han utilizado en sistemas a escala real, su práctica es todavía anecdótica.

La aplicación *in situ* de peróxido de hidrógeno concentrado (35%) permite reducir notablemente la concentración de sólidos volátiles (Behrends et al., 2006) y aparentemente no tiene consecuencias negativas para el tratamiento del agua residual a posteriori (Nivala and Rousseau, 2009). Sin embargo, este tipo de intervención requiere más investigación para poder implementarlo con garantías.

2

Objetivos

Los humedales construidos suponen una alternativa al tratamiento convencional de agua residual para pequeños municipios (hasta 2000 hab-eq) debido principalmente a la facilidad en la operación y mantenimiento, y a los reducidos costes de explotación inherentes a este tipo de tecnologías.

A pesar de estas ventajas, el mayor inconveniente al que se enfrentan los explotadores de sistemas de tratamiento con humedales subsuperficiales es la colmatación del lecho. Con el tiempo, la acumulación de sólidos de diversa naturaleza en los espacios intersticiales del medio filtrante provoca la disminución de la conductividad hidráulica y la porosidad iniciales de la grava. Esto conduce al desarrollo de caminos preferenciales y cortocircuitos en el curso del agua que convergen en la aparición de agua en superficie, y a la larga, pueden comprometer la capacidad depurativa del sistema. Por ello, la colmatación supone el factor limitante de la vida útil de un humedal construido.

Las estrategias para solventar la colmatación, una vez se ha producido, son costosas y pasan por realizar inversiones no despreciables. De ahí la necesidad de medir, de manera fiable, en qué grado un sistema está colmatado, identificar los factores que influyen en el fenómeno y indagar en nuevas configuraciones y estrategias que permitan retrasar el avance de la colmatación. Todo ello debe contribuir a aplazar las intervenciones necesarias para devolver al sistema un estado óptimo de funcionamiento.

Por estas razones, los **objetivos** de esta tesis doctoral son los que se desarrollan a continuación:

- Cuantificar la precisión y exactitud de un método de medición in situ de la conductividad hidráullica, para la determinación del grado y la distribución horizontal de la colmatación de un lecho.
- Determinar el indicador más adecuado para evaluar la colmatación de un humedal de flujo subsuperficial horizontal.
- Identificar criterios de diseño y operación de humedales de flujo subsuperficial horizontal susceptibles de causar o potenciar la colmatación del medio filtrante.
- Cuantificar la eficiencia de eliminación de contaminantes convencionales sujetos a un tratamiento primario y un modo de operación alternativo en humedales construidos de flujo subsuperficial horizontal.
- Evaluar la colmatación de dos configuraciones alternativas de humedales horizontales (tratamiento primario y modo de operación).
- Analizar la posible contribución del tipo de grava usada en un humedal horizontal en el desarrollo prematuro de la colmatación.

La consecución de estos objetivos han de contribuir a establecer nuevos criterios de diseño y operación de humedales construidos de flujo subsuperficial horizontal para minimizar, o cuanto menos retrasar, la colmatación de estos sistemas, sin mermar la eficiencia de eliminación de contaminantes del agua residual.

Para alcanzar de estos objetivos se han realizado campañas de muestreo y medición de la conductividad hidráulica en humedales construidos a escala real (Capítulos 3, 4 y 5). Se han comparado los valores de conductividad hidráulica obtenidos con el método desarrollado con otro método de medición tanto en condiciones controladas (en el laboratorio) como en humedales a escala real (Capítulos 3 y 5). También se ha construido una planta de tratamiento de agua residual a escala experimental y se ha operado durante 3 años para evaluar tanto la eficiencia de eliminación de contaminantes como el desarrollo de la colmatación de sus lechos (Capítulos 6 y 7). Finalmente, con distintas muestras de grava obtenidas de los diferentes muestreos se ha analizado su posible contribución al fenómeno de la colmatación (Capítulo 8).

Los Capítulos 3-8 se corresponden en parte con artículos científicos que han sido publicados durante la fase de investigación de la tesis o bien están en fase de revisión. En el Capítulo 9 se desarrolla una discusión general de los resultados y finalmente en el Capítulo 10 se presentan las conclusiones de la tesis doctoral.

3

Reliability of hydraulic conductivity measurements by falling head method for the evaluation of the best design of horizontal subsurface flow wetlands[1]

The aim of this study was to verify the reliability, repeatability and accuracy of the *in situ* measurement of hydraulic conductivity using a falling head method (FHM). Furthermore, the FHM was used to map the degree of clogging in two field-scale horizontal subsurface flow constructed wetlands (SSF CWs) in order to highlight its application to evaluations of the best design and operational management strategies. The FHM can record values of clogged and unclogged systems (i.e. 4 to ca. 360 m/d). Results had deviations of 20% when compared with a laboratory constant head method for highly conductive media (> 250 m/d) and of 80% for media with low hydraulic conductivity (< 50 m/d), although is acceptable considering difficulties in the measurement of hydraulic conductivity. It is a reliable procedure with no significant variations in repeated measurements. The hydraulic conductivity at both field-scale SSF CWs was low (0 to 40 m/day), with the lowest value (1-10 m/day) near the inlet. The use of the FHM for mapping the hydraulic conductivity of field-scale SSF CWs makes it possible to identify key factors that influence clogging, such as abnormalities in influent feed systems and aspect ratio to reduce preferential water flows.

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[1] This chapter is based on the article:

Pedescoll, A., Samsó, R., Romero, E., Puigagut, J., García, J., 2010. Reliability, accuracy and repeatability of the falling head method for hydraulic conductivity measurements under laboratory conditions. Ecological Engineering, doi: 10.1016/j.ecoleng.2010.06.32.

3.1 Introduction

Subsurface flow constructed wetlands (SSF CWs) have been widely used since the early 1970s to improve water quality. This technology represents a good alternative to conventional wastewater treatment plants, particularly in small towns (Vymazal, 2005; Puigagut et al., 2007; Rousseau et al., 2008). However, SSF CWs filter media suffer from porosity reduction over time (Wallace and Knight, 2006). Porosity reduction is the result of clogging and affects all types of SSF CWs, to varying degrees. Clogging leads to hydraulic malfunction that prevents normal operation and, in very severe cases, can affect treatment efficiency (Platzer and Mauch, 1997; Rousseau et al, 2005; Kadlec and Wallace, 2009).

Diverse processes are involved in the development of clogging. The most significant factors are biofilm and vegetation growth within the filter medium, retention of wastewater particles and precipitation of certain chemical compounds (Brix, 1997; Tanner et al., 1998; Nguyen, 2000; García et al, 2007). All these processes act simultaneously over time and their combination causes the aforementioned reduction of porosity.

When clogging is detected, several techniques can be applied to reverse it. However, techniques such as gravel extraction and cleaning, *in situ* application of chemical oxidants (Cooper et al., 2005; Murphy et al., 2009; Nivala and Rousseau, 2009) and, especially, filter medium replacement can be very costly.

For treatment and financial reasons, it is essential to assess the degree of clogging in SSF CWs. Hence, measures of clogging have been given special attention in current literature. Some of the most widespread measures of clogging include the analysis of accumulated solids in filter media (Tanner et al., 1998; Caselles-Osorio et al., 2007), system hydrodynamics by means of tracer tests (Bowmer, 1987) and hydraulic conductivity measurements (Sanford et al., 1995; Rodgers et al., 2006; Suliman et al., 2006a). Specifically, hydraulic conductivity measurements have proven to be a suitable technique (Knowles et al., 2010a).

The measurement of hydraulic conductivity to assess the degree of clogging in SSF CWs is not a straightforward procedure. Difficulties arise because the filter medium of constructed wetlands is gravel, then, of a non-cohesive nature. This makes it virtually impossible to take unaltered samples of material to perform standardized and controlled laboratory tests. For this reason, most of the techniques that are used to study the hydraulic conductivity of constructed wetlands are based on *in situ* procedures (Reynolds et al, 2000). The most commonly applied methods include the Guelph permeameter (Mastrorilli et al. 2001; Langergraber et al., 2003; Ranieri, 2003), a constant head method (Knowles and Davies, 2009), and a falling head method (Caselles-Osorio et al., 2007). All these methods disturb the medium to a certain extent; although this disturbance is small compared with that caused during the extraction, storing and transportation of a gravel core to be tested in lab facilities.

The main objectives of this study were, first, to verify under lab conditions the reliability, accuracy and repeatability of the measurement of hydraulic conductivity in field SSF CWs by means of the falling head method and, second, to use this method to extensively map the degree of clogging in two mature, full-scale horizontal subsurface flow constructed wetlands, to assess the best operational, design and clogging management strategies that can be applied to SSF CW.

3.2 Methods

3.2.1 The falling head method

The falling head method (FHM) is based on Lefranc's test with falling heads (NAVFAC, 1986), which has been used to measure the saturated hydraulic conductivity of constructed wetlands' filter media (Caselles-Osorio et al., 2007). FHM consists of measuring the time a column of water inside a tube takes to drop a certain height. The tube (made of steel for example, and perforated at its bottom) is inserted within the granular media by hammering until the water level of the wetland is reached (Figure 3.1). The tube is filled with water in a pulse mode. A negative exponential curve of the water level inside the tube is obtained (Figure 3.2), whose slope is related to the hydraulic conductivity according to Lefranc's formula:

$$K = \frac{d^2 \ln(2L/d)}{8Lt} \ln\frac{h_1}{h_2} \tag{3.1}$$

where,

K is saturated hydraulic conductivity, in m/s; h_1 is the water height at time zero, in m; h_2 is the water height at time t, in m; d is the tube diameter, in m; L is the length of the submerged part of the tube (perforated zone), in m; t is time, in s.

Hydraulic conductivity values were obtained from an iterative procedure aimed at minimizing the difference between experimental and modelled data of h_2 (Caselles-Osorio and García, 2006):

$$\sum_{t=0}^{t=100}\left(h_2 - f(h_2)\right)^2 \rightarrow 0 \qquad (3.2)$$

where,

f (h_2) is the modelled data, in m

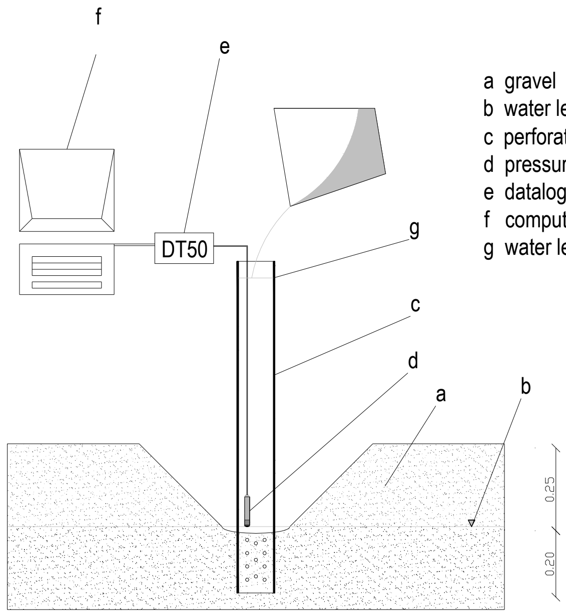

Figure 3.1 Schematic representation of the devices used for hydraulic conductivity measures. Dimensions in m.

In the iterative procedure hydraulic conductivity values (K) are given until the lowest difference between experimental and modelled h_2 values is found. Note that the mathematical model matches reasonably well the observed data in a broad range of hydraulic conductivities (Figure 3.2). Hydraulic conductivity values were estimated from observed data obtained during the first 100 s of the experiments. This is a pragmatic procedure that was chosen because in places where the hydraulic conductivity was very low, the decrease of the water inside the tube was slow and would have required more than one hour per test to measure.

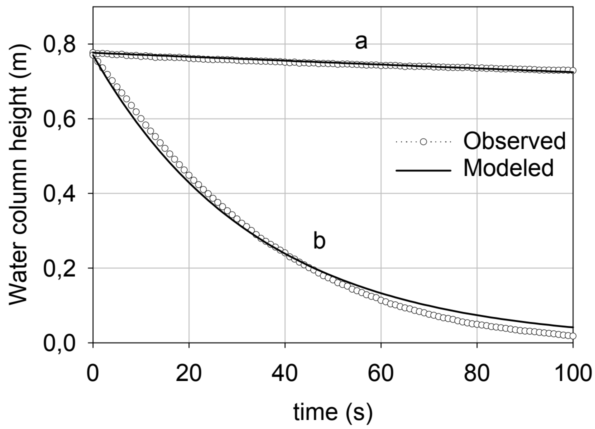

Figure 3.2 Decrease in water height column inside the perforated tube over time in two experiments. one of which corresponds to a site with a hydraulic conductivity of 0.5 m/d (a) and the other to a site with a hydraulic conductivity of 233 m/d (b). Modelled curves are shown for comparison.

In the present investigation, we performed a lab test to evaluate the reliability and repeatability of the FHM for *in situ* SF CWs clogging assessment. The test consisted of applying the FHM under lab-controlled conditions. To this end, two 300 L tanks (0.95 m long x 0.70 width x 0.45 m depth) were filled each with a different media (gravel and sand; a 30 cm layer in each case). To keep a constant water level (5 cm below the material surface), a drain tube in the tanks was connected to a spillway. This was set up at the base of each tank in each bed (tanks were filled with tap water for the tests). The characteristics of the sand and gravel tanks can be found in Table 3.1. The hydraulic conductivity of the filter media was determined by applying the FHM in 3 evenly spaced points along the central longitudinal section of each tank, near the inlet, in the middle and near the outlet (beside the drain tube). Each measuring point was separated from the others by about 25 cm. Some of these measurements were carried out by extracting the metal tube and inserting it again, whereas others were carried out repeatedly without extracting the tube. In these experiments, the hydraulic conductivity was measured three times in each test and the following were evaluated: 1) the effect of the filter media (by comparing measurements in the two tanks), 2) the effect of extracting the tube (by comparing measurements with or without extraction), and 3) the effect of the proximity of the drain tube (by comparing measurements at different locations).

Table 3.1 Important physical characteristics of the laboratory tanks used to evaluate the reliability and repeatability of the falling head test method, and of the wetlands in which field measurements were conducted in the Arnes and Gualba wastewater treatment plants.

		Design Flow rate (m^3/d)	Surface area (m^2)	Length to width ratio	Water depth (m)	Granular medium: D_{50} (mm)	Granular medium: Porosity (%)	Granular medium: Initial hydraulic conductivity (m/d)**
Sand tank		Inapplicable	0.74	1:0.7	0.25	0.9	31	32
Gravel tank		Inapplicable	0.74	1:0.7	0.25	7.1	39	265
Arnes		324	1700	1:3 (1:0.4)*	0.35-0.45	7.3	41	225
Gualba	New gravel	207	1404	1:2	0.35-0.45	7.5	45	181
	Original gravel					7.6	45	102

*Length to width ratio of each individual cell.
**Measured in this study according to the constant head method.

3.2.2 The constant head method

The accuracy of the FHM was assessed by comparison with a standardized method for laboratory measurements of hydraulic conductivity (a constant head method, CHM) (Rodgers and Mulqueen, 2006). The two filter media described in Section 3.2.1 were evaluated with the CHM.

A permeameter was built to measure hydraulic conductivity by the CHM. The aim of the permeameter was to allow hydraulic conductivity to be measured in a wide range of sample materials. Thus, it was designed to be suitable for measurements of both low and high conductivity materials (Figure 3.3).

To run the CHM test, a constant water flow rate was required. For this purpose, water was stored in an upper reservoir (750 L). From the reservoir, the water flowed through the permeameter cell (made from metacrylate, 70 cm long and 11.4 cm diameter), in which the granular material was placed. Attached to the permeameter cell were 3 piezometric tubes (10 cm apart from each other). Once water had passed through the medium, it ended up in a lower reservoir, in which a constant water level was maintained by a spillway. The height of the lower reservoir was changed to vary the overall test pressure gradient on demand. When water flowed through the medium, a pressure drop was obtained by friction with particles. This pressure drop was monitored with a GE Druck LPM 5480 Differential Pressure Transductor (DPT) that allows measurements from 0 to 15 mbar. A calibration procedure was used to correctly convert the information provided by the DPT (in Volts) to pressure units (in cm of water column). The DPT calibration was carried out by artificially generating pressure differences between the apparatus piezometers (by applying different flow rates) before the tests were run.

The calibration curve obtained is shown in Figure 3.4.

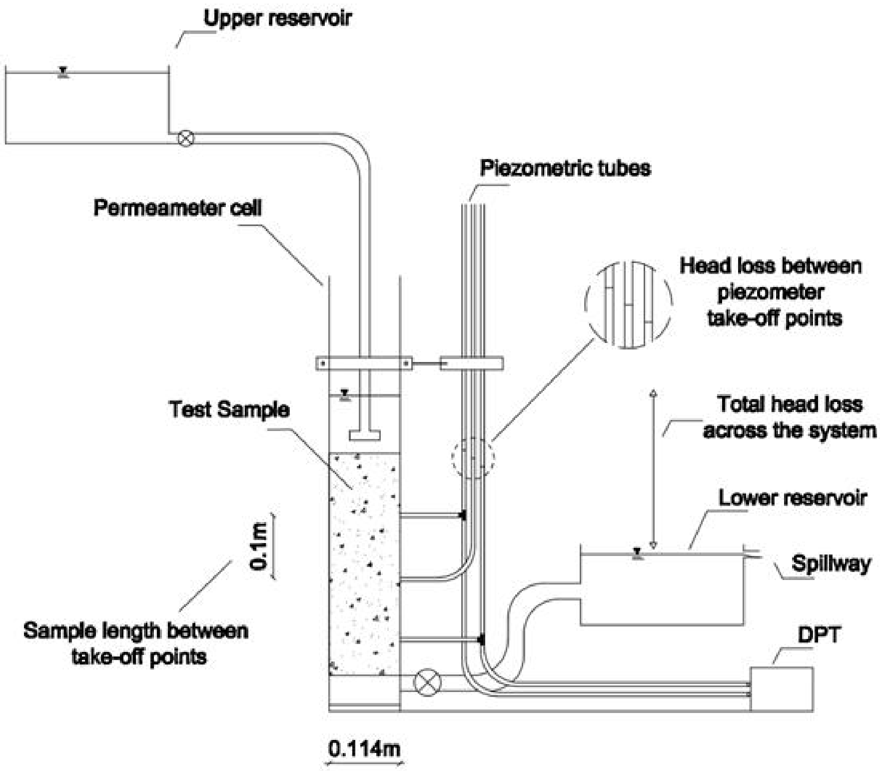

Figure 3.3 Diagram of the laboratory permeameter (constant head).

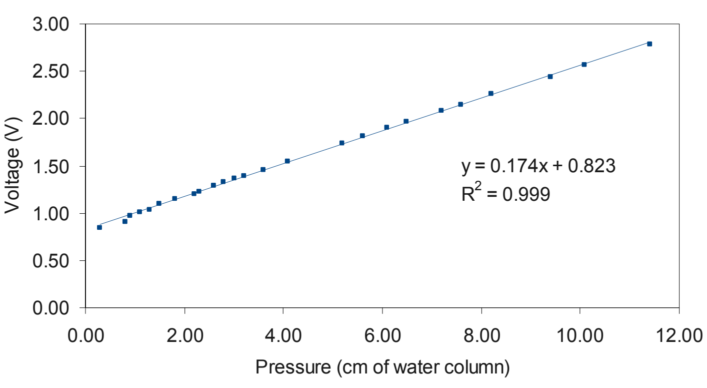

Figure 3.4 Calibration curve for the Digital Pressure Transducer.

If we know the vertical distance between the piezometric tubes, the cross-sectional area of the permeameter cell and the hydraulic pressure differences measured in the tests, the hydraulic conductivity of the sample can be determined with Darcy's Law as follows:

$$K = \frac{Q}{A}\frac{L}{\Delta H} \tag{3.3}$$

Where:

K is the hydraulic conductivity of the studied material, in m/s; Q is the flow rate through the sample, in m^3/s; L is the vertical distance between piezometric tubes, in m; A is the cross-sectional area of the permeameter cell, in m^2; ΔH is the head loss between piezometric tubes, in m.

Two conditions must be met in order to apply Darcy's Law. Firstly, the saturation index must be over 85% (Arnold, 1995). To this end, the permeameter cell was filled slowly with water from the bottom to the top prior to any test, which forced the entrapped air bubbles to leave the medium pores. Secondly, the flow during the test must be laminar. To fulfil this premise, Reynolds Number must be below 12, as this ensures laminar conditions (Jiménez Salas and De Justo Alpañes, 1975). A total of 14 and 9 tests were conducted on sand and gravel respectively in order to estimate the hydraulic conductivity (in each test there were slight variations and therefore changes in infiltration velocity).

3.2.3 Field Campaigns

Arnes wastewater plant

The treatment plant at Arnes (Tarragona, Spain) was put in operation in 1999 and treats the wastewater from a population of 512 people equivalent. The whole system consists of pre-treatment (screening), followed by primary treatment (Imhoff tank), two horizontal subsurface flow constructed wetlands working in parallel, the effluent of which is finally discharged into a surface flow constructed wetland. Historical records on water quality for selected pollutants are summarized in Table 3.2.

Each one of the two subsurface flow wetlands is divided by evenly spaced vertical walls into 8 cells that are 8.5 m wide and 25 m long. The surface of each cell is 212.5 m^2 (1700 m^2 for the whole wetland). Cells were initially planted with *Phragmites australis*, although some *Typha latifolia* specimens were also spotted during field campaigns. Water is discharged into the SSF

CW by a perforated pipe. At the end of the outlet pipe, an adjustable spillway keeps a constant water level for the whole wetland that can be modified on demand. The granular media is rounded chalky gravel, whose properties are shown in Table 3.1.

Table 3.2 Average values ± SD of selected pollutants in the Arnes and Gualba wastewater treatment plants.

WWTP	BOD_5 (mg/L)		TSS (mg/L)	
	Influent	Effluent	Influent	Effluent
Arnes[1]	469±248	108±109	286±327	100±120
Gualba[2]	309±260	20.7±18.5	407±245	25.9±23.0

[1]Average values based on annual records from 2000 to 2009. n=10

[2]Average values based on annual records from 2003 to 2009. n=7

Tests of this system were focused on one of the two subsurface flow wetlands and on four of its eight individual cells (A, B, C & D) (Figure 3.5). We studied a longitudinal transect along the central section of each of the considered cells, which consisted of 9 measurement points. In addition, we analysed two extra longitudinal transects of 3 measurement points each at both sides of the central transect. This procedure allowed us to interpret both the longitudinal and transverse clogging distribution of each cell. At each measurement point, the hydraulic conductivity tests were performed at a depth of 0.15 m from the water table level. Samples of gravel were also taken from several locations in each cell for hydraulic conductivity measurements at the lab facility.

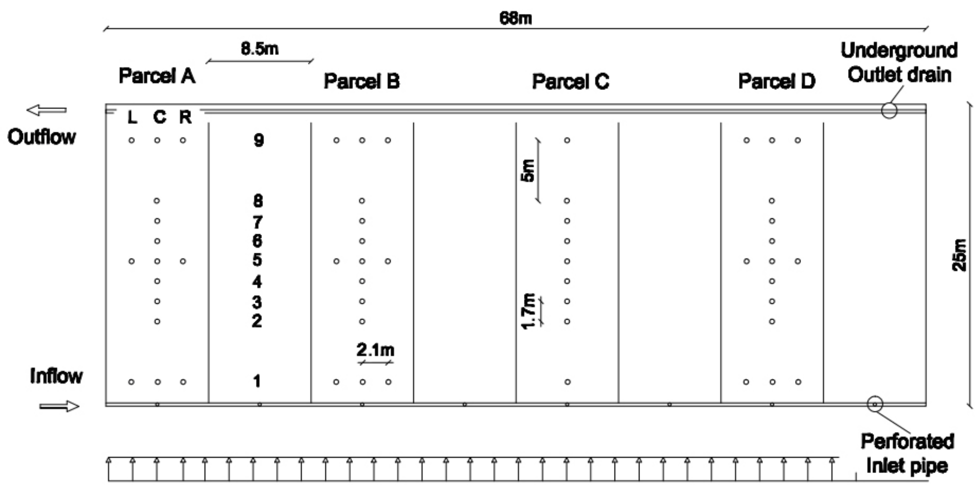

Figure 3.5 Plant view of the wetland at Arnes including all studied measurement points.

Gualba wastewater plant

The treatment plant at Gualba (Barcelona, Spain) was put into operation in 2002 and treats the wastewater of a population of 1072 people equivalent. The plant consists of pretreatment (screening) followed by two SSF CWs operated in series (there is no primary treatment). Each wetland is 27 m long and 52 m wide. Both the first and the last two metres of the length of each bed are composed of coarser gravel. The total wetland surface area is 2808 m^2 and it is all planted with *Phragmites australis*. Table 3.2 shows values of selected water quality parameters of the SSF CW since it was put in operation.

Filter medium from the first wetland in the series was partially replaced in January 2008, just 1 year and 3 months before the field measurements were carried out (Table 3.1). Specifically, the first 8 m from the inlet pipe were totally replaced with new gravel, whereas only the upper 20 cm layer in the rest of the bed was extracted and replaced. During measuring campaigns, it was noted that clogging caused water to overflow up to the mid-length of the cell.

Similarly to the field work at Arnes, the tests on this wetland consisted of *in situ* measurements of saturated hydraulic conductivity of the gravel together with filter medium samples taken for hydraulic conductivity measurements at the lab facility. Tests carried out on this system were focused on the first wetland of the series. Field hydraulic conductivity measurements were carried out with the FHM along 4 longitudinal transects of 7 sampling points each (Figure 3.6).

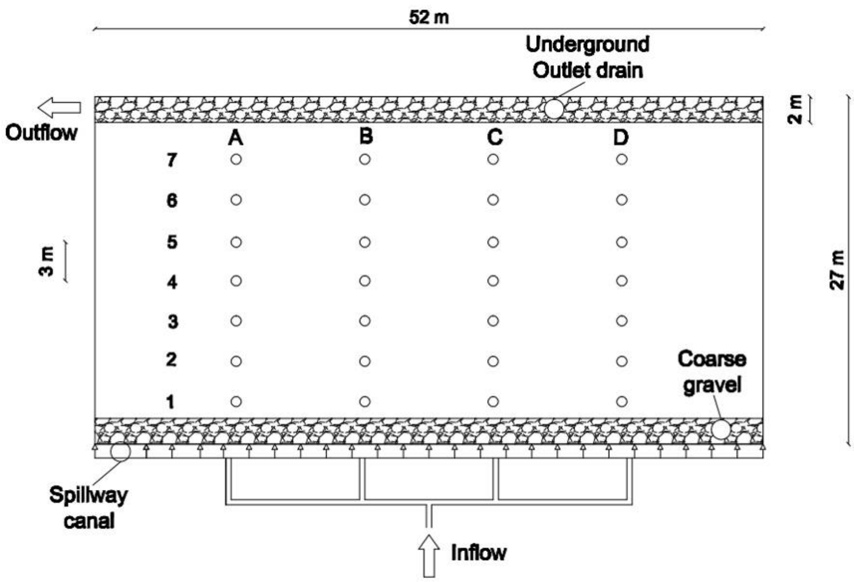

Figure 3.6 Plant view of the wetland at Gualba including all studied measurement points.

At each measurement point, the hydraulic conductivity tests were performed at a depth of 0.15 m from the water table level.

3.2.4 Filter media used in lab experiments

In order to assess the reliability and the accuracy of the FHM under lab-controlled conditions, different types of filter media were analysed following both the FHM and the CHM described in Sections 3.2.1 and 3.2.2, respectively. Specifically, washed sand and gravel filter media were selected for the purposes of the experiment (Table 3.1).

Furthermore, gravel samples obtained from the two field-scale SSF CW surveyed in this study were also tested under lab conditions following the CHM. The gravel samples extracted from the wetlands at Arnes and Gualba were washed to remove all existing organic matter. They were then analysed with the lab permeameter. In the case of Gualba, the two existing gravels (old gravel and the gravel introduced in 2008) were evaluated separately. Hydraulic conductivity values obtained with washed gravel from field-scale systems were considered to be representative of the system with no clogging (initial stages of the systems).

3.2.5 Statistical analyses

Differences among experimental conditions were assessed by applying the ANOVA test of variance (one way or three-way-ANOVA depending on the experimental design). All data subjected to the ANOVA test met the required conditions for the test to be applied. All statistical analyses were carried out using the software package SPSS 16.

3.3 Results and discussion

3.3.1 Reliability and repeatability of the falling head method

Lab tanks filled with sand and gravel were used to assess the reliability and repeatability of the FHM, and three experimental conditions that may affect hydraulic conductivity measurements were considered. The first experimental condition addressed the suitability of using the FHM for the analysis of clogged/un-clogged substrates in SSF CW. Current literature describes the *in situ* use of the FHM for hydraulic conductivity measurements, due to its action as an indicator of clogging in SSF CW (Pedescoll et al, 2009; Knowles and Davies, 2009). However, there is no scientific evidence that this method is reliable enough (evaluation of the method under controlled conditions) or to what extent (if applicable) variations in the procedure described in Pedescoll et al. (2009) may affect the conductivity values obtained for *in situ* surveys. Assessment of the reliability of the FHM on sand media was chosen as a model of a highly clogged filter medium (low hydraulic conductivity media), whereas measurements on gravel was chosen as a model of a non-clogged filter medium (high hydraulic conductivity). According to the results (Table 3.3), the FHM appears to be a suitable technique for the evaluation of clogging in SSF CW, since it is sensitive enough to record high significant differences in hydraulic conductivity measurements between the gravel and the sand media. Specifically, hydraulic conductivity values ranged from ca. 330 to ca. 365 m/day and from ca. 4.0 to 6.0 m/day for the gravel and sand media, respectively (Table 3.3). These hydraulic conductivity values are in the range of those described for different granulometric materials (Wilson et al., 2000), and also match those previously recorded in literature for severely and not severely clogged areas of SSF CW (Caselles-Osorio et al., 2007).

Table 3.3 Average values ± SD of hydraulic conductivity obtained with the falling head method applied to the two lab tanks with different filter media (sand and gravel) used for evaluation of reliability and repeatability of the method. Each average based on n=3. Data are in m/day.

		Gravel			Sand			
		Inlet	Middle	Outlet	Inlet	Middle	Outlet	
Measurement without tube extraction	ns[1]	349 ±41.1	365 ±55.7	334 ±17.9	4.3 ±0.6	4.0 ±1.0	6.0 ±1.7	ns[4]
			ns[2]			ns[2]		
				sd[3]				
Measurements with tube extraction		359 ±50.1	338 ±56.8	332 ±29.3	5.0 ±0.0	4.3 ±0.6	4.0 ±0.0	
			ns[2]			ns[2]		
				sd[3]				

ns: no significant differences according to three-way-ANOVA.
sd: significant differences according to three-way-ANOVA .
[1]Statistical analyses between tests carried out with or without tube extraction in the tanks with gravel as filter media.
[2]Statistical analyses between near inlet, middle and near outlet location of measurements.
[3]Statistical analyses between gravel and sand as filter media.
[4]Statistical analyses between tests carried out with or without tube extraction in the tanks with sand as filter media.

The two other experimental conditions that we studied were the effect of tube extraction (the tube is inserted and extracted in each measurement), and the location of the measurement spot (to assess the effect of the proximity of the drain tube). Accordingly, when the location factor is analysed, differences in hydraulic conductivity within the same media (gravel or sand) are not significant ($p<0.05$) (Table 3.3). Likewise, differences are not significant within the same medium that is analysed when tube extraction/no extraction is considered. In this case, the variation among treatments was around 2% and 7% for gravel and sand media, respectively. Therefore, the FHM is reliable and repeatable for a wide range of hydraulic conductivity conditions. However, it is necessary to point out that authors could be underestimating the variation of the method for highly clogged facilities. Accordingly, for highly clogged wetlands the tube insertion/extraction could produce higher variability than that here recorded (variation for sand media is as low as 7%) due to the non-cohesive nature of accumulated solids when compared to the cohesive nature of sand (used as a model for highly clogged wetlands).

3.3.2 Accuracy of the falling head method

In Section 3.3.1, we addressed the reliability and repeatability of the FHM. However, although it has been demonstrated that the method is reliable and repeatable for clogging assessment, there is no empirical evidence of its accuracy in current literature.

Moreover, there is a notable degree of uncertainty in the measurement of hydraulic conductivity of highly conductive materials (such as gravel media). Such coarse materials show a certain degree of variation in hydraulic conductivity depending on the flow rate, regardless of the method used. This is partly due to the fact that the limit between laminar and turbulent flow regimes is not always clear in highly conductive media (Wilson et al., 2000).

In spite of the uncertain degree of accuracy in measurements of the hydraulic conductivity of any highly conductive media, it is widely accepted that the CHM represents a standardized methodology for the assessment of such parameters in a wide range of materials under lab conditions (Rodgers and Mulqueen, 2006). Therefore, to assess the accuracy of the FHM, the authors chose to compare both methods. For the CHM, we decided to use the permeameter constructed for this purpose.

All 14 CHM tests conducted on the sand had Reynolds Numbers clearly lower than 12 (from 0.10 to 0.27). In the case of gravel, only 5 of 9 of the tests had Reynolds Numbers lower than 12 (values ranged from 9 to 22), and the values of these 5 tests were considered for calculations. Results on the hydraulic conductivity measured ranged from 30 to 45 m/d (average value of 35 m/d) and from 177 to 364 m/d (average value of 265 m/d) in the sand and gravel media, respectively (Figure 3.7). Therefore a good amount of variation was observed for the gravel, which in fact was connected to the infiltration velocity (which ranged from $1.3 \cdot 10^{-4}$ to $3.7 \cdot 10^{-4}$ m/s and from $2.8 \cdot 10^{-3}$ to $7.0 \cdot 10^{-3}$ m/s in the sand and gravel media, respectively). These results show that when different infiltration velocities are used, there is a notable amount of uncertainty in measurements of hydraulic conductivity, even under standardized methods such as the CHM.

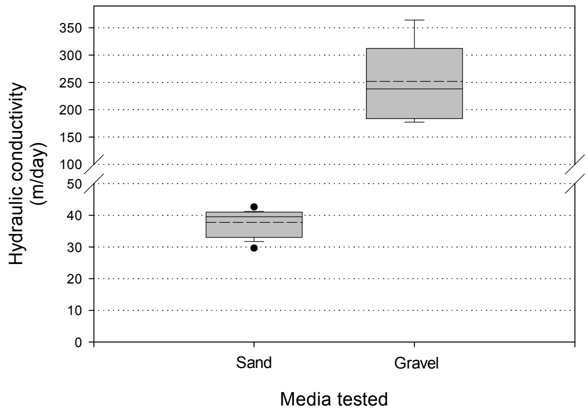

Figure 3.7 Box and whisker plots of hydraulic conductivity measured with the constant head method for the two filter media tested in the lab (n = 14 and 5 for sand and gravel, respectively).

A comparison of the average values of hydraulic conductivity obtained with both methods revealed differences, regardless of the media considered (one-way-ANOVA test of variance; $p<0.05$). Specifically, values obtained from the FHM method were ca. 80% lower than values obtained with the CHM for the sand media and ca. 20% higher when gravel media was considered (compare Table 3.3 and Figure 3.7). These deviations have to be considered acceptable in the context of hydraulic conductivity measurements (Bagarello et al., 2004, 2006), and we must conclude that the deviation in values between methods is acceptally small when compared to the range of hydraulic conductivity values measured throughout the system. These results indicate that exact values of hydraulic conductivity are difficult to be obtained because variations can exist depending on the measuring technique used, flow rate and system dimensions, sample collection technique and size, and physical and hydrological characteristics of the media (Reynolds et al., 2000; Wilson et al., 2000).

Even though the FHM might not be as accurate as the CHM under lab conditions, it is still a good and practical option to assess *in situ* measurements of hydraulic conductivity. The extraction of a gravel sample from field-scale SSF CWs and its evaluation in lab facilities may lead to even higher degrees of inaccuracy, due to the effect of sample collection and transportation. Although the accuracy of the information provided by the FHM is limited by the discussed uncertainties, it can still be used as relative information that will serve to compare different SSF CW and be of use to assess the best design, operation and clogging management strategies.

3.3.3 Clogging map of the SSF CW at Arnes and Gualba treatment plants to assess the best operational, design and clogging management strategies

Organic and TSS loadings are among the most important operational factors for clogging development in SSF CW (Langergraber et al., 2003). From the BOD_5 historical records, we calculated the organic loading at both SSF CW. The wetlands in Arnes have received, on average, 6 g BOD/m^2d in the last 7 years, whereas in Gualba the first wetland (there are two in series) has received around 27 g BOD/m^2d. In Gualba, the organic load is well above the recommended limit of 4 to 6 g BOD/m^2d. As a result, the wetlands are overloaded (Faulwetter et al., 2009; García et al., 2004, 2005). This is probably the cause of the massive hydraulic failure of the system which led to major replacement of the gravel in 2008.

Figures 3.8 and 3.9 show 2D representations of the hydraulic conductivity measured with the FHM in the wetlands at Arnes and Gualba. Note that results of cell C are not shown in Figure 3.8 due to lack of enough measurements. The values are quite similar in both systems and range from 1 to 40 m/d at Arnes and from 0 to 37 m/day at Gualba. These values are low if we consider that the hydraulic conductivity of the gravel media at its origin was in all cases higher than 100 m/d (Table 3.1). Therefore, both systems are developing clogging. This is of particular interest at Gualba, as most of the gravel in the wetland was replaced 1 year before our measurements were carried out. In contrast, the wetlands at the time of field measurements were over 10 years old. Thus, these results clearly indicate the major role of organic load in clogging development.

In general terms, the hydraulic conductivity in both systems showed a different pattern both horizontally and longitudinally (Figures 3.6 and 3.7). Specifically, at Arnes, the hydraulic conductivity increased the farther the measurement points were from the inlet pipe (Figure 3.8). Therefore, in this wetland, the highest conductivity values were measured near the outlet, and the lowest were measured near the inlet. These results are in accordance with previous studies (Caselles-Osorio et al., 2007; Knowles et al., 2010a). In contrast, at Gualba wetland, the highest values were randomly distributed (Figure 3.9), which suggests that shortcircuiting and preferential flow paths arise. The lowest hydraulic conductivity values were measured near the inlet and outlet of the bed (Figure 3.9). This behaviour was described by Pedescoll et al. (2009), but there is still a lack of explanations for this pattern.

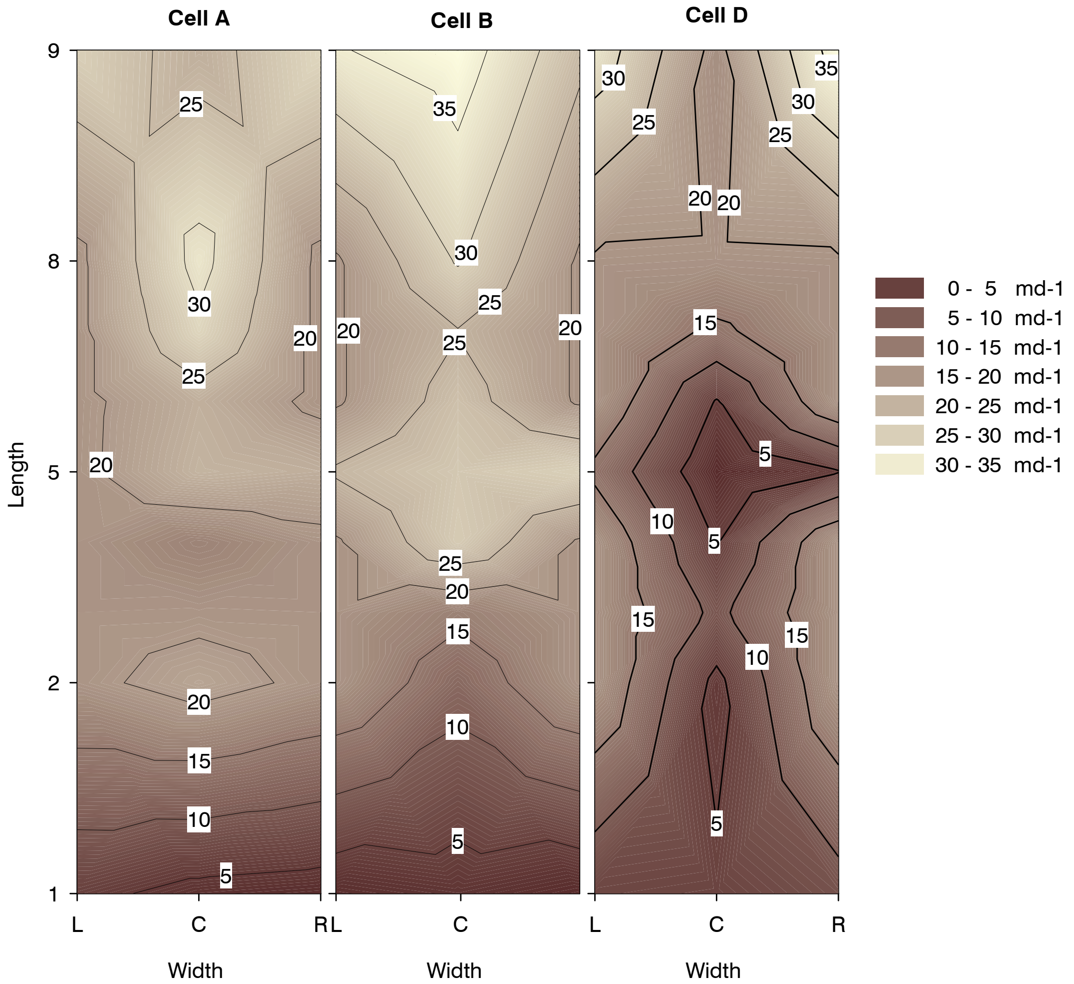

Figure 3.8 2D representation of the hydraulic conductivity at a depth of 15 cm in cells A, B and D in the wetland of the Arnes treatment plant. The water flow direction is from the bottom to the top of the graph.

The two wetlands differ in terms of organic loading, width-to-length ratio, the water inlet distributor, gravel type and time of operation. Therefore, the comparison of the results obtained from the hydraulic conductivity mapping and these operational and design differences may facilitate decisions on which design and/or operational scenarios are most suitable to counteract the clogging. Furthermore, extensive mapping of the wetlands will provide useful information for minimizing the costs when clogging must be managed. It can also serve as a guide for minimizing the costs when the only possible strategies to avoid the consequences of clogging are excavation and washing or replacement of the gravel.

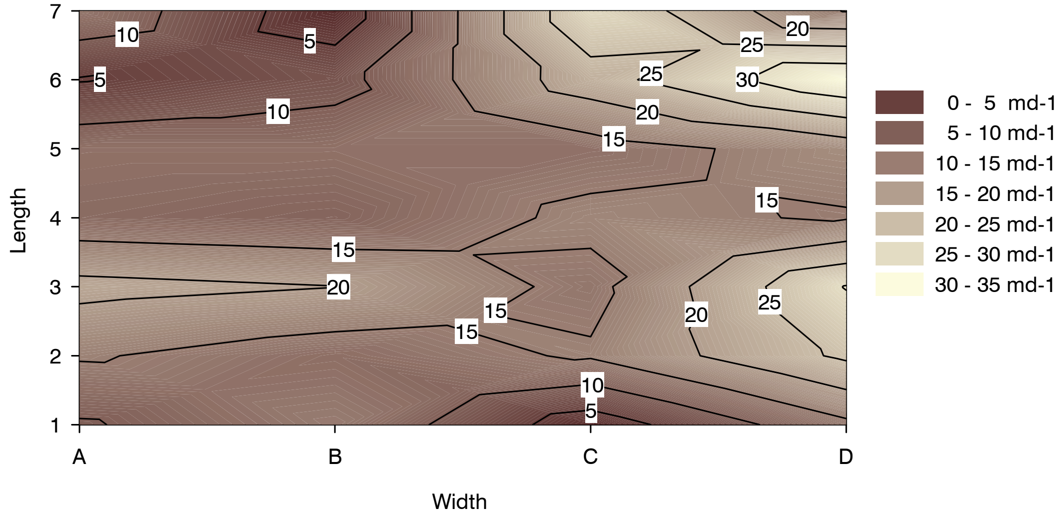

Figure 3.9 2D representation of the hydraulic conductivity at a depth of 15 cm in the wetland of the Gualba treatment plant. The water flow direction is from the bottom to the top of the graph.

The effect of the feed system and associated preferential water flows on clogging distribution

Both wetlands had fairly uniform horizontal conductivity values near the inlet pipe. At Gualba, the inlet system is a concrete channel, in which the water overflows through a weir. This distribution system is better than most alternatives at achieving uniform distribution, and is easier to maintain (Griffin et al., 2008). The inlet system at Arnes consists of a perforated PVC pipe, with one hole per cell. The water distribution is relatively uniform in the Arnes system, although less so than in the Gualba system. However, the inlet pipe settlement on the ground is not uniform, so there is a loading tendency to the right side of the wetland. This loading promotes higher flow in cells located near the right of the wetland (cells C and D), which in turn leads to lower hydraulic conductivities to the middle of the cells. In cells C and D, the excess of inflow water led to the formation of a layer of sludge and plant leaves, which prevented water from infiltrating. In areas where water had eventually infiltrated through to the subsurface, clogging had rapidly developed. When the reduction of pore volume was enough onto the surface, the water flowed over the top of the bed until it is able to fully infiltrate into the subsurface. This behaviour was also described by Knowles et al. (2010a) in subsurface flow wetlands for tertiary treatment.

The middle section of cell D of the wetland in Arnes (Figure 3.8) was more clogged than the inlet zone. This unexpected situation may have been induced by the fact that this individual bed receives more water flow than the rest of the beds, due to improper settlement of the inlet pipe. In this situation, the gravel cannot drain a high volume of water. Therefore, surplus water is forced to overflow and can only infiltrate half way through the length of the gravel bed. In the area where water infiltrates, all the particulate matter is deposited and may contribute to the reduction of hydraulic conductivity shown in this area. Therefore, unless it is properly maintained, the use of an inlet pipe may lead to differential settlement on the ground and preferential water flows.

The impact of the aspect ratio on clogging distribution

The horizontal clogging distribution suggested by hydraulic conductivity measurements in each of the multiple cells at Arnes is more uniform than in the wetland at Gualba. This is due to the smaller shape of Arnes' individual cells (Table 3.1). In Gualba's cells, the flow in the inlet zone seems to be quite uniform, whereas in the outlet zone the presence of clogging is more noticeable in transects A and B. This suggests that water preferentially flows diagonally through the gravel bed from inlet to outlet (Figure 3.9).

In the wetland at Arnes, the uniform horizontal distribution of the water flow is maintained from the inlet to the outlet of the cells, especially in cells A and B, which are not affected by the improper settlement of the influent pipe (Figure 3.8). This tendency seems to be encouraged by the narrow aspect ratio of each cell. Unlike the Gualba's system, the low width-to-length cell ratio in Arnes (1:3), prevents water from concentrating in certain paths and forces it to occupy the whole transverse area. The width-to-length ratio is not a decisive factor in the removal efficiency of horizontal SSF CWs, at least during the first few years of operation (García et al., 2005). However, it may promote preferential water flows in SSF CW, which, in turn, might have an effect on clogging development.

Therefore, both the aspect ratio and the inlet/outlet systems might play a significant role in the good hydraulic functioning of SSF CW and, thus, in the clogging distribution in such systems.

3.4 Conclusions

The falling head method (FHM) for hydraulic conductivity measurements in SSF CW is a reliable and repeatable technique. It is also suitable for detecting the hydraulic conductivity variations that might be encountered in SSF CW (from 4 to 365 m/d).

Even though the FHM is not as accurate as normalized methods such as the CHM under lab conditions (especially for low conductive media), it is still a good and easy option to assess *in situ* measurements of hydraulic conductivity for clogging assessment. The method is more accurate for highly conductive media (i.e. 200-300 m/d) than for lowly conductive (i.e. less than 50 m/d).

Both studied wetlands present low hydraulic conductivity values which range from 0 to 40 m/day. Therefore, the two systems were moderately clogged at the moment that the study was conducted. Variations in the design of the two systems such as the inlet distributor and the aspect ratio results in differences in clogging distribution patterns.

This study demonstrates that the use of the FHM for mapping the hydraulic conductivity of field scale SSF CW allows key design factors to be identified that influence the degree of clogging. Furthermore, extensive mapping of a SSF CW following the FHM turns out to be a suitable tool for guiding *in situ* clogging corrective measures, such as media chemical cleaning, excavation and washing or replacement.

4

Linking indirect measures to clogging phenomena in full scale subsurface flow constructed wetlands[2]

This chapter presents an application of the method for evaluating clogging of subsurface flow constructed wetlands based on saturated hydraulic conductivity measurements (presented in Chapter 3). The method was applied along two transects that spanned the length of two full-scale wetlands, where solids and belowground plant biomass were quantified. X-ray diffraction analyses were carried out to evaluate the mineral composition of accumulated sludge and gravel. Patterns for hydraulic conductivity were the same in both wetlands: very low values from the inlet zone to the middle (< 20 m/d), clearly higher from the middle to 4/5 of the length (600 to 800 m/d), and lower very near the outlet (40 to 70 m/d). These results indicate that the first half of the length of both wetlands is highly clogged. Total solids (TS) were generally higher near the inlet than the outlet (TS_{inlet}= 3-15 kg/m^2; TS_{outlet}= 1-9 kg/m^2). Belowground plant biomass did not show a clear pattern. The inorganic fraction of the solids represented more than 75% of TS in most of the samples and its mineral composition coincided with that of the granular medium (mostly calcite and quartz). The proposed method is straightforward to use, does not require costly devices and allows evaluate the degree of clogging.

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[2] This chapter is based on the article:

Pedescoll, A., Uggetti, E., Llorens, E., Granés, F., García, D., García, J., 2009. Practical method based on saturated hydraulic conductivity used to assess clogging in subsurface flow constructed wetlands. Ecological Engineering 35(8), 1216-1224.

4.1 Introduction

The use of subsurface flow constructed wetlands (SSF CWs) for wastewater treatment in Spain is relatively new. In recent years an increasing number of full-scale constructed wetlands (CWs) have been put into operation (García, 2004; Puigagut et al., 2007). This rise in the implementation of such treatment systems is because CWs have several advantages over conventional wastewater treatment systems, particularly in small villages (≤2000 population equivalent (PE)): staff do not require specific training, the operation and maintenance costs are lower, and they integrate well into the surrounding landscape (Vymazal, 2005; Rousseau et al., 2008).

The main operational problem of SSF CWs is the progressive clogging of the granular medium. The development of clogging can be detected by the appearance of water on the surface of the granular medium near the inlet zone. When clogging becomes severe, water is seen overflowing onto the medium surface. Clogging results in various counterproductive situations: it decreases the hydraulic conductivity and porosity of the granular media, it causes preferential water flows along the wetland, and it results in dead zones and/or short circuits. These processes diminish hydraulic performance, which consequently can affect the contaminant removal efficiency and life span of the system (Platzer and Mauch, 1997; Knowles et al., 2008).

The causes of clogging were reviewed by Blazejewski and Murat-Blazejewska (1997). They concluded that the main causes of clogging were the accumulation of wastewater solids and vegetal debris, the growth of the biofilm onto the granular medium, rhizomes, and roots, and finally the depostion of chemical precipitates.

Functionally, in spite of the significance of clogging processes in the performance of CWs, companies in charge of the operation and maintenance (O & M) of wetlands have not yet established standard protocols to allow them to assess the degree of clogging in a simple way. General practice is that the granular medium is left clogging until a decrease in contaminant removal efficiency is observed. At this stage one of three routes can be followed: (1) the clogged medium is washed and returned to the wetlands, (2) it is partially or completely replaced with a new one, or (3) it is exposed to an oxidising agent like H_2O_2 (Behrends et al., 2006; Nivala and Rousseau, 2008; Kadlec and Wallace, 2009). Any of these actions come with a subsequent high cost. As example, in early 2008 the partial replacement of the granular medium from one of the two wetlands (1,600 m^2 each) in a wastewater treatment plant located in Gualba (Barcelona, Catalonia, Spain, 1,000 PE) cost approximately €130,000 (Francesc Llenas, *pers. comm.*), whereas the annual O & M budget for such facilities in the zone is around €30,000 (Robusté, 2004). Another example is the inlet zone maintenance of two wetlands in Minnesota, which had a cost from 10 to 19% as percentage of the initial construction cost

(Kadlec and Wallace, 2009). This kind of situation could be eased by the development of a simple method for evaluating the degree of clogging that would allow sufficient time to plan for progressively replacing or washing the granular medium if required. Furthermore, it would help determine the amount of granular medium that would have to be replaced or cleaned. This paper presents a simple method to evaluate the degree of clogging based on saturated hydraulic conductivity measurements.

The method was applied to two full-scale horizontal SSF CWs located in the wastewater treatment plants of two small villages in the province of Lleida (Catalonia, Spain). The accumulation of solids in these two wetlands was evaluated in 2006 in a survey of six wastewater treatment plants in the region of Catalonia (Caselles-Osorio et al., 2007). In the present study we also quantified the accumulated solids and the belowground plant biomass to further explore their relation with hydraulic conductivity measurements. An assessment of the mineral composition of accumulated solids and granular medium was also carried out.

4.2 Methods

4.2.1 Description of horizontal SSF CWs

The wetlands where the study was carried out were located in the wastewater treatment plants (WWTPs) of Verdú and Corbins (Lleida, Catalonia, Spain, both with a PE of approximately 2,000). Verdú WWTP comprises three parallel septic tanks, four secondary horizontal SSF CWs, two parallel aerobic ponds, and two parallel polishing horizontal SSF CWs. Corbins WWTP comprises one Imhoff tank, two parallel secondary horizontal SSF CWs, three ponds in series (one facultative and the other two aerobic), one polishing horizontal SSF CW, and three parallel sand filters. All wetlands are planted with common reed (*Phragmites australis*). These two facilities were put into operation in 2002, and are currently managed by Aigues de Catalunya SA. Additional details of these WWTPs can be found in Caselles-Osorio et al. (2007).

Table 4.1 Main characteristics of the two horizontal subsurface flow constructed wetlands (HSSF CWs) evaluated in this study.

HSSF CW	Flow rate* (m^3/d)	Surface area (m^2)	Length to width ratio	Water depth (m)	Granular medium		
					D_{60} (mm)	C_u	Porosity (%)
Verdú	177	977	1:1.1	0.5	9.0	1.8	40
Corbins	218	1225	1:1	0.5	9.2	1.8	38

* Flow treated by the entire wastewater treatment plant. Note that this is an estimated flow rate because there is no flowmeter in the plants.

A secondary horizontal SSF CW from each WWTP was chosen for present study. Table 4.1 shows the main characteristics of these two wetlands. Table 4.2 shows historical data of TSS and BOD_5 in the influent and effluent of the entire WWTPs (data were provided by the company in charge of the operation and maintenance of the WWTPs; analyses carried out according the methods described in APHA-AWWA-WPCF (2001)).

Table 4.2 Average and standard deviation (in brackets) of influent (Inf) and effluent (Eff) TSS and BOD_5 concentrations in the two studied systems. Calculations based on 9 to 12 data per each year. Yearly average surface organic load and cross sectional load applied to the two wetlands chosen for the present study are also shown.

System	Year	TSS (mg/L)		BOD_5 (mg/L)		Surface organic load* (g/m^2.day)	Cross sectional load* (g/m^2.day)
		Inf	Eff	Inf	Eff		
Verdú	2003	200 (140)	10 (8)	120 (77)	5 (2)	3.8	220
	2004	150 (60)	19 (11)	88 (29)	9 (5)	2.8	170
	2005	250 (340)	22 (20)	230 (190)	16 (12)	7	430
	2006	110 (80)	15 (10)	110 (40)	12 (6)	3.4	210
	2007	330 (270)	24 (12)	250 (120)	15 (7)	7.7	470
	2008	270 (170)	20 (20)	230 (120)	11 (6)	7	430
Corbins	2003	190 (42)	32 (9)	160 (57)	19 (13)	9.8	700
	2004	280 (240)	23 (12)	160 (110)	13 (8)	9.8	700
	2005	150 (150)	17 (11)	280 (80)	23 (13)	17	1200
	2006	190 (150)	16 (10)	390 (140)	15 (11)	25	1700
	2007	1300 (2200)	19 (9)	740 (780)	16 (9)	46	3200
	2008	480 (440)	40 (38)	450 (180)	11 (3)	28	2000

* Calculated assuming a 30% of removal in the by primary treatment.

4.2.2 Assessment of degree of clogging

Saturated ydraulic conductivity measurements, sampling, and analyses were carried out in March and April 2007. A full day of measurements and sampling was carried out once each month. Hydraulic conductivity measurements were taken in two transects along the length of each wetland (Figure 4.1). This took place at regular intervals of approximately 3.3 m in the Verdú wetland and 3.5 m in the Corbins wetland. Part of the granular medium at the inlet at Corbins had recently been replaced with a new medium that was not representative of the rest of the wetland so the first measurements were not taken precisely at the inlet, but slightly further from it.

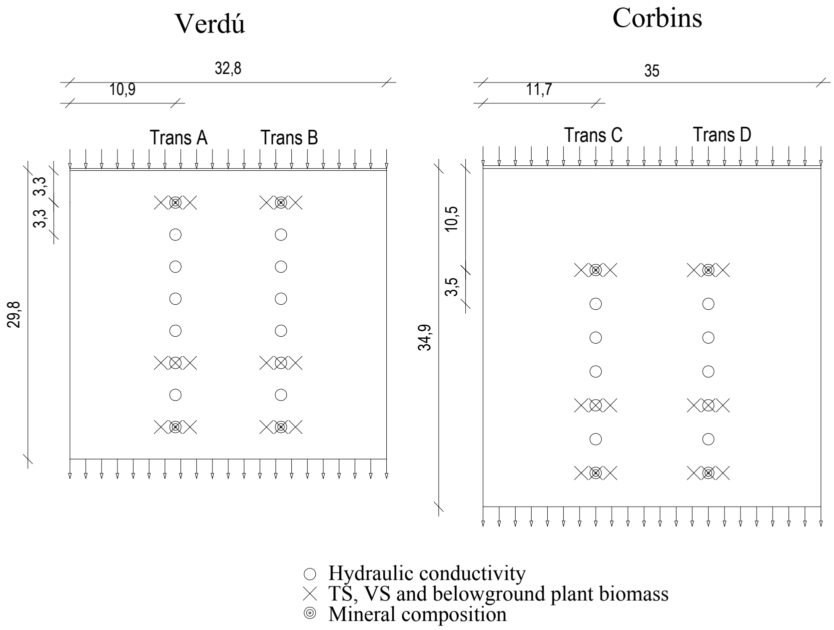

Figure 4.1 Location of transects (Trans) and sampling sites for hydraulic conductivity. Sampling sites for solids (TS and VS), belowground biomass and analyses of mineral composition were removed. All dimensions in m.

Saturated hydraulic conductivity was measured using the falling-head test method (NAVFAC, 1986) according to the procedure described in Chapter 3. A small hole was dug in the granular medium using a spade until the water level was reached (which was usually located approximately 10-25 cm below the surface of the granular medium). A steel tube perforated on one side with an internal diameter of 10.5 cm and a length of 100 cm was then inserted to a depth of 20 cm using a mallet. The tube was completely filled with water (in a pulse using a bucket) and the decrease of the water column height inside the tube was recorded using a TNS-119 pressure probe (0-1 m water column meter) connected to a computer via a Datataker DT50. The pressure probe was placed inside the tube on the granular medium surface. Note that in this experiment pressure variations were proportional to the water column height inside the tube. Negative exponential curves were produced, which represent a decrease in the water height inside the tube over time; the curve is linked to hydraulic conductivity.

A mathematical model described in NAVFAC (1986), which is obtained by combining the mass conservation principle and Darcy's law, was used to estimate hydraulic conductivity:

$$K = \frac{d^2 \ln(2L/d)}{8Lt} \ln \frac{h_1}{h_2} \tag{4.1}$$

Where:

K is the hydraulic conductivity of the studied material, in m/s; d is the diameter of the tube, in m;

L is the submerged length of the tube, in m; h_1 is the height of the water table level inside the tube at time zero, in m; h_2 is the height of the water table level inside the tube at time t, in m; and t is time, in s.

The quadratic difference between the theoretical curve and that obtained in the field is minimized to estimate the value of the hydraulic conductivity.

4.2.3 Quantification of solids and belowground plant biomass

The granular medium (mixed with belowground plant biomass) was sampled simultaneously with hydraulic conductivity measurements in three different places along the transects for the quantification of both accumulated total solids (TS) and volatile solids (VS) (Figure 4.1). These sampling places were selected based on the hydraulic conductivity values so that samples were representative of the sites with both low and high conductivity. Three replicas of granular medium were taken one metre apart at each place using a steel tube with a diameter of 20 cm and a length of 40 cm. In order to obtain each replica the steel tube was almost entirely inserted into the granular medium (in three different points, one per replica) using a mallet. The granular medium of the unsaturated zone inside the tube was carefully removed with a shovel, and a wetted granular medium volume of around 3 L was taken out from the tube, stored in plastic bags and maintained at 4º C until it was processed in the laboratory the next day. While the steel tube was being inserted and the granular medium was taken out, rhizomes and roots of the plants were collected and stored in refrigerated plastic bags for belowground plant biomass quantification. Note that it was checked that almost all plant belowground biomass was in the unsaturated zone and in the first 10 cm of the wetted zone. Below this depth plant biomass was very scarce. Granular medium samples were obtained from the first 10-20 cm of the wetted granular medium (Figure 4.2). The amount of solids accumulated in the granular medium may vary with depth, meaning that our measurements in one section of the water column only gave an approximation of the amount of accumulated solids. In addition, when the tube is inserted a certain degree of compaction can occur, and therefore the replicas are not intact. However, for the practical purposes of this study it was considered to be a representative procedure.

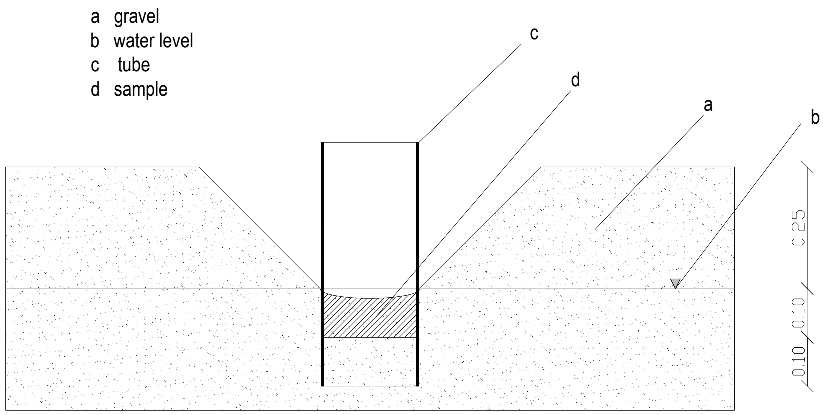

Figure 4.2 Schematic representation of the devices used to removed accumulated solids and below-ground plant biomass samples. Dimensions in m.

The process of assessing the granular medium samples (replicas) started by removing recognisable belowground plant biomass not detected when the samples were taken out in the field. A total of 0.5 L of granular medium was then collected and mixed with 0.5 L of distilled water; the samples were then shaken by hand to wash them (Caselles-Osorio et al., 2007). The resulting water (containing accumulated solids) was filtered through a 1 mm metal mesh to further remove any macroscopic plant material or detritus and analysed for TS and VS using conventional methods (APHA-AWWA-WPCF, 2001). Note that with this methodology accumulated solids can not be separated from fine materials of the granular medium. The rest of the granular medium (~ 2.5 L) was washed with tap water and subsequently filtered through the metal mesh to collect the belowground plant biomass. All vegetal biomass separated in the laboratory was then mixed with the biomass collected in the field. The biomass was dried for 24 hours at 105º C and then weighed.

4.2.4 Mineral composition of the granular medium and accumulated solids

Mineral composition was analysed from two different places (near the inlet and the outlet) from composite samples obtained from the samples (replicas) collected for the quantification of solids (Figure 4.1). Mineral composition was evaluated from accumulated solids and the washed granular medium that was taken from the previous sample processing procedure. Furthermore, samples of the original granular medium (without accumulated solids) were taken from the unsaturated zone for comparison. Both granular medium and solids samples were processed according to the EN-1477-1 (AENOR, 1998b) and EN 932-2 (AENOR, 1999) standards and were submitted for X-ray diffraction. A Siemens D-500 diffract-meter with copper pipe was used. The diffractometry was carried out from 4º to 70º with steps of 0.05º every 3 seconds.

4.3 Results

Table 4.2 shows average influent and effluent TSS and BOD_5 concentrations in the two WWTPs. Removal rates range between 86-95% and 83-99% for TSS and 89-96% and 88-98% for BOD_5 in Verdú and Corbins respectively. In all cases the average BOD_5 concentration in the effluent is under the 25 mg/L limit of the European Union Directive 91/271/ECC (Council of the European Communities (1991)). Surface organic loads applied to the two studied wetlands range from 2.8 to 7.7 g BOD/m^2d in Verdú and from 9.8 to 46 g BOD/m^2d in Corbins. Therefore, the wetlands at Corbins operate with a clearly higher organic load than in Verdú, which is well beyond the limit of around 6 g BOD/m^2d reccommended for horizontal SSF CWs (García et al., 2004).

4.3.1 Degree of Clogging

To assess the degree of clogging, measurements of saturated hydraulic conductivity were taken in two transects along the length of both wetlands (Figure 4.3). The results from these measurements revealed a general pattern common to both wetlands. From the inlet zone to approximately the middle zone, hydraulic conductivity values were very low, with most values lower than 20 m/d. The highest values were recorded between the middle zone and a point along the wetland that was almost 4/5 of its length; in this segment the values were in the range of those reported in other researchs for clean gravel (Kadlec and Watson, 1993; Suliman et al., 2006a) and were generally 1 to 3 orders of magnitude greater than those from the inlet zone. Near the outlet zone hydraulic conductivity decreased unexpectedly, with the majority of values ranging from 40 to 70 m/d. This decrease occurred systematically and was surprising because a more logical trend would have been an increase in the hydraulic conductivity from the inlet to the outlet.

The clogging patterns in each of the two transects at Verdú were very similar. Clogging was very apparent along the transect between the inlet to a point reaching slightly less than halfway along the bed. Small ponds were observed during the field work in this zone. At Corbins, the degree of clogging showed a pattern which was common to both transects, with the exception of the values recorded in a relative length from 0.6 to 0.8. In transect C the a general lower hydraulic conductivity was recorded (Figure 4.3) and seemed to be related to the stagnant water observed along the entire transect during field work. In fact, at Corbins there were several places in the wetland where flooding had occurred. The results for hydraulic conductivity at Corbins clearly indicate that the wetland was clogged to a relative length of almost 0.6; the rest of the bed had areas with low conductivity (as shown in transect C).

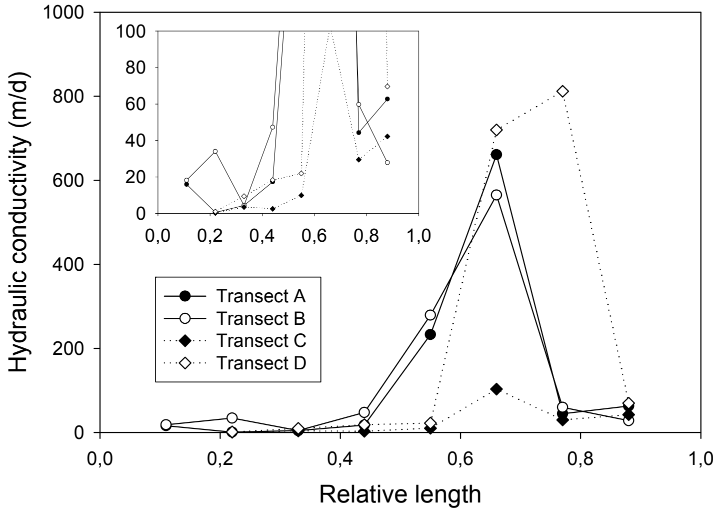

Figure 4.3 Hydraulic conductivity values in transects along the length of the Verdú wetlands (A and B) and Corbins (C and D). In the top left the same graph is shown with a hydraulic conductivity range (Y-axis) from 0 to 100 m/d, to remark the difference between inlet and outlet zone of the wetlands. The length of the wetlands is represented in a relative form to make the data comparable.

4.3.2 Solids and belowground plant biomass

Solids and belowground biomass were particularly variable, even within the same sampling places, which is illustrated by the width of the ranges shown in Table 4.3. It is important to note that although SSF CWs seem uniform from a distance, there is a strong heterogeneity at the local scale.

In the Verdú wetland the amount of accumulated total solids (TS) decreased from inlet to outlet (Table 4.3); furthermore, the places with the lowest TS content were not the places with the highest hydraulic conductivity (compare data at ¾ of the length with data at the outlet zone). In the Corbins wetland this decreasing trend in TS content was not observed, although the values observed near the inlet zone were higher than in the other places. In fact as Figure 4.4 illustrates, the TS content near the outlet in transect C distorts a very obvious decreasing trend in TS content observed in transect D. In transect C, the lowest hydraulic conductivity was recorded at a point that was ¾ of the length of the transect.

When the TS values of the two wetlands were compared it was observed that overall the Corbins wetland system had a higher solids content than the Verdú system, which is a result of the higher organic loading rate (OLR) applied in the former.

Table 4.3 Ranges of accumulated solids and belowground plant biomass in the Verdú and Corbins wetlands. Average values in brackets (averages calculated from 6 values). Note that the percentages of VS refer to the averages.

	Verdú			Corbins		
	Inlet zones	¾ of the length	Outlet zone	Inlet zone	¾ of the length	Outlet zone
TS (kg/m^2)	3.15 – 14.98 (8.06)	2.40 – 4.15 (2.99)	1.17 – 2.52 (1.83)	3.04 – 10.07 (6.05)	1.73 – 4.58 (3.31)	1.95 – 9.04 (4.89)
VS (kg/m^2)	0.60 – 1.78 (1.73)	0.26 – 1.69 (1.00)	0.21 – 0.41 (0.26)	0.67 – 2.83 (1.57)	0.27 – 0.67 (0.50)	0.23 – 1.88 (0.83)
VS (%)	21	33	14	26	15	17
Belowground biomass (g/m^2)	67 – 643 (355)	131 – 838 (484)	59 – 932 (495)	202 – 726 (464)	20 – 937 (478)	33 – 109 (71)

Interestingly, the percentage of volatile solids of all samples was low and ranged from 15 to 33%, which makes evident the high mineral content of the accumulated solids. These percentages are low when one considers that these solids are indeed sludge from a wastewater treatment plant. However, they are similar to those reported by Caselles-Osorio et al. (2007) for 6 full-scale wetland systems (including the two wetlands evaluated in the present study).

There was no clear trend in either of the wetlands in terms of the amount of plant belowground biomass (Table 4.3). At Corbins the amount of biomass was lowest in the outlet zone.

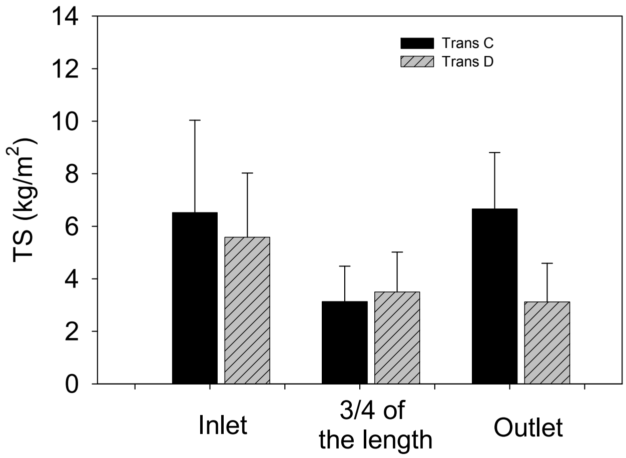

Figure 4.4 Average values of TS in transect C and D in each site in the Corbins wetland. Standard deviations are shown. Averages were calculated from 3 data sets.

4.3.3 Mineral typesetting of the granular medium and accumulated solids

X-ray diffraction analyses were carried out in order to compare the mineral composition of the original granular medium, the granular medium at wetted depths, and the accumulated solids associated to these media. Data from the analyses revealed that the mineral fraction of the original granular medium in both CWs was mainly composed of Calcite ($CaCO_3$) and Quartz (SiO_2), with each mineral accounting for approximately 30% of the total mineral composition (Figure 4.5 and Table 4.4). Other substances that were present in lower proportions include clays and silicic minerals. The mineral composition of the granular medium at wetted depths was almost identical in both wetlands for the different sampling locations. Furthermore, the mineral composition of the accumulated solids was similar to that observed for the granular media, with only a few differences in the relative proportions. This result indicates that accumulated solids in the two wetlands mainly correspond to fine materials with a similar composition of that of the granular medium.

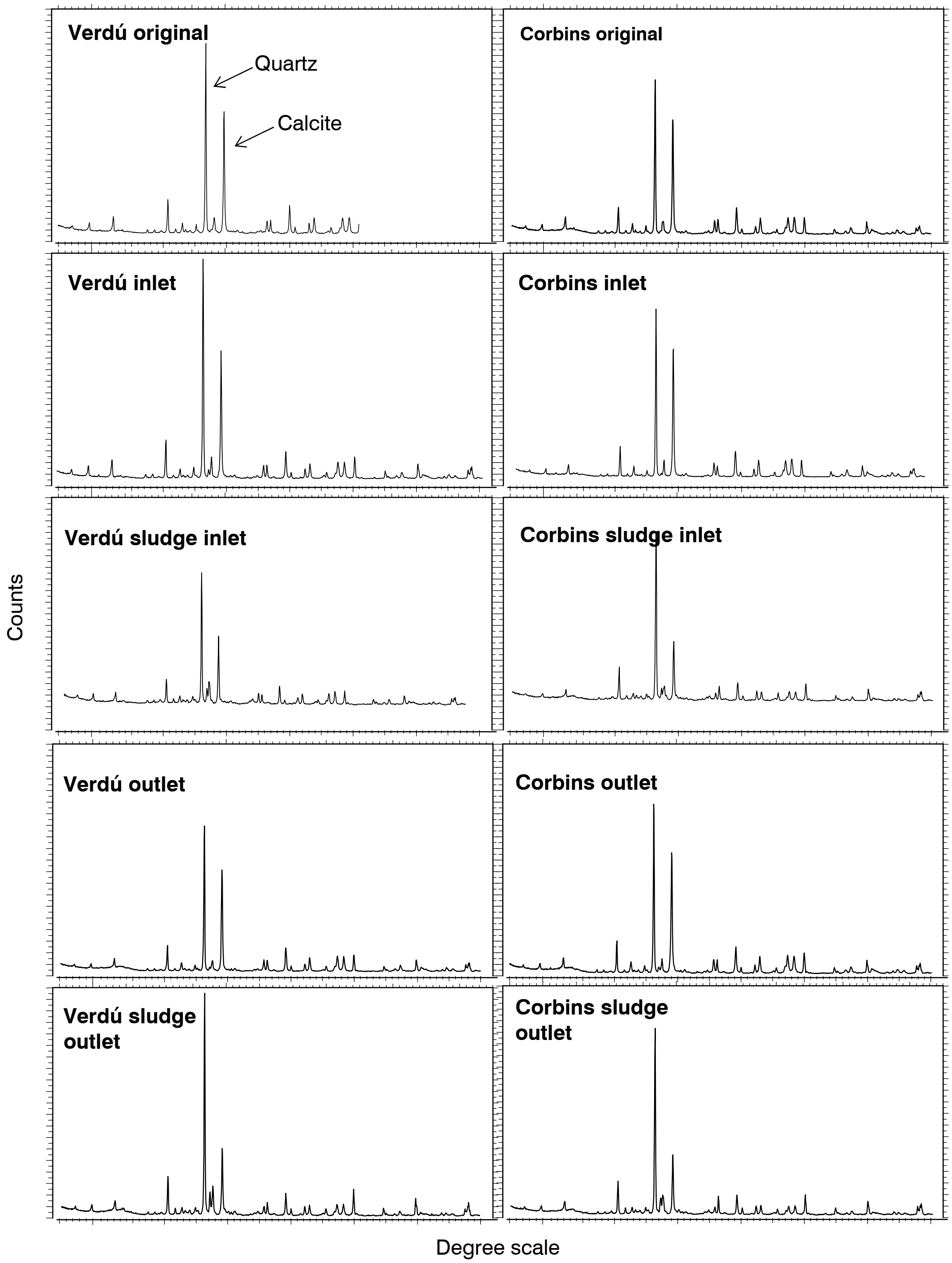

Figure 4.5 Diffractograms of original granular medium. granular medium at the wetted depth. and accumulated solids in the Verdú and Corbins wetlands.

4.4 Discussion

This paper presents the results of an evaluation of a simple method that can be rapidly applied for assessing clogging in horizontal SSF CWs. Two transects were used and saturated hydraulic conductivity was measured at several places along the length of the wetlands with a falling head-test method. The minimum number of transects recommended for practical purposes is two; however, if there is substantial hydraulic conductivity variation in width, more transects would be required to capture this variability. In practice, strong changes in hydraulic conductivity can occur near the inlet zone due to non-uniform wastewater distribution along the width (Cooper et al., 2005). The present study has shown that great differences in the width also can be found in other locations due to preferential flows (at ¾ of the length of the wetland at Corbins, see Figure 4.3).

In this study, hydraulic conductivity was measured at intervals of approximately 3 m along the length of the 30 m wetlands. Although it is recommended that for practicality the wetland be divided into at least 5 sections, the intervals chosen will vary depending on the objectives of the work (routine maintenance or exact determination of the wetland clogging status). A strategy that could be used is to have 4 measurement places along the transects, and to take additional measurements if strong changes are recorded in particular zones. This would make it possible to know exactly where the changes occur. The dimensions and proportions of the wetlands are important for determining both the number of sampling areas and the number of transects for measuring hydraulic conductivity.

The falling-head test is a straightforward technique and does not require additional costly devices. In fact during routine maintenance the method could be made more simplistic still by replacing the use of the pressure probe and datataker with a ruler to measure the decreasing water level inside the tube (in the internal part of the tube) every ten seconds, for example. Although this more straightforward method would indeed be less accurate than if the probe and datataker were used, it would be sufficient for routine maintenance.

The falling-head test is applied in the common practice of civil works and geotechnical studies (Zhao, 1998; Rodgers and Mulqueen, 2006; Kalkan, 2009). It only gives approximate saturated hydraulic conductivity values because vertical conductivity is measured, whereas the wetlands have a horizontal flow. Moreover, hydraulic conductivity could change with depth while the test gives a “global” conductivity of the granular medium plug inside the tube. When the tube is inserted a certain degree of compaction can occur, which is an additional source of error. Despite these disadvantages, in our study we have observed that the changes in hydraulic conductivity in the studied wetlands are of 1 to 3 orders of magnitude. This variation can be successfully detected with the proposed method. These wide ranges have also been reported in other works (Sanford et al., 1995; Drury and Mainzhausen, 2000; Suliman et al., 2006a; Knowles et al., 2008). For practical applications the method is appropriate for recording saturated hydraulic conductivity.

The proposed method requires 1 to 2 days of intensive field work conducted by at least 2 people. The duration of the work depends on the number of transects and the sampling areas. Changes in hydraulic conductivity occur over a long period of time, and therefore for practical purposes measurements should be taken at least once a year depending on whether the wetland is highly clogged and whether the medium is going to be replaced or washed. The method can be more easily carried out by cutting away the macrophytes before starting. In the region where this study was conducted the common practice is to cut the aboveground biomass every year in early winter.

The results revealed very clearly that the Verdú and Corbins wetlands are quite clogged, particularly at Corbins where the hydraulic conductivity in more than a half of the wetland (almost 60% of the length) is lower than 20 m/d. This is related to the higher OLR in this wetland (Table 4.2). Positive relationships between applied loads and the degree of clogging have also been reported in previous studies (Tanner and Sukias, 1995; Nguyen, 2000). It is anticipated that at Corbins the granular medium of the wetlands will have to be replaced or washed relatively soon (at least in the next 3 years) due to the current level of clogging and the fact that the plant was put into operation in 2002.

Caselles-Osorio et al. (2007) conducted some saturated hydraulic conductivity measurements in the two wetlands evaluated in the present study. The re-evaluation of their data in comparison with ours reveals significant differences only in the outlet zone in the wetland at Corbins. Hydraulic conductivity decreased 3 to 4 times from late 2005 (Caselles-Osorio et al., 2007) to March 2007 (when the field work of this study was conducted). The fact that significant differences have been recorded at Corbins and not in Verdú seems to be connected with the higher OLR in Corbins.

The results of the present study show a trend of larger quantity of solids in the inlet than at the other locations from which solids were sampled (Table 4.3). In addtion, hydraulic conductivity was lower in the inlet zone, which suggests an inverse general pattern between hydraulic conductivity and the quantity of solids. However, this inverse pattern is not direct, which is demonstrated by the fact that the places with the lowest TS content were not necessarily the places with the highest hydraulic conductivity (for example compare in Verdú data at ¾ of the length with data at the outlet zone). Therefore, hydraulic conductivity is not exclusively a function of the amount of solids; properties such as density for example may substantially influence hydraulic conductivity (Tanner and Sukias, 1995; Caselles-Osorio et al., 2007; Llorens et al., 2009).

In this study an unexpected systematic decrease in hydraulic conductivity was observed near the outlet zone of both wetlands, which supports the fact that hydraulic conductivity is not a direct function of the amount of solids. There is no clear explanation for this decrease, althought could be mediated (at least in part) by the growth patterns of biofilm following a mechanism that has been recently described by Wallace and Knight (2006). According to these authors, biofilm growths more near the inlet zone (where the loads are higher) and this can entail a local loss of porosity in the inlet zone that force the water to flow onto the medium surface until reaches locations where it can infiltrate again into the medium. This trend can cause extensive ponding and even flooding on the surface of the wetland and therefore maintaining the high hydraulic conductivity of the granular medium in the middle of the wetland (because the water flows throught the top of the granular medium). Near the outlet the water can infiltrate into the granular medium and the biofilm can grow with a subsequent reduction of the hydraulic conductivity. This mechanism approximately matches with the results of the hydraulic conductivity observed in our study, especially in the wetland at Corbins, where it was observed extensive ponding throughout one of the transects.

According to Wallace and Knight (2006) the extent of ponding (and therefore clogging) can be related with the D_{10} of the medium and the cross sectional load in the wetlands. In Verdú and Corbins the wetlands have a granular medium with a D_{10} of 5 mm and the cross sectional loads are greater than 200 g/m^{2}d (which is the limit reccomended for a D_{10} = 5 mm in order to avoid clogging, see Wallace and Knight (2006)). Therefore the cross sectional loads are far above the reccomended limits and subsequently it is predicted that wetlands will have ponding (as it is observed in the field).

The amount of belowground plant biomass was highly variable. No clear relationship was observed between this and hydraulic conductivity trends. In general it is assumed that increasing plant biomass decreases hydraulic conductivity (Brix, 1997; O'Brien et al., 2004). When a wetland is put into operation there is a progressive decrease in hydraulic conductivity, partly due to plant development (Edwards, 1992). However, if the plants are already fully grown, it is difficult to prove that they do indeed influence hydraulic conductivity, which was observed in the present study. Perhaps a more thorough sampling procedure would reduce the local variability and a more clear relationship would have appeared between hydraulic conductivity and plant biomass. Also the fact that the plant root zone was in the upper region of the wetlands may also be a factor in the lack of correlation between belowground plant biomass and the measured hydraulic conductivity.

In the present investigation it was observed that the accumulated solids were mostly of mineral origin, in which the volatile fraction barely represented 25% of the total solids. In fact this trend was already reported for the two evaluated wetlands in the previous study conducted by Caselles-Osorio et al. (2007). Kadlec and Watson (1993) also found a high proportion of the mineral fraction in similar CW systems, while Tannner et al. (1998) and Nguyen (2000) reported values equal to or lower than 20% for the mineral fraction. These findings may be relevant in the context of clogging processes and would need more reseach efflorts because mineral and organic solids likely have very different densities.

The X-ray diffraction analyses revealed that the mineral composition of the accumulated solids was similar to that of the granular medium (mainly quartz and calcite). Yagüe et al. (2005) also analysed the mineral composition of anaerobically digested secondary sludge and found a predominance of calcite and quartz. However, the volatile solids fraction was three times greater than that observed in the present study.

The similarity between the mineral composition of the granular medium and the accumulated solids suggests that the inorganic fraction of accumulated solids could come from fine materials present in the granular medium when it was disposed into the wetlands and/or the disintegration of the granular medium. The dissolution and disintegration of the granular medium in wetlands can related to the formation of acids, mostly linked to the sulphur cycle (dihydrogen sulphide and sulphuric acid), particularly to sulphate reduction. This reaction has been reported as being an important biochemical pathway for the removal of organic matter in the conditions prevailing in horizontal SSF CWs (Caselles-Osorio and García, 2006). The use of machinery for routine maintenance operations such as cutting plants may also promote the dissolution and disintegration of the granular medium.

4.5 Conclusions

In this study a practical method for assessing the degree of clogging SSF CWs was successfully applied to two full-scale wetlands. The method measured saturated hydraulic conductivity using the falling head-test procedure in two transects along the length of the wetlands. When clogging processes become apparent there is a dramatic decrease in hydraulic conductivity along the transects. This change is easily detected using the method proposed in this paper. Furthermore, the falling-head test is straightforward to use and does not require costly devices.

At the time of the investigation, the two wetlands studied were rather clogged, particularly that of Corbins (which treats a larger load), where the hydraulic conductivity in more than half of the wetland was lower than 20 m/d. In both wetlands, the highest hydraulic conductivity values were recorded between the middle zone and a point along the wetland that was almost 4/5 of its length. Another observation was an inexplicable decrease in hydraulic conductivity towards the outlet zone. Despite the clogging of the two studied wetlands, the two systems are very reliable and produce effluents of good quality with TSS and BOD_5 under 25 mg/L.

An inverse trend between solids amount and hydraulic conductivity was observed. This trend, however, is not direct because the places in the wetlands with the lowest TS content were not necessarily the places with the highest hydraulic conductivity.

In both wetlands the mineral fraction of the solids represented in general more than 75% of the total solids. This is due to the presence of fine materials that could come from from fine materials present in the granular medium when it was disposed into the wetlands and/or the disintegration of the granular medium. Therefore it is reccommended the use of granular media clean and with a low calcite content or other degradable minerals.

5

A comparison of *in situ* constant and falling head permeameter tests to assess the distribution of clogging within wetlands[3]

The measurement of saturated hydraulic conductivity has proven to be a suitable technique to assess clogging within horizontal subsurface flow constructed wetlands (HSSF CWs). The vertical and horizontal distribution of hydraulic conductivity have been assessed in two full-scale HSSF CWs by using two different *in situ* permeameter methods (falling head –FH– and constant head –CH– methods). The obtained horizontal hydraulic conductivity profiles show that both methods are well correlated by a power function ($FH = CH^{0.7821}$, $r^2 = 0.76$) within the recorded range of hydraulic conductivities (0-70 m/d), such that the FH method provides values of hydraulic conductivity one to three times lower than the CH method. Despite discrepancies between the magnitudes of reported readings, the relative distribution of clogging within the systems obtained via both methods was similar. Therefore, both methods are useful when exploring the general distribution of clogging and corresponding preferential flow-paths within full-scale HSSF CWs. Greater discrepancy between the methods was evident from the vertical hydraulic conductivity profiles. It is believed this can be attributed to procedural differences between the methods, such as the method of permeameter insertion (twisting versus hammering). Results suggest that clogging develops along preferential flow-paths which correspond to the shortest distance between where water enters and exits the system. Accordingly, the design and maintenance of inlet distributors and outlet collectors appear to have a great influence on the development of clogging of HSSF CWs.

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[3] This chapter is based on the article:

Pedescoll, A., Knowles, P.R., Davies, P., García, J. and Puigagut, J. A comparison of *in situ* constant and falling head permeameter tests to assess the distribution of clogging within horizontal subsurface flow constructed wetlands. Science of the Total Environment, submitted.

5.1 Introduction

Clogging is considered to be the main operational problem associated with subsurface flow constructed wetlands (SSF CWs) for wastewater treatment. Clogging occurs because treatment results in the accumulation of solids, biofilm, plant matter and chemical precipitates which reduce the porosity of the gravel filter media over time (Brix, 1997; Tanner et al., 1998; Nguyen, 2000; Wallace and Knight, 2006; García et al., 2007). In advance stages of SSF CW clogging, symptoms may include sludge accumulation (usually near the inlet), overland flow and decreased treatment efficiency (Platzer and Mauch, 1997; Rousseau et al., 2005; Caselles-Osorio et al., 2007; Kadlec and Wallace, 2009).

Solutions to reverse clogging include gravel extraction and cleaning, *in situ* application of chemical oxidants or filter medium replacement (Cooper et al., 2005; Murphy et al., 2009; Nivala and Rousseau, 2009). In all case the financial investment involved can represent a substantial fraction of the cost of a new system (Cooper, 2009), and for this reason it is essential to assess the degree of clogging in SSF CWs to accurately determine whether intervention is required. Measures of clogging include the analysis of accumulated solids in filter media (Caselles-Osorio et al., 2007), hydrodynamic visualisations by means of tracer tests (Bowmer, 1987: Knowles et al., 2010a) and determination of the hydraulic gradient between points in the filter media (from which the media hydraulic conductivity can be estimated) (Sanford et al., 1995; Rodgers and Mulqueen, 2006; Suliman et al., 2006a).

Ultimately it is desirable to directly measure the hydraulic conductivity of the media, which none of the aforementioned methods can achieve. However, the gravel media in constructed wetlands is non-cohesive, so it is difficult to extract unaltered samples of material to perform standardized and controlled laboratory tests. For this reason, *in situ* procedures have been developed to directly measure the hydraulic conductivity of constructed wetlands (Reynolds et al, 2000). The most commonly applied methods include the Guelph permeameter (Mastrorilli et al. 2001; Langergraber et al., 2003; Ranieri, 2003), a constant head (CH) method (Knowles and Davies, 2009), and a falling head (FH) method (Caselles-Osorio et al., 2007). Each method disturbs the medium to a certain degree; although this disturbance is small compared with that caused during the extraction, storing and transportation of a gravel core to be tested in lab facilities. Ideally these methods are able to measure a wide range of media hydraulic conductivities, as governed by media properties and the stage of clogging (i.e. 0-500 m/d might be typical of the transition between completely clogged and clean media).

This work aims to compare two different methods for the in-situ measurement of saturated hydraulic conductivity for assessing the clogging within horizontal subsurface flow constructed wetlands. Two constructed wetlands in relatively contrasting stages of clogging (advanced clogging and unclogged) were evaluated with each method, to highlight whether the obtained results are influenced by which method is employed. Furthermore, the distribution of hydraulic conductivity throughout the beds will elucidate which design factors affect clogging processes.

5.2 Methods

To assess saturated hydraulic conductivity two different tests were used (falling head and constant head tests) which have been specifically developed to measure the potentially high hydraulic conductivities associated with wetland gravels (Knowles and Davies, 2009; Chapters 3 and 4). Both tests measure the hydraulic conductivity through vertical cores of gravel, although by interpolating between sample points it would be possible to predict a horizontal conductivity based on the assumption that flow would behave identically in both planes. If only the permeability of the wetted gravel is of interest, the unsaturated layer should be removed from the permeameter cell before starting the experiment, and the subsequent analysis modified to reflect the new test conditions.

5.2.1 Falling head permeameter

The falling head permeameter is based on Lefranc's test with falling head (NAVFAC, 1986) and has been used to measure the saturated hydraulic conductivity of constructed wetlands (Chapters 3 and 4). It consists of timing a falling column of water between points along the height of a permeameter cell (100 cm long and 10.5 cm diameter steel tube with a perforated wall over the length which is immersed into the gravel). The cell is hammered into granular medium until the desired wetted depth within the bed is reached. A pressure probe (TNS119, Desin Intruments S.A., Barcelona) connected to a laptop by means a data logger (Datataker DT50, Thermo Fisher Scientific Australia Pty Ltd, Melbourne), is placed inside the pipe on the surface of the bed and used to measure the water level inside the tube (Figure 5.1). A pulse of water added to the tube will yield a negative exponential curve between water level and time, the slope of which is related to the hydraulic conductivity according to Lefranc's formula:

$$K = \frac{d^2 \ln(2L/d)}{8Lt} \ln\frac{h_1}{h_2} \tag{5.1}$$

Where:

K is the hydraulic conductivity of the studied material, in m/s; d is the diameter of the tube, in m; L is the submerged length of the tube, in m; h_1 is the height of the water table level inside the tube at time zero, in m; h_2 is the height of the water table level inside the tube at time *t*, in m; and *t* is time, in s.

The quadratic difference between the theoretical curve and that obtained in the field is minimized to estimate the value of the hydraulic conductivity.

$$\sum_{t=0}^{t=n} \left(h_2 - f(h_2)\right)^2 \to 0 \qquad (5.2)$$

where,

f (h_2) is the modelled data, in m; *t* is the time, in s.

Measurement of vertical hydraulic conductivity using the falling head permeameter can be achieved via two techniques: 1) measuring the hydraulic conductivity at a first depth (K_1), extracting the gravel above this depth and repeating the test at a second depth by further submersing the tube (K_2); or 2) inserting the tube at two different depths (K_1 and K_T) without extracting the gravel, and deducing K_2 by using Equation 3:

$$\frac{1}{K_T} = \frac{1}{K_1} + \frac{1}{K_2} \qquad (5.3)$$

Where:

K_T is the total hydraulic conductivity (obtained with the second test), in m/s; K_1 is the hydraulic conductivity at the first depth (obtained with the first test), in m/s; and K_2 is the hydraulic conductivity at the second depth, in m/s.

Experimental procedure

The instructions followed when performing the experiment are listed below, where the labels in square brackets refer to the components illustrated in Figure 5.1.

1. Hammer the permeameter cell [a] into the gravel to a depth of 400 mm. The end of the pipe is sharpened to help its insertion.

2. Place the pressure probe [b] into the pipe, resting on the gravel surface.

3. Connect the probe to a laptop [c] by means of a data logger [d]. Check the pressure value at atmospheric pressure. Take values every second or more depending on the hydraulic conductivity expected (a minimum number of 100 measurements is recommended for accurate calculation of hydraulic conductivity).

4. Fill the pipe with a pulse of water and measure the decrease of level inside the tube via the pressure probe.

5. Use Equation 2 and an iterative procedure to fit a value of gravel core hydraulic conductivity which best describes the obtained dataset. This can be done easily using either the Solver or Goal-Seek functions in MS Excel™.

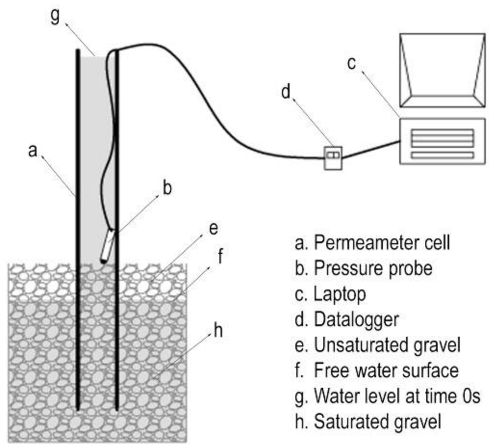

Figure 5.1 Experimental set-up for in-situ determination of the hydraulic conductivity with falling head permeameter.

5.2.2 Constant head permeameter

A constant head permeameter has also been used to asses the saturated hydraulic conductivity in constructed wetland filter media (Knowles et al., 2010a). A permeameter cell made of a non perforated PVC tube (50 cm length) is inserted into the gravel by twisting until a 40 cm core is encapsulated. A water reservoir (with a graduated measuring tube) discharges water into the permeameter cell via the Mariotte Siphon technique (Figure 5.2), thus enabling a constant head to be maintained. The hydraulic conductivity of the encapsulated gravel core can be measured via Darcy's Law, by timing the discharge from the reservoir required to maintain the constant head.

$$K_T = \frac{\dot{q} L_{CELL}}{A_{CELL} h_T} \tag{5.4}$$

Where:

K_T is the hydraulic conductivity of the gravel core, in m/s; $\dot{q}$ is the flow in the reservoir, in m^3/s; L_{CELL} is the submerged length of the permeameter cell, in m; A_{CELL} is the cross-sectional area of the permeameter cell, in m^2; and h_T is the total head loss across the permeameter cell at constant head, in m.

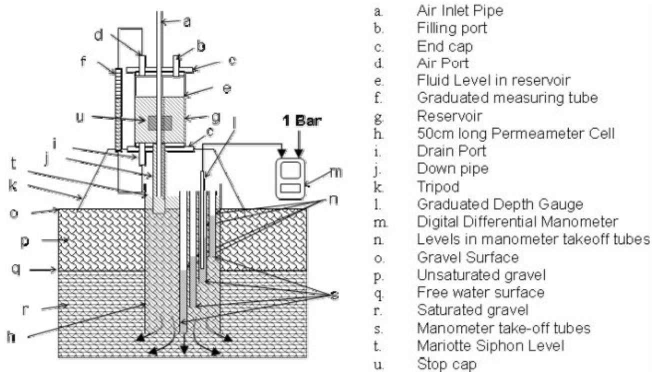

Figure 5.2 Experimental set-up for in-situ determination of hydraulic conductivity with the constant head permeameter method (not to scale: reservoir stands approx. 1 m off the ground). Note: this figure has been adapted from Knowles and Davies (2009).

Inserting two piezometer take-off tubes into the permeameter cell at different depths (in this study 20 and 40 cm), within which are inserted two digital differential manometer depth probes, allows the vertical hydraulic conductivity profiles to be calculated from the following formula:

$$K_n = \frac{1}{n\left(\frac{h_n}{h_t K_T}\right)} \tag{5.5}$$

Where:

K_n is the hydraulic conductivity at each depth, in m/s; n is the number of sections or core divisions (in this case 2); h_n is the vertical head loss across each depth, in m; h_T is the water level into the permeameter cell at constant head, in m; and K_T is the hydraulic conductivity of the gravel core, in m/s.

Four measurements per depth are required to deduce the vertical hydraulic conductivity profile along the gravel core, as depicted by Figure 5.3. Before the experiment begins record the digital manometer readings after they have stabilised (static water level), and the distance the probes have been inserted into the take-off tube. Any readings obtained should be roughly equal. Any disparity between the readings will be caused by minor differences between the vertical alignments of the top of the take-off tubes, and therefore, recording the different static water level readings will allow these discrepancies to be accounted for. Once the experiment has started, record the water level inside the permeameter cell so that the total applied head across the permeameter cell can be calculated. Finally record the digital manometer readings after they have re-stabilised (dynamic water level). This stage must be completed before the reservoir empties.

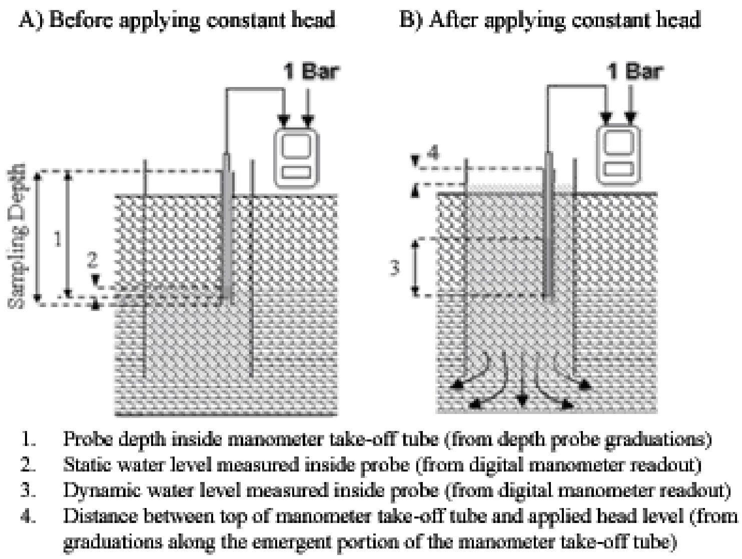

Figure 5.3 Measurements that are taken during the experiment, depicted for one take-off tube. Corresponding readings will need to be taken in each individual take-off tube. N.B. For clarity, the reservoir device which maintains the constant head has been omitted from the graphic. A) Before applying constant head; B) After applying constant head. Note: this figure has been adapted from Knowles and Davies (2009).

Further details about the exact experimental procedure and apparatus depicted in Figure 5.2 can be found in Knowles and Davies (2009).

5.2.3 Hydraulic conductivity measurements

The two tests were used to profile the hydraulic conductivity of two different SSF CWs operated by Severn Trent Water (a UK water utility provider) in Warwickshire, UK, which are used for the tertiary treatment of municipal wastewater in decentralised wastewater treatment plants (Table 5.1). The treatment plants at Fenny Compton (FC) and Moreton Morrell (MM) include parallel networks of tertiary SSF CWs which provide downstream support for secondary treatment Rotating Biological Contactors, although MM serves a population roughly double that of FC (1500 versus 800 PE). The four CWs at MM and two CWs at FC have been in operation since 1993-94, although those at FC underwent a partial refurbishment in February 2007 due to clogging issues, whereby the top 20 cm of gravel was extracted and replaced with clean media (notably, the replacement media was of a larger specification than the original media). All the wetlands are planted with common reeds (Phragmites australis). One CW was chosen from each site and each hydraulic conductivity test applied across a 4x4 matrix of sampling points (Figure 5.4), measuring at two different depths (20 and 40 cm).

Table 5.1 Characteristics of the wetlands studied.

	Fenny Compton	Moreton Morrell
Wastewater treatment	Tertiary	Tertiary
Population (PE)	800	1500
Set in operation	1993 (replacement of the top 20 cm of gravel in 2008)	1994
Flow (m^3/d)	120	134
Surface area (m^2)	450	225
Length to width ratio	2.9	1.0
Depth (m)	0.6	0.6
Gravel media size (mm)	3-6 upper 0.2 m 6-12 lower 0.4 m	3-9

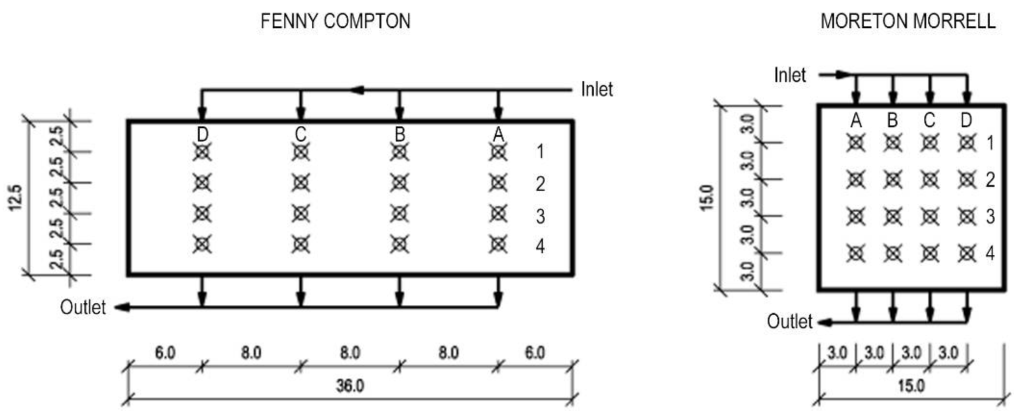

Figure 5.4 Plan view of the two studied wetlands and the matrix of sampling points. Distances in m.

5.3 Results

5.3.1 Horizontal profile maps

Horizontal profile maps, depicted in Figures 5.5 and 5.6, represent the nominal hydraulic conductivity of the entire 40 cm gravel cores measured using each method, and do not delineate variations in vertical hydraulic conductivity. Horizontal hydraulic conductivities range between 0-70 m/d and 0-35 m/d for FC and MM, respectively. However, the average core conductivity of MM was three times lower than those of FC (5.22±9.26 and 18.65±21.21 m/d measured with CH method for MM and FC, respectively).

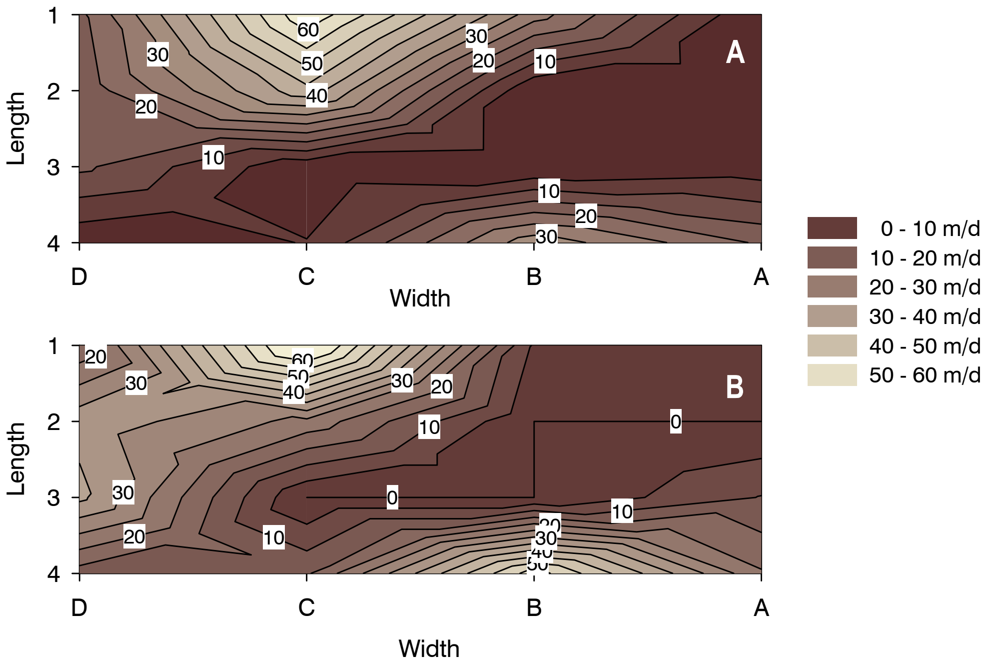

Figure 5.5 The horizontal hydraulic conductivity profile (m/d) of Fenny Compton obtained with A) falling head permeameter and B) constant head permeameter. The water flow direction is from the top to the bottom of the graph.

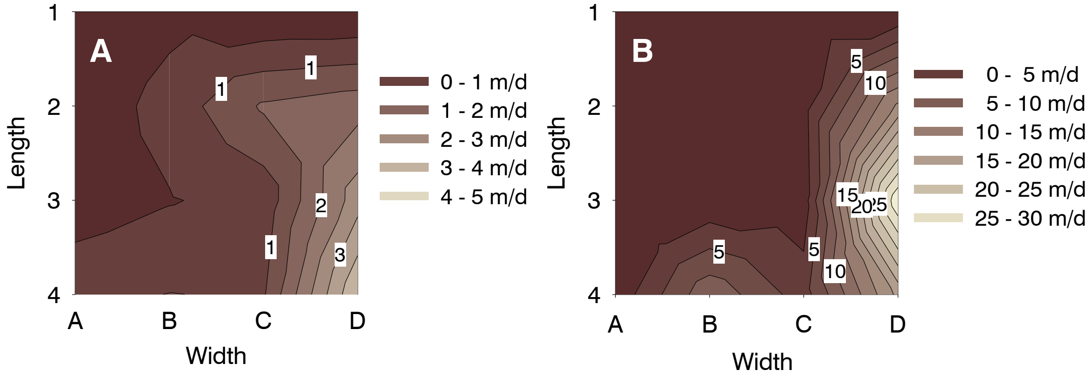

Figure 5.6 The horizontal hydraulic conductivity profile (m/d) of Moreton Morrell obtained with A) falling head permeameter and B) constant head permeameter. The water flow direction is from the top to the bottom of the graph.

At the studied sites both methods describe the same hydraulic conductivity pattern. Specifically, at FC (Figure 5.5) a region of preferential clogging (corresponding to a preferential flow path) extends from where flow enters the system at the far right of the inlet distributor (circa point A1), diagonally through the media, towards where the water exits the system at the far left of the effluent collection pipe (circa point D4). This preferential flow path corresponds to values of hydraulic conductivity ranging from 0 to 12.5 m/d and 0.07-1.2 m/d for CH and FH methods, respectively. In the case of MM (Figure 5.6) a preferential flow path exists along Transect A (from point A1 to point A4), with both methods returning hydraulic conductivity values below 1 m/d.

5.3.2 Vertical profile maps

The obtained vertical hydraulic conductivity profiles for FC differ depending on the method employed, both in terms of magnitude and distribution of clogging (Figure 5.7). The FH method describes that the hydraulic conductivity of the top 20 cm is ca. 10 times higher than the bottom 20 cm of gravel (65-450 m/d and 0.05-95 m/d, respectively). However, these differences were less pronounced according to the CH method, (0-90 m/d and 0-190 m/d for the top 20 cm and bottom 20 cm, respectively).

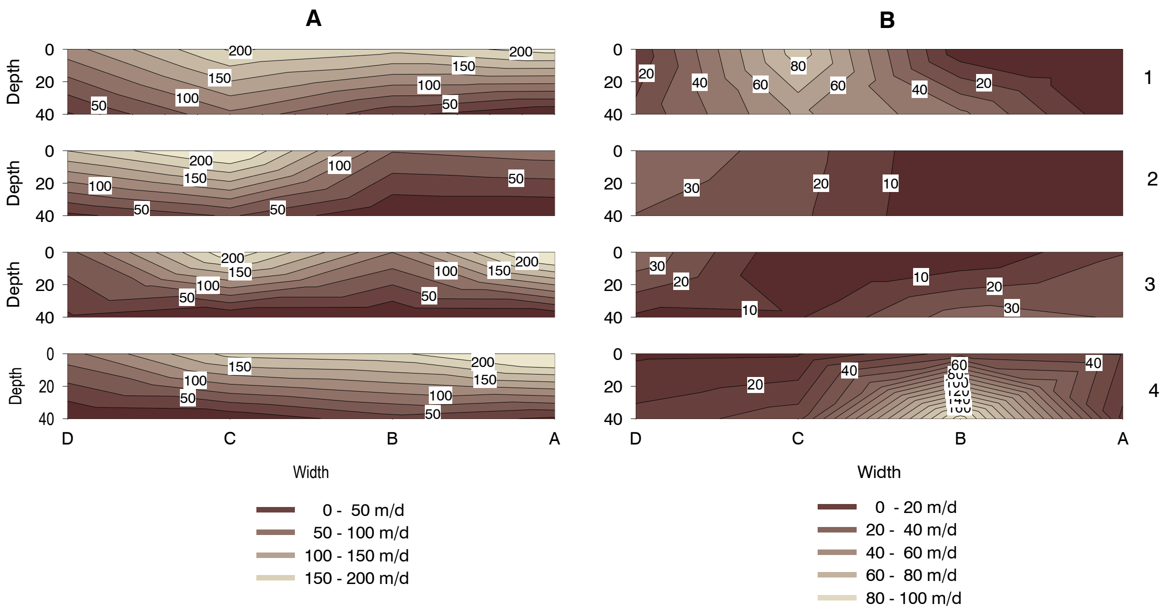

Figure 5.7 The vertical hydraulic conductivity profiles (m/d) obtained at Fenny Compton with A) falling head permeameter and B) constant head permeameter. Note that each plot is a vertical section of the wetland at different locations. Therefore the water flows from back to front of the graph.

At MM the vertical hydraulic conductivity profiles (Figure 5.8) also differed depending on the method applied. According to the CH method a preferential flow path was registered along transect A. Along this transect conductivities lower than 1 m/d were registered across the top 20 cm, whereas conductivity ranged between 0-5 m/d in the bottom 20 cm of gravel. The highest conductivities were recorded along transect D, with values ranging from 0.05 to17.4 m/d in the top 20 cm, and from 0.15-100 m/d in the bottom 20 cm. According to the FH method the lowest conductivities were also registered along transect A (with values from 0.45-6.53 m/d in the top 20 cm and 0.02-0.09 m/d in the bottom 20 cm). The highest conductivities were recorded along transect D; ranging from 0.97-11.75 m/d in the top 20 cm and between 0.01-6.07 m/d in the bottom 20 cm.

Generally, the highest conductivities were recorded at the outlet areas of both studied wetlands, irrespective of the applied method.

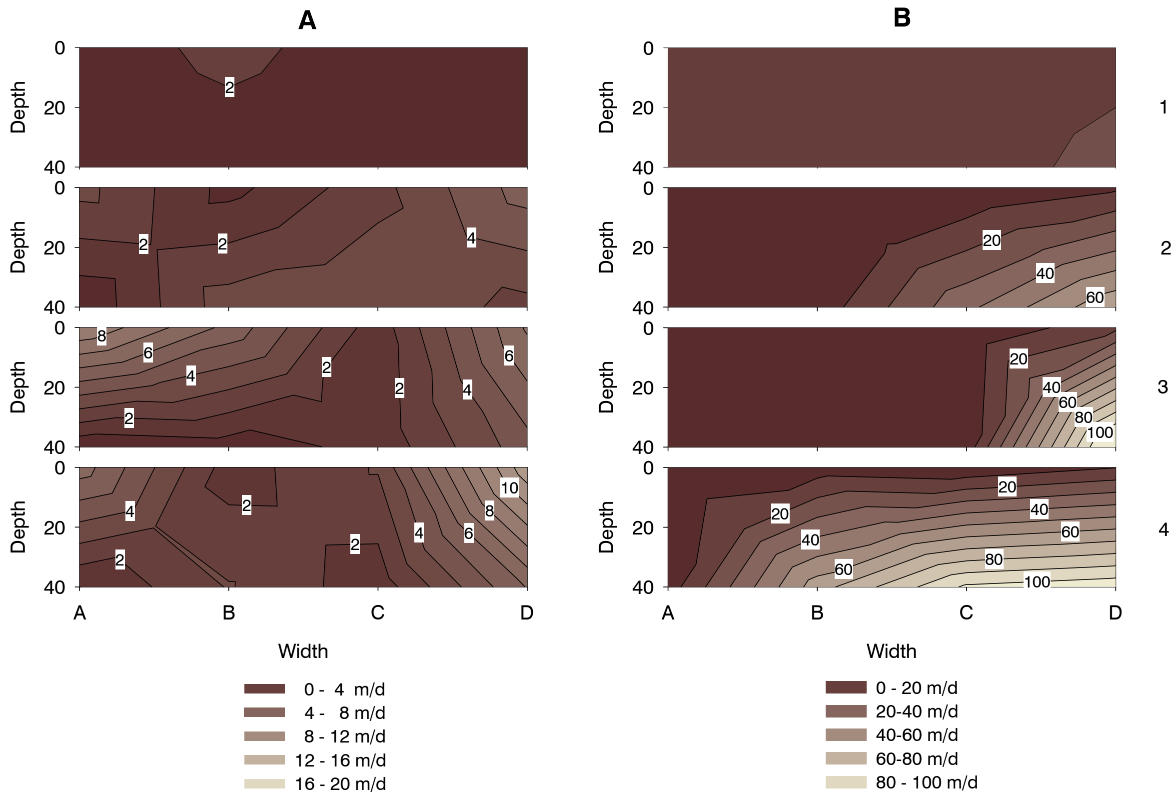

Figure 5.8 The vertical hydraulic conductivity profiles (m/d) obtained at Moreton Morrell with A) falling head permeameter and B) constant head permeameter. Note that each plot is a vertical section of the wetland at different locations. Therefore the water flows from back to front of the graph.

5.4 Discussion

5.4.1 Comparison of methods

For each system studied, the horizontal hydraulic conductivity distributions obtained via each method are similar (Figure 5.5 and 5.6). These results suggest that both methods are useful when trying to identify general patterns of clogging and preferential flow-paths in the primary (longitudinal) flow direction. However, as emphasised in Figure 5.9, the FH method generally produces lower hydraulic conductivity measurements than the CH method, especially for the more clogged CW at MM. This phenomenon may be attributable to the differing manner by which the permeameter cell is immersed into gravel during each method (Carter and Ball, 1993). In the CH method a twisting action is used which may act to shear clog formations and hence artificially increase hydraulic conductivity of clogged media. Conversely, the FH method uses a hammer action which may compact clog formations, thus artificially decreasing the hydraulic conductivity of clogged media. This would also explain why the disparity between the methods was not so apparent in the relatively unclogged media of FC. Unfortunately, no experiments were conducted to test this hypothesis and, therefore, further research should be carried out to determine the extent of the effect of permeameter cell insertion on hydraulic conductivity discrepancies between both methods.

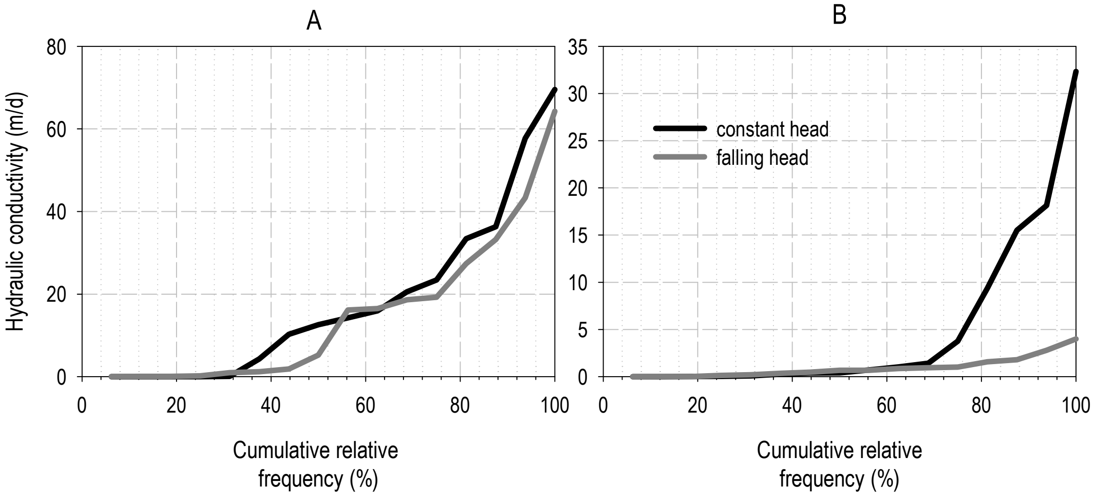

Figure 5.9 Cumulative relative frequency distribution of the hydraulic conductivities obtained for both methods (n=16 per plot) in A) Fenny Compton and B) Moreton Morrell.

Discrepancies between the methods are more clearly emphasised by comparison of the vertical hydraulic conductivity profiles, which demonstrated variance in both distribution and magnitude (Figure 5.7 and 5.8). At FC, the top 20 cm of gravel was replaced one year before the experiments were carried out, and therefore it would be logical to assume that the upper gravel strata would be more conductive than the lower gravel strata (especially because the replacement gravel was of a larger size than the original gravel). However, these differences were only detected with the FH method (with hydraulic conductivity values of 65-450 m/d and 0.05-95 m/d for the top and the bottom, respectively) (Figure 5.7). Contrastingly, MM demonstrated advanced symptoms of clogging in the upper gravel media, including surface sludge formation and overland flow, and it would therefore be logical to assume that the upper gravel strata would be less conductive than the lower gravel strata (a widely abounded theory regarding horizontal SSF CW clogging mechanisms - Kadlec and Wallace (2009)). However, this trend was only identified with the CH method, with the FH method generally suggesting that values in the upper strata were more conductive.

Again, these observations may be artefact of procedural differences between the methods. The FH method repeats tests at different depths to obtain the vertical conductivity profile. The multiple insertions and potential for solids washout in the upper layers during the first test may explain why the results of the FH method suggested that the upper media at MM was more conductive than the lower media. In the CH method the values at different depths are calculated from the difference between the dynamic water levels inside each piezometer takeoff tube. Thus, in order to obtain a reliable value the level inside the takeoff tubes must be stable, however, at very high media conductivities it is possible that the recharge reservoir empties before the manometer completely stabilises. To avoid this inconvenience the reservoir should be enlarged. Another potential issue at FC may arise because testing took place imminently after a snowmelt (February 2009), where the combination of cold temperatures and high hydraulic loads may have had an eclectic effect on hydraulic conductivity distribution.

Interestingly, in Chapter 3 it was reported a disparity between laboratory versions of the FH and CH permeameter test which may support the above findings. The laboratory FH test provided lower conductivity values than the CH method for low conductive material (a D_{50} of 0.9 mm sand with a hydraulic conductivity of 35 m/d), however, the opposite trend was observed in highly conductive materials (a D_{50} of 7.1 mm gravel with a hydraulic conductivity of 265 m/d), with the CH method returning lower hydraulic conductivity values than the FH method. Therefore the observed effects, i.e. the CH method detecting low conductivities in the upper strata of FC and the FH method detecting high conductivities in the upper strata of MM, may be explainable due to a similar effect. Evidently, care must be taken while extrapolating the relation between both methods since it may vary according to the infiltration properties of the media tested and the clogging status of the wetland analysed.

Figure 5.10 compares the results of the horizontal hydraulic conductivity profile obtained by each method to elucidate whether the discrepancy between methods can be quantified. As evident from Figure 5.10, across the measured range of hydraulic conductivities (from 0 to 70 m/d) the results of the FH and CH methods are well related via a power function ($FH = CH^{0.7821}$), with a regression of $r^2 = 0.76$. From this correlation it is possible to calculate that differences in hydraulic conductivity measurements between the two methods generally differ by a factor of one to three; which is an acceptable discrepancy (Elrick and Reynolds, 1992; Reynolds and Zebchuk, 1996; Elrick et al., 2002). Note that for these calculations absolute zeros were excluded. The power function elucidates that the discrepancy between FH and CH results increases as the hydraulic conductivity value increases. Furthermore, this function means that in clogged systems with lower hydraulic conductivities, the CH method will be able to describe small differences in hydraulic conductivity with a greater resolution than the FH method.

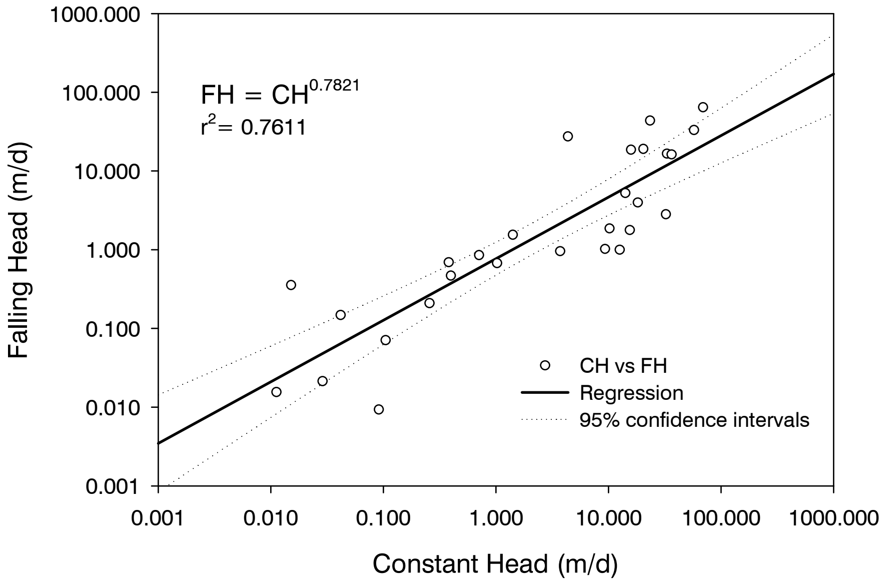

Figure 5.10 Regression curve and 95% confidence intervals of the comparison between experimental values of hydraulic conductivity obtained via the falling head and constant head methods (n=28). Note that axes are represented in logarithmic scale.

5.4.2 Clogging distribution

The horizontal hydraulic conductivity profile for FC indicate a range of values between 0-70 m/d, whereas the range of values at MM was 0-35 m/d; aptly reflecting that MM was in a more advanced stage of clogging than FC (Figure 5.5 and 5.6). These values are in accordance with previous studies performed on the same wetlands (Knowles and Davies, 2009; Knowles et al., 2010a) and with studies performed at wetlands with similar characteristics in terms of treatment type, dimensions and substrate size (Caselles-Osorio et al., 2007).

Results indicated that preferential flow paths had developed in both wetlands, which generally corresponded to the shortest distance between where flow enters and exits the system and thus supports the idea that flow finds the path of least resistance (Figure 5.5 and 5.6). At FC the most clogged areas were diagonally distributed from point A1 at the inlet to point D4 at the outlet. At MM clogging was localised along transect A from point A1 to point A4.

It is thought that this effect is exacerbated by inlet distributor and effluent collector design. At FC flow is distributed by a pipe with 7 points and water flows within the pipe from transects A to D (Figure 5.4). Since the pipe is flatly emplaced on the gravel surface there is no compensation for internal pipe losses and the majority of the water tends to flow onto the bed near point A1. Thus, at point A1 the registered hydraulic conductivity was the lowest for the entire inlet area, irrespective of the method employed (Figure 5.5 and 5.6) (0.1 and 0.07 m/d for constant head and falling method, respectively). Water entering the effluent collection pipe would flow along the pipe in the direction from transects A to D in order to exit the system. Again, no provision is made for internal pipe losses and the water tends to exit at point D4 where hydraulic conductivity was the lowest for the entire outlet area, irrespective of the method used (12.57 and 0.99 m/d for constant and falling head method, respectively). At MM water is distributed by a pipe with four vertical distribution risers and the influent flows along the pipe from transects A to D., Larger solids tend not to travel up the risers and instead are conveyed towards the end of the pipe, with the result that during the field campaign all of the risers with the exception of that near point A1 were completely clogged.

The horizontal flow patterns observed in the present work for the surveyed systems (Figure 5.5 and 5.6) are in accordance with previous studies in two different HSSF CWs for secondary treatment in Spain (data not published). These authors attributed the observed preferential flow paths to uneven influent distribution at the inlet and to the aspect ratio (W:L) of the beds. They concluded that aspect ratios less than 1 can compensate for uneven influent distribution by damping the influence of transverse short-circuiting. Fenny Compton and Moreton Morrell have aspect ratios of 3 and 1 respectively (Table 5.1) and resultantly the observed clogging profiles suggest that transverse short-circuiting is prevalent in these systems. Consequently, aspect ratio and flow distribution systems will greatly influence the development of clogging within a wetland.

Regarding the vertical distribution of clogging in each system,, at FC the hydraulic conductivity at the surface of the wetland (top 20 cm) was 10 times higher than at the bottom (due to a complete replacement of the surface gravel one year before the present study) (Figure 5.7a). However, at MM the relation was opposite, with lower hydraulic conductivity at the surface of the system (Figure 5.7b). This can be ascribed to the surface based influent distributors that feed these systems and result in the formation of a clog matter layer on the surface of the gravel that thickens with time. Indeed, during the field campaign, sludge accumulation and overland flow were visually detected at the inlet area of MM. Subsurface based influent distributors, such as infiltration chambers, have been recommended by some authors to prevent this occurrence (Wallace and Knight, 2006).

Sludge layer formation can be exacerbated by uneven influent distribution. This was evident at MM where the inlet area in the vicinity of the only functional riser pipe (point A1) corresponded to the thickest sludge layer accumulation. For this reason, Severn Trent Water have identified that maintenance of influent distributors is paramount, and newly designed systems have incorporated trough style influent distributors which improve distribution uniformity and are easier to maintain than vertical risers (Griffin et al., 2008). Other factors that may affect the accumulation of sludge at the inlet of HSSF CWs include deposition of reeds (which are not harvested) and the solids loading rate (Cooper et al., 2005). Note that these wetlands are systems for tertiary treatment which should receive influent with relatively low BOD and TSS concentrations (Cooper, 2009). However, typical TSS loads to HSSF CWs for tertiary treatment in the UK range from 2 to 30 g TSS/m^2d (Knowles et al., 2010), which is similar to that described for secondary treatment units (between 2.2-10 g TSS/m^2d) (Tanner et al., 1998; Caselles-Osorio et al., 2007). The atypically high BOD and solids loads to these tertiary systems are due to the regular presence of biological flocs (especially during wet weather) that are sloughed from secondary treatment units upstream of the wetland.

5.5 Conclusions

Preferential flow paths and corresponding clogging profiles in CWs can be assessed by the *in situ* application of either constant head or falling head permeameter methods. However, discrepancies between the apparatus and procedures involved in each method can affect the range of obtained results. Accordingly, for the horizontal hydraulic conductivity profiles (range of values 0-70 m/d), the falling head method provides values one to five times lower than those obtained by the constant head method. This suggests that the CH method offers finer resolution than the FH method when studying small variations of hydraulic conductivity in clogged media. Despite this, the datasets obtained via the two methods can be well correlated by a power function (FH = $CH^{0.7821}$, r^2= 0.76). It is believed that the differing method of permeameter insertion (twisting versus hammering) is largely accountable for the discrepancy.

Vertical profiles of hydraulic conductivity measured with the falling head and constant head methods differ both in terms of magnitude and relative distribution. It appears that the falling head method produced more logical results at Fenny Compton (hydraulic conductivity was highest in the recently replaced upper gravel layers), whereas the constant head method produced more logical results at Moreton Morrell (hydraulic conductivity is lowest in the upper gravel layers were surface layer accumulation occurs). It is believed this is inherent in the difference between constant head and falling head methods, although more work is required to ascertain the exact cause of this variability.

Both methods provide results which support that the development of clogging in these systems is highly influenced by the design and maintenance of the influent distribution and effluent collection systems. Generally speaking, flow follows the shortest path (that of least resistance) between entering and exiting the system, which due to the large aspect ratios (W:L) and uneven influent distribution at FC and MM results in acute transverse short-circuiting. Therefore, this study emphasizes the need for improved HSSF CW designs in order to ameliorate the hydraulic behaviour and longevity of these systems.

6

Improving contaminant removal according to primary treatment and operational strategy in horizontal wetlands[4]

This study aimed at evaluating the contaminant removal efficiency of shallow horizontal SSF CWs as a function of 1) primary treatment (hydrolytic upflow sludge blanket (HUSB) reactor vs. conventional settling) and 2) operation strategy (alternation of saturated/unsaturated phases vs. permanently saturated). An experimental plant was constructed and operated over a period of 2.5 years. The plant had 3 treatment lines: a control line (settler-wetland saturated), a batch line (settler-wetland operated with saturated/unsaturated phases) and an anaerobic line (HUSB reactor-wetland saturated). In each line wetlands, planted with common reed, with surface area of 2.80 m^2 and a water depth of 25 cm, were operated at a hydraulic and organic load of 28.5 mm/d and 4.7 g BOD/m^2d, respectively. Effluent redox potential was lower for the anaerobic line (-45±78 mV) than for the other two lines (3±92.7 and -5±71 mV for control and batch, respectively). Overall, chemical oxygen demand (COD), biochemical oxygen demand (BOD_5) and ammonium mass removal efficiencies were slightly greater for the batch line (88%, 96% and 87%, respectively) than for the control (83%, 94% and 80%) and the anaerobic line (80%, 87% and 73%) and higher during cold seasons (30% and 50% higher for COD and ammonium removal than in the control line, respectively). The implementation of a HUSB reactor as primary treatment did not enhance the treatment capacity of the system (in comparison with a conventional settler), while the efficiency of horizontal SSF CWs can be improved using a batch operation strategy.

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[4] This chapter is based on the article:

Pedescoll, A., Corzo, A., Álvarez, E., Puigagut, J., García, J. Improving contaminant removal according to primary treatment and and operation strategy in horizontal subsurface flow constructed wetlands. Ecological Engineering, submitted.

6.1 Introduction

Subsurface flow constructed wetlands (SSF CWs) are extensive systems used for wastewater treatment that are particularly suitable for the sanitation of small communities (Rousseau et al., 2004; Puigagut et al., 2007; Vymazal and Kröpfelová, 2009). Contaminant removal efficiency attained by wetlands (such as ammonium removal) depends to a large extent on the redox status of the systems (Caselles-Osorio and García, 2007a; Dušek et al., 2008). Accordingly, more oxidising conditions lead to greater contaminant removal efficiencies. The removal of certain contaminants, such as ammonium, is especially affected by the redox status.

Horizontal SSF CWs are complex bioreactors in which the removal of organic contaminants occurs mainly by means of anaerobic reactions and, to a lesser extent, by means of anoxic and aerobic pathways (Calheiros et al., 2009). Water depth is one of the main design factors of horizontal SSF CWs influencing the redox status of the system (Faulwetter et al., 2009). Accordingly, the lower the depth the higher the oxidising conditions within the bed (Singh et al., 2009). Studies on experimental and pilot scale have repeatedly proven that shallower horizontal SSF CWs wetlands (water depth about 0.2 to 0.35 m) provide higher removal efficiency than conventional horizontal SSF CWs (water depth about 0.5 to 0.7 m), and this has been linked to the more oxidising conditions present in these shallower systems. The higher efficiency of shallow horizontal SSF CWs has been demonstrated for a wide variety of contaminants such as organic matter (COD and BOD_5) and ammonium (García et al., 2004, 2005; Albuquerque et al., 2009), linear alkylbenzene sulfonates (Huang et al., 2004), pharmaceuticals (Matamoros et al., 2005), and estrogens (Song et al., 2009). Overall, conventional horizontal SSF CWs with a water depth of about 0.5 to 0.7 m are considered to rarely have ammonium removal efficiencies above 40% to 55% because of their inability to provide enough oxidised conditions for nitrification (Austin and Nivala, 2009; Vymazal and Kröpfelová, 2009). On the other hand, shallow SSF CWs have been described as providing ammonium removal efficiencies higher than 80% (Caselles-Osorio and García, 2006, 2007a, 2007b).

Primary treatment is a previous key step to horizontal SSF CWs that is most often achieved by using septic or Imhoff tanks (Knowles et al., 2010b). However, other technologies, such as upflow anaerobic sludge blanket (UASB) reactors, have recently been investigated as a suitable primary treatment for SSF CWs, mainly because they can provide effluents with lower total suspended solids (TSS) and COD concentrations than standard primary treatments (Barros et al, 2008). Another suitable primary treatment used in combination with SSF CWs is the hydrolytic upflow sludge blanket (HUSB) reactor, though it has been little investigated within the context of treatment wetland technology (Álvarez et al., 2008a). HUSB reactors are essentially

UASB reactors operated at a lower hydraulic retention time (HRT) (from 2 to 5 hours) in order to avoid methanogenesis reaction wherever possible. In general, solids retention time (SRT) in HUSB reactors is maintained for over 15 days in order to achieve high hydrolysis rates of wastewater solids.

Field experience gained during the last few years has demonstrated that alternating unsaturated phases during the operation of SSF CWs is of capital importance to maintaining aerobic conditions within the wetlands (Molle et al., 2005). Accordingly, Behrends et al. (1993) reported reaeration rates four times faster in drain and fill conditions than in static conditions. This result was explained by a rapid oxygenation of the wetted gravel that was exposed to atmospheric oxygen during the drain period. Moreover, several studies have shown that wetlands operated under batch conditions (more oxidised conditions) perform better than those operated under continuous conditions (Tanner et al., 1999; Stein et al., 2003; Caselles-Osorio and García, 2007a). Higher contaminant removal efficiencies reported for batch-operated systems have also been linked to higher water level fluctuations (mediated by evapotranspiration) in comparison with continuous feeding systems. These fluctuations expose more granular medium to the atmosphere, thus promoting more oxidised conditions within the bed (Breen, 1997; Tanner et al., 1999). Vymazal and Masa (2003) found that changes in water level for batch-operated field-scale horizontal SSF CWs had a positive effect on the removal of several contaminants, including COD and ammonium.

This study aims to evaluate the contaminant removal efficiency of experimental shallow horizontal SSF CWs as a function of: 1) type of primary treatment (HUSB reactor or conventional settling) and 2) operating strategy (alternating batch-unsaturated phases and permanently saturated). For this purpose an experimental plant was constructed and operated over a period of two and half years after system start-up. To our knowledge, this is the first time that the effect of the type of primary treatment and operating strategy has been evaluated on rigorously controlled horizontal SSF CWs (in terms of hydraulic and organic loading rates) and over a sufficiently long period of time. Our aim was to obtain remarkable results useful for improving the design and operation of constructed wetlands.

6.2 Methods

6.2.1 Experimental plant

The plant used is set in the open at the experimental facility of the Department of Hydraulic, Maritime and Environmental Engineering of the Universitat Politècnica de Cataluya, Barcelona, Spain. Built in 2006, the plant became operational in February 2007 and treats urban wastewater pumped directly using 2 pumps from a nearby municipal sewer. Firstly, the wastewater is coarsely screened and subsequently stored in a 1.2 m^3 plastic tank, which is continuously stirred in order to avoid sedimentation of solids. This tank is equipped with level buoys that control the operation of the feeding pumps. Accordingly, the pumps start automatically when the volume of wastewater in the tank is about 600 L and stop when the volume is about 1000 L. Wastewater retention time in this tank is approximately 12 hours. From the storage tank, the wastewater is conveyed to 3 different treatment lines which, for ease of understanding, have been named batch, control and anaerobic lines (Figure 6.1). Differences between treatment lines are related to the type of primary treatment and the operation strategy applied.

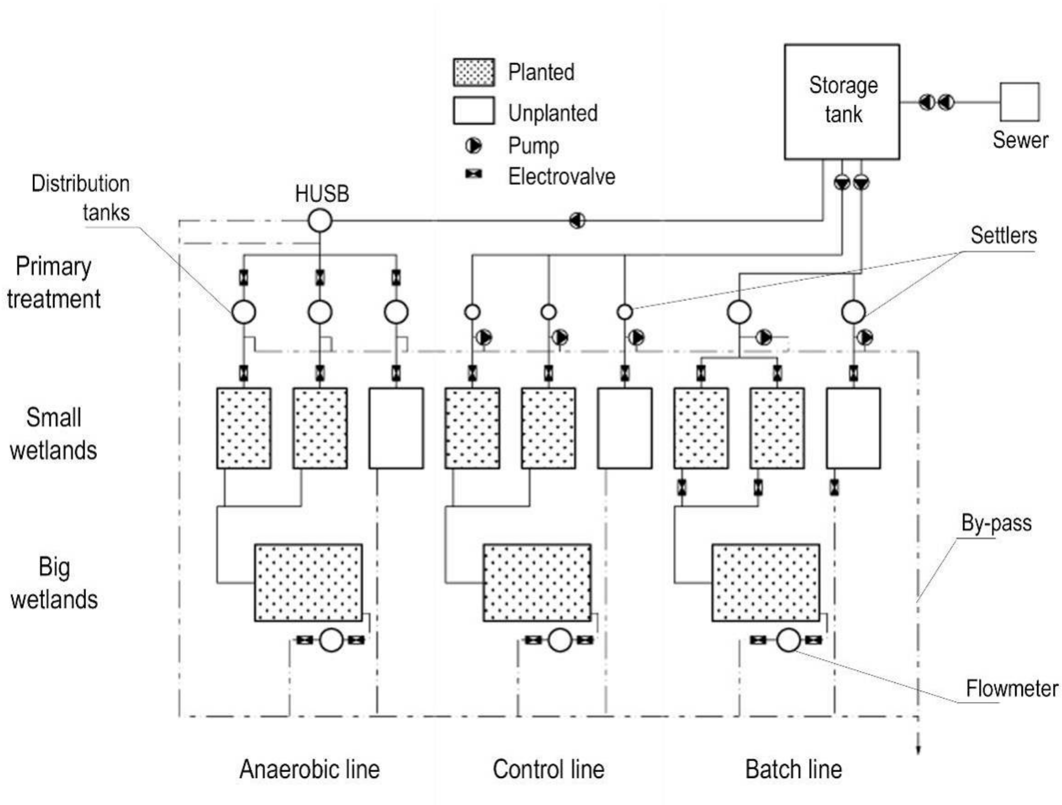

Figure 6.1 Schematic diagram of the experimental plant.

The layout of the wetlands is the same in all three lines: three small wetlands in parallel (0.65 m^2 each), two of them connected to a big wetland in series (1.65 m^2) (Figures 6.1 and 6.2). One of the three small wetlands was left unplanted and discharges directly to the sewer and, strictly speaking, does not belong to the treatment lines. These unplanted wetlands were constructed in order to study plant influence on clogging processes. However, clogging processes fall outside the scope of this paper and unplanted wetlands will therefore not be considered here. The two small parallel wetlands were necessary for the operation of the batch line. This system was also adopted in the other two lines for comparative purposes. These two small wetlands have a joint surface area (1.3 m^2), which is approximately 45% of the total surface area of the treatment line (2.95 m^2). Note that the appearance of clogging in this type of wetlands is more evident in the inlet zone of the wetlands (Caselles-Osorio and García, 2007b), and this is why the total surface of the wetland area was split in two (one big and two small wetlands). Each of the three lines received a flow of 84 L/d (not including the unplanted wetlands).The wetlands were therefore operated at a hydraulic loading rate of 28.5 mm/d. The wetlands of each line were designed to have a maximum surface loading rate of approximately 6 g BOD/m^2.d, as recommended by García et al. (2004 and 2005).

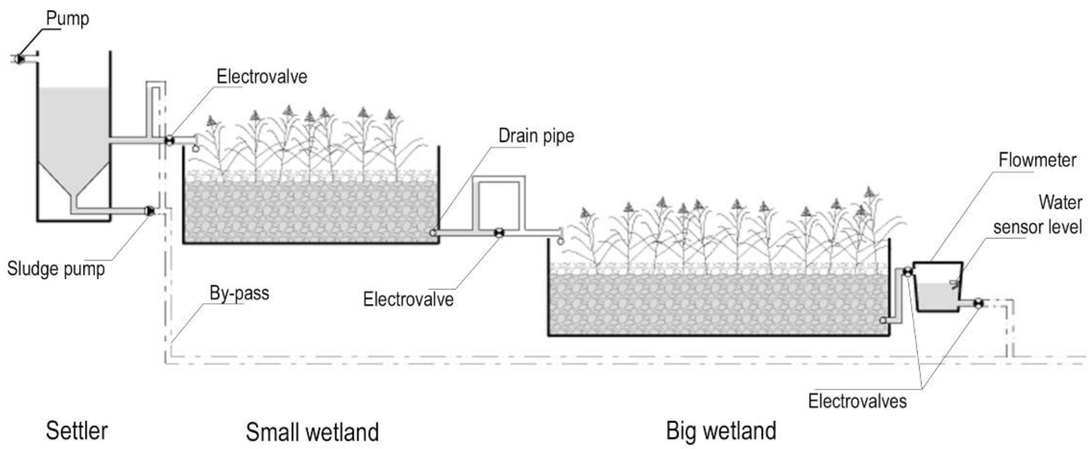

Figure 6.2 Schematic side view of the batch line. The set-up of wetlands in the other lines was similar except for the presence of the electrovalve between big and small wetlands. The electrovalve in the batch line enabled draining of the small wetlands.

Primary treatment

Batch and control lines have cylindrical PVC static settlers as primary treatment and are filled with screened wastewater pumped from the storage tank every 4 hours. The control line has three settlers, one for each small wetland (planted and unplanted), with an internal diameter of 190 mm and an effective volume of 7 L (Figure 6.3). The batch line has only two settlers because the small planted wetlands operate alternately. Settlers for the batch line have an internal diameter of 300 mm and an effective volume of 14 L (Figure 6.3). In the control line, after 2 hours of settling from each settler, 7 L of wastewater are discharged into the small wetlands (in total 14 L to the planted wetlands at each discharge). In the batch line, after 2 hours of settling, 14 L of wastewater are discharged into one small wetland. Irrespective of the treatment line (control or batch), wastewater discharges from settlers by means of electrovalves, while sludge wasting is realized by means of pumps that convey sludge back to the municipal sewer. The effect of differences in diameter and volume of the settlers between the control and the batch line was evaluated at the beginning of the study and there were no significant differences for TSS removal efficiency (results not shown).

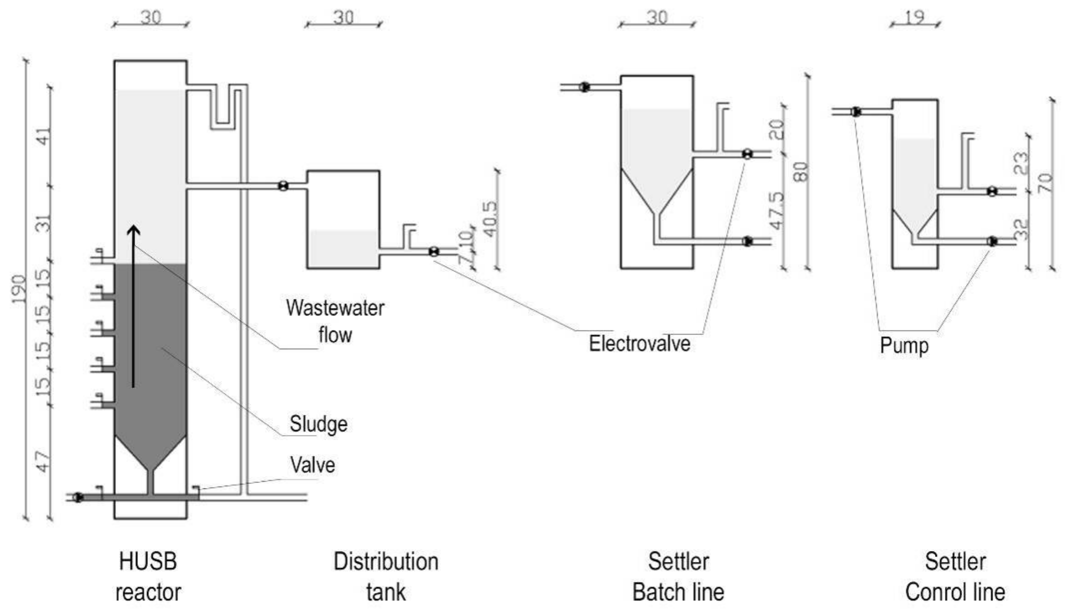

Figure 6.3 Schematic side view of the primary treatments. Distances expressed in cm.

The anaerobic line has a cylindrical PVC HUSB reactor as primary treatment, which has an internal diameter of 300 mm, a total height of 1900 mm and an effective volume of 105 L (Figure 6.3). The reactor is continuously fed with wastewater from the storage tank by means of a peristaltic pump that supplies a known flow. The HUSB reactor was operated at two different HRTs during the study period in order to assess optimum operational conditions. Specifically, the HUSB reactor was operated at 3 hours of HRT from February 2007 to December 2008 and at 5 hours of HRT from January 2009 to July 2009. The sludge blanket inside the HUSB reactor was kept as much as possible at a volatile solids (VS) concentration of < 10 g/L by means of manual purges. Solids concentration was taken from 5 taps located in a vertical series starting from a height of 480 mm and each placed at a respective distance of 150 mm (Figure 6.3).

Overall, VS concentration was estimated every two weeks measuring concentrations at each tap. The minimum and maximum solids retention time (SRT_{min} and SRT_{max}) were estimated according to the following equation (Álvarez et al. 2008b):

$$SRT_{max} = \frac{VX_R}{Q_W X_W} \tag{6.1}$$

$$SRT_{\min} = \frac{VX_R}{[(Q_W X_W) + (QX_e)]} \tag{6.2}$$

where V is the reactor volume (m^3), X_R is the weighted average biomass concentration in the reactor (kg VS/m^3), Q_w is the purged biomass flow rate (m^3/d), X_w is the concentration of purged biomass (kg VS/m^3), Q is the influent flow rate (m^3/d) and X_e is the effluent volatile solids concentration (kg VS/m^3).

Every 4 hours the content of the upper part of the HUSB reactor was discharged into three distribution tanks (one per each small wetland) with an internal diameter of 300 mm and an effective volume of 7 L. Without time for particle settling (tanks were charged and discharged in 10 min), wastewater was discharged from these tanks into the small wetlands by means of electrovalves. This set-up ensured a flow of 7 L to each small wetland every 4 hours.

Wetlands

Big and small wetlands consist of plastic containers 1.5 m long, 1.1 m wide and 0.50 m high, and 0.95 m long, 0.70 m wide and 0.45 m high, respectively. Wastewater from the primary treatment was discharged by means of perforated pipes located along the width of the wetlands. Each container had a drainage pipe on the flat bottom for effluent discharge. The uniform gravel layer (D_{60} = 7.3 mm, C_u = 0.83, 40% initial porosity) was 0.3 m deep and the water level was kept 0.05 m below the gravel surface to give a water depth of 0.25 m. In April 2007 the wetlands were planted with developed rhizomes of common reed (*Phragmites australis*) and by July 2007 the plants were well-established and covered the entire surface of the wetlands.

In anaerobic and control lines all the wetlands remained permanently saturated. The batch line, however, operated under a scheme of a four-day cycle and the small wetlands were not permanently saturated (Figure 6.4). Accordingly, for the first two days of the cycle the small wetlands were fed in the same way as the control line, on the third day the wetland rested under saturated conditions but received no influent and on the fourth day the wetland was drained (by means of an electrovalve, see Figure 6.2) and rested under unsaturated conditions.

Note that the two small wetlands of the batch line are at different phases at any time (Figure 6.4). Figure 6.5 shows an example of the effluent flow rate discharged by the three lines during a period of approximately one month. As can be seen, the batch line shows an important variation in the daily flow rate due to its filling-resting-draining phases. However, the accumulated water volume was approximately the same for all three experimental lines.

Effluent from each line was discharged into a tank (flow meter) with a water sensor level that was activated every 5 L of effluent. When this level was reached, the tank emptied. The total effluent volume was then calculated by multiplying the times that the sensor had been activated by 5 L. Flow measurements were recorded on a daily basis.

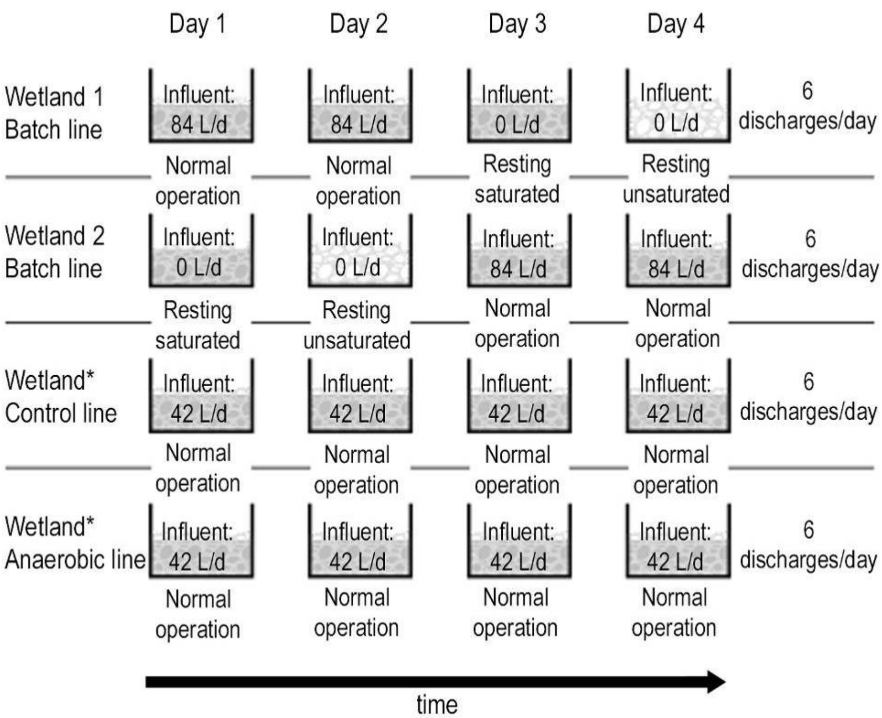

Figure 6.4 Schematic diagram of an operation cycle (4 days) of the batch line compared with the control and the anaerobic line (small wetlands). * The operation of one of the two small wetlands is shown for control and anaerobic lines only.

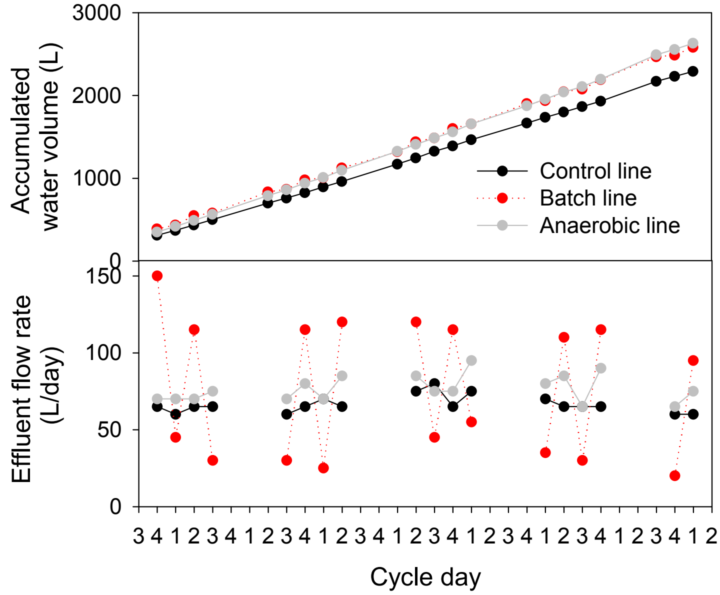

Figure 6.5 Changes in effluent flow rate (bottom graph) and accumulated water volume (upper graph) for the three treatment lines over a period of one month. Abscissas range from 1-4, showing the cycles of the batch line. Missing values correspond to weekends, when the flow rate was not monitored.

6.2.2 Samples and analyses

Samples of the influent (storage tank), primary treatments (settlers and HUSB reactor) and effluent (big wetlands) were taken 3 to 4 times per month from April 2007 to July 2009. From April to July 2007, samples were analysed weekly for pH, redox potential, turbidity, COD, TSS, ammonium and sulphates. In addition, from October 2007 to July 2009, BOD_5, dissolved COD, nitrites, nitrates, TKN and total phosphorus were also analysed approximately once a month. Redox potential values were corrected for the potential of the hydrogen electrode. Analyses were carried out following the methods described in APHA-AWWA-WPCF (2001). Differences between lines were statistically evaluated through the two-way (time and treatment) ANOVA (without replication) test using the software package SPSS 17.0.

Online sensors were used to monitor different points of the experimental plant. Redox potential probes (Digimed TH-404) were placed at the outlet of each treatment line (before the flow-meter) to measure redox potential every 10 seconds. A turbidimeter (Digimed TB-44M) and ammonium probe (Digimed AI-NH3) were used to measure a sample per day of one settler, the effluent of the HUSB reactor and the outlet of each line. From each measuring spot a peristaltic pump diverted the water to the equipment. Data from online sensors was collected and stored automatically in a datalogger DT50. These data can be downloaded via a Wi-Fi Internet connection. Only a small part of the data from the online sensors is shown in this paper. All electrical devices of the plant (pumps, electrovalves, etc.) were controlled by an OMRON ZEN® programmable relay, programmed with ZEN Support 4.0® software.

6.3 Results

6.3.1 Operation of HUSB reactor

Table 6.1 shows the main operational parameters of the reactor as a function of the HRT. From start-up to December 2008 (first period, with an HRT of 3 hours), the HUSB reactor worked with at flow rate of 0.42 L/min. This created an upflow velocity of 0.46 L/h. During the second period (5 hours HRT), the flow rate changed to 0.25 L/min, decreasing the velocity into the digester to 0.27 L/h. The excess sludge of the HUSB reactor was purged every two weeks during the period of operation with an HRT of 3 h. From this point, the reactor was purged daily for the purpose of improving control of the biomass concentration (< 10 g VS/L). The average biomass concentration in the sludge bed was slightly higher than 10 g VS/L during the period operated at an HRT of 3 h, while it was lower during the later period (5 hours of HRT). Maximum solids retention time (SRT_{max}) decreased by 30% during the period when HUSB was operated at 5 hours of HRT.

Table 6.1 Values of the main operational parameters of the HUSB reactor as a function of the HRT. Standard deviations in brackets. n = 10.

HRT(h)	Flow rate (L/min)	OLR[a] (kg COD/m^{3}d)	SLR[b] (kg TSS/m^{3}d)	X_R[c] (gVS/L)	SRT_{min} (d)	SRT_{max} (d)
3	0.42	0.45 (0.21)	0.25 (0.17)	12.25 (4.09)	3.64 (1.25)	24.46 (18.36)
5	0.25	0.23 (0.09)	0.18 (0.13)	7.78 (4.41)	3.40 (3.55)	16.46 (14.12)

[a] Organic loading rate
[b] Solids loading rate
[c] Biomass concentration in the reactor

6.3.2 Primary treatment efficiency

The HUSB reactor produced effluents with significantly lower redox potentials (P = 0.00) than the settler (Table 6.2). This result concurs with the significantly lower sulphate concentration (P = 0.00) at the effluent of the HUSB reactor, indicative of a greater sulphate reduction activity. TSS clearly decreased from raw wastewater to the effluents of the primary treatments, although significant differences between primary treatments were only observed during the second period (P = 0.02). Hydrolysis and solubilisation of particulate matter in the HUSB reactor was clearly favoured during the second period, when the dissolved COD was significantly higher than in raw wastewater and settler effluent (P = 0.04). BOD_5 experienced very low decrease throughout the primary treatment (either HUSB or settler), indicating that the particles removed in the primary treatment were not readily biodegradable.

Table 6.2 Average values and standard deviations (in brackets) of water quality parameters analysed in raw wastewater and effluents of the primary treatments. Data sorted in two periods according to the HRT of the HUSB reactor. First period from March 2007 to December 2008 and second from January 2009 to July 2009. n changes depending on the frequency of each parameter analysed.

Period			First				Second	
	n	Raw wastewater	Settler	HUSB (HRT=3h)	*n*	Raw wastewater	Settler	HUSB (HRT=5h)
pH	58	7.99 (0.22)	7.89 (0.25)	7.70 (0.23)	18	7.79 (0.33)	7.74 (0.29)	7.45 (0.26)
E_H (mV)	58	15.5 (103)	109 (148)	-102 (43.7)	19	180 (118)	172 (103)	-103 (100)
Turbidity (NTU)	57	151 (61.1)	104 (40.8)	102 (46.7)	18	103 (44.8)	93.3 (27.0)	74.0 (30.2)
TSS (mg/L)	52	223 (154)	101 (47.5)	95.1 (56.9)	21	161 (116)	98.8 (47.5)	62.1 (29.7)
COD (mg/L)	46	439 (243)	312 (169)	307 (157)	18	293 (94.1)	235 (52.01)	234 (50.02)
COD dissolved (mg/L)	6	193 (107)	189 (98.3)	184 (72.2)	4	146 (46.0)	136 (44.8)	176 (19.5)
BOD_5 (mg/L)	18	180 (50.2)	162 (50.7)	174 (52.1)	17	170 (59.7)	165 (49.1)	149 (50.5)
NH_4^+-N (mg/L)	55	26.4 (9.81)	25.1 (9.67)	27.8 (9.73)	21	25.2 (7.56)	20.7 (7.48)	26.4 (9.09)
$SO_4^=$-S (mg/L)	44	234 (53.6)	221 (55.9)	188 (52.9)	19	165 (55.3)	162 (63.0)	133 (60.4)

6.3.3 Treatment lines efficiency

During the study period the wetlands operated with a hydraulic loading rate of 28.5 mm/d. Average surface organic loads were 8.2±3.3 g COD/m^{2}d and 4.7±1.4 g BOD/m^{2}d for the control and batch lines, respectively (same primary treatment), and 8.8±3.7 g COD/m^{2}d and 4.7±1.5 g BOD/m^{2}d for the anaerobic line (with the HUSB reactor as primary treatment). Average surface ammonium load was approximately 1.1 g NH_4^+-N/m^{2}d, irrespective of the experimental line analysed.

Removal efficiencies were calculated in terms of mass removal as a consequence of considerable variations in evapotranspiration, which was estimated from the difference between influent and effluent flow rates (Figure 6.6). Evapotranspiration showed a typical trend, with higher values in summer (ranging from 25 to 30 mm/d) and lower in winter (ranging from 2 to 10 mm/d).

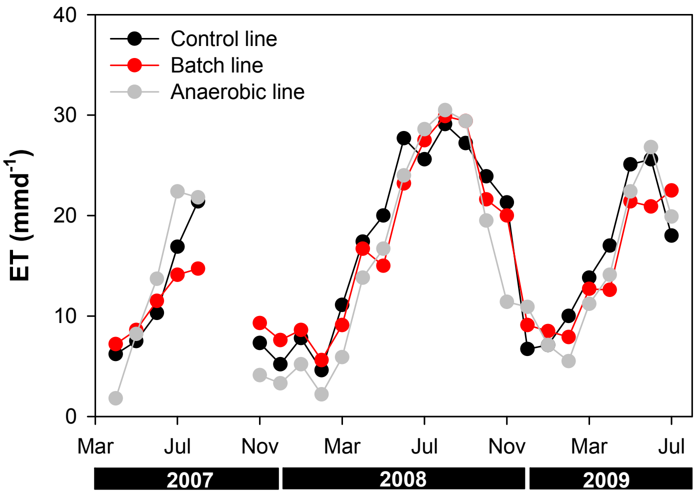

Figure 6.6 Temporal changes in evapotranspiration in each treatment line. Monthly averages are shown for the sake of clarity.

The average effluent redox potential of the anaerobic line was significantly lower than for the other lines (P = 0.00) (Table 6.3). However, there were no significant differences in the average redox potential between batch and control lines (P = 0.21). All treatment lines removed TSS above 90%, and the average effluent mass loads were not significantly different (P = 0.147).

Table 6.3 Average values and standard deviations (in brackets) of physical and chemical water quality parameters of the effluents of each line. Concentrations and mass loads are shown for TSS, COD, BOD_5, and $NH4^+$-N.

		n	Control	Batch	Anaerobic
pH		76	7.1 (0.4)	7.1 (0.4)	7.2 (0.4)
E_H (mV)		70	3 (92.7)	-5 (71)	-45 (78)
Turbidity (NTU)		75	7.3 (8.0)	7.0 (7.0)	8.6 (10.9)
TSS	(mg/L)	75	11.4 (8.44)	10.5 (9.75)	10.6 (10.4)
	(g/m^2d)		0.3 (0.7)	0.2 (0.3)	0.2 (0.2)
COD	(mg/L)	65	60.3 (76.04)	55.8 (86.06)	63.3 (62.8)
	(g/m^2d)		2.0 (2.9)	1.6 (2.4)	2.4 (2.6)
BOD_5	(mg/L)	35	20.3 (15.6)	15.0 (13.3)	35.4 (14.8)
	(g/m^2d)		0.3 (0.2)	0.2 (0.2)	0.6 (0.6)
NH_4^+-N	(mg/L)	76	6.07 (8.43)	4.11 (6.69)	7.36 (9.13)
	(g/m^2d)		0.15 (0.23)	0.10 (0.16)	0.20 (0.28)
TN	(mg/L)	10	7.45 (8.00)	4.84 (4.28)	5.58 (5.63)
NO_x^--N	(mg/L)	28	0.09 (0.11)	0.14 (0.21)	0.10 (0.13)
TP (PO_4^{3-}-P)	(mg/L)	23	2.53 (3.16)	1.87 (2.51)	3.47 (4.61)
SO_4^{2-}	(mg/L)	63	374 (393)	453 (473)	262 (301)

The batch line had slightly higher COD removal efficiency (88%) than control (P = 0.084) and anaerobic lines (83% and 80%, respectively). There was no significant difference between anaerobic and control lines (P = 0.270). COD effluent mass loads presented a seasonal pattern with higher values in winter than in summer (Figure 6.7). These higher values were less noticeable for the control and batch lines in winter 2008-09.

Batch and control lines had higher BOD_5 removal efficiencies (96% and 94%, respectively) than the anaerobic line (87%); in fact, effluent mass loadings were significantly higher (P = 0.00) for the anaerobic line than the other two lines. Note that effluents of the anaerobic line had a higher organic matter content (measured as COD or BOD_5) in accordance with the significantly lower redox potential values.

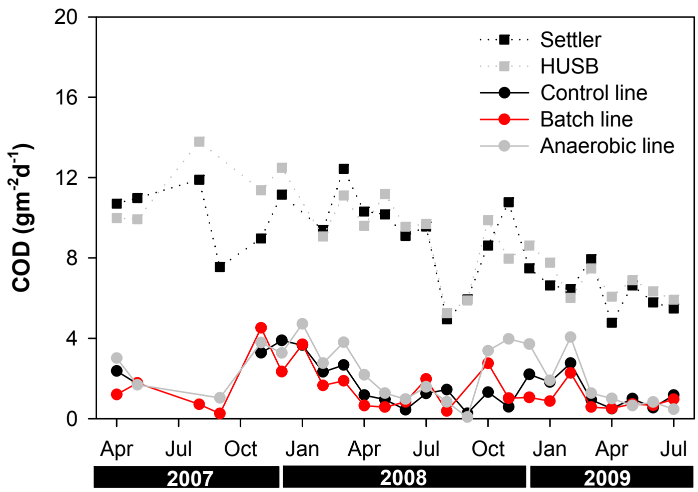

Figure 6.7 Temporal changes of the influent COD mass loads (settler and HUSB reactor) and the effluent mass loads of each treatment line. Monthly averages are shown for the sake of clarity.

The batch line had a higher ammonium mass removal efficiency (87%; P = 0.016) than the control line (80%) and the anaerobic line (73%). Lower ammonium removal efficiencies observed in the anaerobic line again concur with the significantly lower redox potential. Ammonium effluent mass loads also presented a seasonal pattern, which was even more noticeable than the pattern observed for COD (Figure 6.8). Note that the ammonium removal efficiencies ranged from almost zero in winter time to 100% during most of the summer. In fact, in summer the three lines were very efficient in ammonium removal. In addition, it is interesting to point out that the effluent ammonium mass was clearly lower in winter for the batch line (on average

±SD 0.24 g/m^2d ±0.18) than for the other two lines (0.46 ±0.13 and 0.53 ±0.20 g/m^2d for control and anaerobic lines, respectively), which leads to a higher overall removal efficiency observed during the entire study for the batch line.

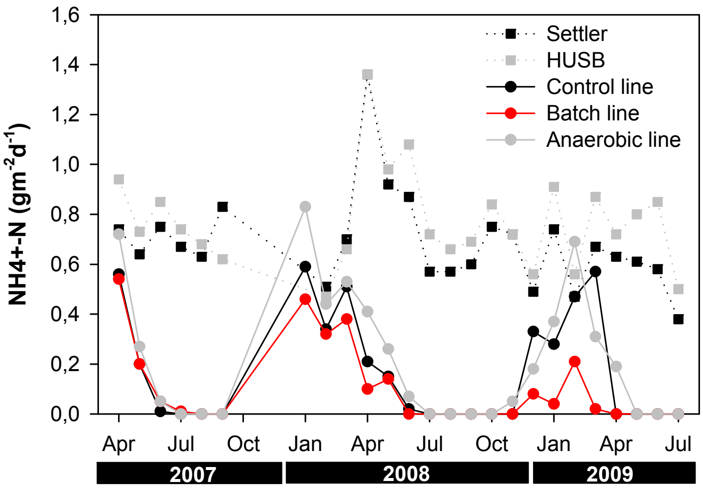

Figure 6.8 Temporal changes of the influent ammonium mass loads (settler and HUSB reactor) and the effluent mass loads of each treatment line. Monthly averages are shown for the sake of clarity.

Sulphate concentration increased from the influents to the effluents due to the overall driving evapotranspiration. Effluent concentrations were significantly different (P = 0.002) and lower in the anaerobic line (30% lower than the control line) in connection with the lower redox potential (Table 6.3). Sulphate concentrations were highest in the batch line, suggesting that this configuration provides a more overall oxygenated environment than the rest, even through the average redox potential was similar to the control line.

Comparable to the other parameters studied, total phosphorus removal efficiency was higher for the batch line (75%) than for the control (66%). And again, the anaerobic line presented the worst efficiencies (54%) (Table 6.3), although it was not statistically different (P = 0.273).

Table 6.4 Average values and standard deviations (in brackets) of physical and chemical water quality parameters of the effluents of each line. Concentrations and mass loads are shown for TSS, COD, BOD_5, and $NH4^+$-N.

		n	Control	Batch	Anaerobic
pH		76	7.1 (0.4)	7.1 (0.4)	7.2 (0.4)
E_H	(mV)	70	3 (92.7)	-5 (71)	-45 (78)
Turbidity	(NTU)	75	7.3 (8.0)	7.0 (7.0)	8.6 (10.9)
TSS	(mg/L)	75	11.4 (8.44)	10.5 (9.75)	10.6 (10.4)
	(g/m^2d)		0.3 (0.7)	0.2 (0.3)	0.2 (0.2)
COD	(mg/L)	65	60.3 (76.04)	55.8 (86.06)	63.3 (62.8)
	(g/m^2d)		2.0 (2.9)	1.6 (2.4)	2.4 (2.6)
BOD_5	(mg/L)	35	20.3 (15.6)	15.0 (13.3)	35.4 (14.8)
	(g/m^2d)		0.3 (0.2)	0.2 (0.2)	0.6 (0.6)
NH_4^+-N	(mg/L)	76	6.07 (8.43)	4.11 (6.69)	7.36 (9.13)
	(g/m^2d)		0.15 (0.23)	0.10 (0.16)	0.20 (0.28)
TN	(mg/L)	10	7.45 (8.00)	4.84 (4.28)	5.58 (5.63)
NO_x^--N	(mg/L)	28	0.09 (0.11)	0.14 (0.21)	0.10 (0.13)
TP (PO_4^{3-}-P)	(mg/L)	23	2.53 (3.16)	1.87 (2.51)	3.47 (4.61)
SO_4^{2-}	(mg/L)	63	374 (393)	453 (473)	262 (301)

6.4 Discussion

In this study an experimental plant was constructed in order to evaluate the effect of the type of primary treatment and operating strategy on the contaminant removal efficiency of shallow horizontal SSF CWs. The wetlands operated with the same wastewater flow rate (and therefore with the same hydraulic loading rate of 28.5 mm/d) and approximately the same organic loading rate. An overview of the general performance of the plant will be discussed in this section. The effect of primary treatment will be addressed by comparing the control and anaerobic lines, whereas the effect of operating strategy will be examined by comparing the control and batch lines.

6.4.1 Wetlands general performance

The average organic loading rate in the wetlands of the three lines (4.7±1.5 g BOD/m^2d) was slightly lower than the design load (6 g BOD/m^2d), but still in the range of 4 to 6 g BOD/m^2d generally recommended for horizontal subsurface flow constructed wetlands (Kadlec and Knight, 1996; García et al., 2005; Akratos and Tsihrintzis, 2007). Within this load range, 80% to 90% of COD removal efficiencies are commonly attained, and ammonium removal efficiencies usually range from 45% to 55% (Austin and Nivala, 2009; Vymazal and Kröpfelová, 2009). In the 3 lines of the experimental plant the overall organic matter (COD) removal efficiency was in the commonly attained range (80% for anaerobic, 83% for control and 88% for batch, respectively). Ammonium removal efficiency, however, was higher than the values usually described for this type of systems (73% for anaerobic, 80% for control and 87% for batch). This is due to two characteristics observed in several previous studies and considered in the design of the wetlands for this study: 1) low water depth (0.25 m) and 2) intermittent feeding (Caselles-Osorio and García, 2007a, b). Shallow wetlands have been demonstrated to be more efficient than wetlands with standard depth (0.5 to 0.7 m) because of their more oxidised conditions (García et al., 2003 and 2005). Moreover, intermittent feeding has also been shown to promote more oxidised conditions than continuous feeding (Caselles-Osorio and García, 2007a). Finally, these shallow wetlands have a good capacity to carry out whole nitrogen removal and the results of the present study support this statement (low effluent concentrations of ammonium as well as oxidised nitrogen, see Table 6.3).

A seasonal pattern for COD and ammonium was observed, with lower mass values in warm months (Figure 6.7). Seasonal variations in COD are not generally seen in these types of systems (García et al., 2005; Ruíz et al., 2008; Serrano et al., 2009;Vymazal, 2009), although seasonal differences have been largely demonstrated for ammonium (Caselles-Osorio and García, 2007a; Kuschk et al., 2003). In this study, when COD removal was analysed in terms of concentration, higher removal in summer was not clearly observed and consequently there was no clear seasonal pattern. This could be due to the effect of evapotranspiration, which was very high in these months (25-30 mm/d) (Figure 6.6). Accordingly, an increase of evapotranspiration leads to an increase in organic matter concentration, which could mask greater removal in summer.

6.4.2 Effect of primary treatment

Settlers only provide physical wastewater treatment. However, a HUSB reactor not only entraps particles but also promotes their bacterial hydrolysis (providing additional biological treatment). Thus, the lower redox potentials and pH observed in the effluents of the HUSB during the entire study period are a result of biological activity. Despite this biological activity, differences in contaminant removal between the two types of primary treatment were only clearly observed in the period when the HUSB was operated at an HRT of 5 days (efficiencies were higher in the HUSB reactor). Solids loading rate (SLR), upflow velocity, HRT and solids retention time (SRT) are among the most important operational parameters for the operation of HUSB reactors (Serrano et al., 2009; Álvarez et al., 2008b). Accordingly, higher removal efficiencies for TSS were recorded during the second operation period, when the HUSB operated at 5 hours HRT and the upflow velocity decreased from 0.46 to 0.27 L/h. Specifically, during this period TSS removal efficiency was improved (from 52% to 66%) and greater COD solubilisation rates were recorded (12% and 40% for 3 and 5 hours of HRT, respectively) (Table 6.2). HUSB reactor operating at 5 hours of HRT produced water with 30% fewer suspended solids and more biodegradable water than the settlers. Accordingly, in this period the BOD_5/COD ratio changed from 41% in raw wastewater to 64% and 53% in HUSB and settler effluents, respectively. Similar values were recorded by Barros et al. (2008) for UASB reactors.

The anaerobic line had high removal efficiencies for selected pollutants (on average 80%, 87% and 73% for COD, BOD_5 and ammonium, respectively), with results similar to those of previous studies (Singh et al., 2009; Ruíz et al.,2008), and effluent concentrations were lower than the standards set by Council Directive 91/271EEC. However, removal efficiencies for the anaerobic line were in all cases below those recorded for the control line (83%, 94% and 80% for COD, BOD_5 and ammonium removal efficiency, respectively; $P = 0.475$, $P = 0.000$ and $P = 0.007$ for COD, BOD_5 and ammonium, respectively). These lower removal efficiencies concurred with lower average redox potentials observed in the effluents of the anaerobic line than in the control line, which was indicative of a less oxygenated environment in the wetlands (see Table 6.3). The biological activity of the HUSB reactor in the anaerobic line produced more reduced primary effluents, which were eventually responsible for the less oxygenated environment of the wetlands in comparison to the wetlands of the control line.

In winter ammonium removal efficiency was lower in the anaerobic line than in the control line ($P = 0.000$), whereas this difference was minimized during warm seasons ($P = 0.165$) (see Figure 6.8). For the whole period of study there was a significant difference between the two treatments ($P = 0.007$) and, in relation to this, it is generally accepted that in warm seasons the active macrophytes enhance pollutant removal (Tanner et al., 1998; Brix, 1997). Accordingly, active plants of the wetlands in the anaerobic line may have improved the treatment efficiency of the system during these months.

The results of this study indicate that the HUSB reactor produces reduced effluents leading to a less oxidised environment in the wetlands and eventually to lower removal efficiency. Therefore, implementation of a HUSB reactor coupled with treatment wetlands within the conditions tested in this study does not benefit contaminant removal. However, the HUSB reactor has a higher degree of solids removal in comparison with a conventional settler and implementation could prevent early clogging of the wetlands (Shepherd et al., 2001).

6.4.3 Effect of operation strategy

In general, the batch line had slightly higher removal efficiencies (between 2% and 10%) than the control line for the main contaminants (Table 6.3), showing a statistical difference for ammonium ($P = 0.016$). Furthermore, these removal efficiencies of the batch line are similar to those generally described for vertical SSF CWs (Brix and Arias, 2005). Currently, the operation strategy of this line, alternating saturated and unsaturated phases, is comparable to the operation of vertical flow systems.

Higher removal efficiencies for the batch line seem to be due to the prevalence of a more oxygenated environment of the wetlands in this line. This hypothesis is confirmed by 32% less ammonium and 21% more sulphate concentrations in the effluents of the batch line than in the control line (Table 6.3). However, the results of average redox potential (without significant differences between the batch and the control lines, Table 6.3) contrast with the more oxygenated environment for the batch line stated above. Lack of differences in redox potential may have been an artefact, due to the operation of the batch line with cycles alternating batch-unsaturated and permanently saturated phases. As a result of this type of operation, high variations in the flow rate and redox potential for the batch line are encountered (see Figure 6.5 for flow rate variations and Figure 6.9 for redox potential variations). Thus, the time taken to measure redox potential in the effluents of the batch during the sampling seems to have greatly influenced the recorded value.

Figure 6.9 shows temporal changes in redox potential and ammonium concentration obtained from the online sensors in four representative weeks during the cold season (winter time, Figure 6.9a and 6.9c) and the warm season (spring time, Figure 6.9b and 6.9d) in control and the batch lines. As stated above, redox potential variations in the batch line are considerable in relation to the type of operation. Redox values were generally lower in February (average -141 mV and -1 mV for the control and the batch lines, respectively) than in May (149 mV and 189 mV for the control and the batch lines). This behaviour concurred with ammonia concentrations, which were lower in May (0.4 ± 0.3 mg N/L for both lines) than in February (26.9 ± 6.1 mg N/L and 10.7 ± 3.01 mg N/L for the control and the batch lines). Caselles-Osorio and García (2007a) found a similar temporal opposite relationship between redox potential and ammonium concentration. Differences between lines in redox potential were statistically significant in February ($P = 0.05$) but not in May ($P = 0.58$), and these trends match the fact that ammonium concentrations were similar (and low) in both lines in May, yet higher in the control line than in the batch line in February. It can therefore be concluded that differences between systems occur because the batch operation creates a more oxidised environment in cold months, which improves its performance during these months (approximately 50% higher removal efficiency) and is eventually responsible for improved overall annual removal efficiencies (approximately 10% higher removal efficiency).

Good ammonium removal efficiency of the batch and control lines in summer (without differences) may be due to the aeration effect of the plants, which may indeed have had more bearing on the oxidation-reduction status than the type of operation. The presence of macrophytes has been described as favouring oxidized environments in the wetlands, thus enhancing contaminant removal (Tanner et al. 1999; Tanner, 2001; Davies et al., 2006). In winter, when the reeds were dry and harvested, the aeration effect may have been negligible and the batch operation may have promoted a more oxidised environment, subsequently comparatively enhancing ammonium removal.

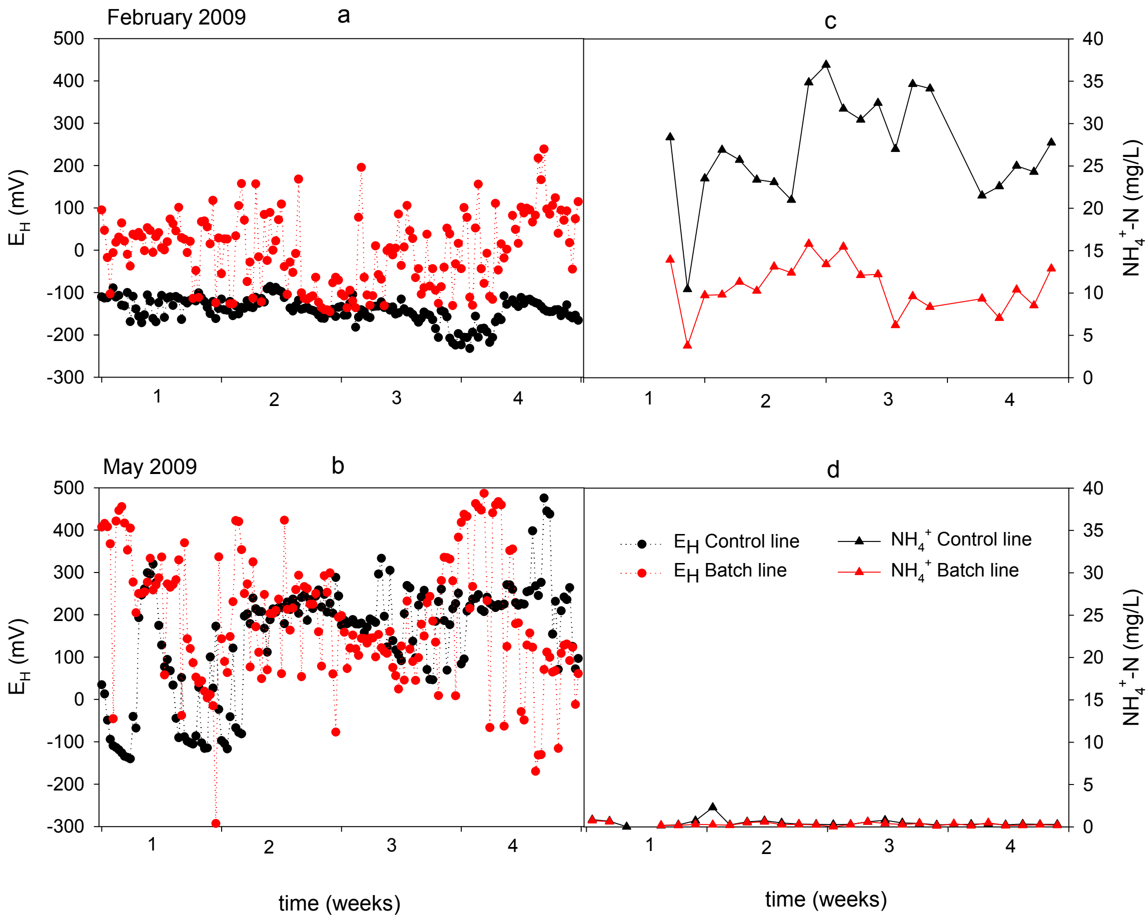

Figure 6.9 Temporal changes of redox potential and ammonium in control and batch lines during one month in winter (a, c) and another in spring (b, d).

The results of this study confirm that aeration of the granular medium of the wetlands occurring after draining of the batch line in small wetlands helps to increase the removal efficiency of organic matter and ammonium in treatment wetland systems (Tanner et al., 1999; Chazarenc et al., 2009). The difference in our study in comparison with previous reports is that this increased efficiency is only observed in winter months. This trend seems to be related to the low depth of the wetlands tested in this study.

6.5 Conclusions

This study demonstrates that shallow horizontal SSF CWs are good systems for wastewater treatment and can remove contaminants at high rates (around 80% for COD and ammonium, and 90% for TSS and BOD_5) because of shallow beds and intermittent feeding.

The implementation of a HUSB reactor as a primary treatment for horizontal subsurface-flow constructed wetlands does not improve contaminant removal efficiency in comparison with a conventional settler in the conditions tested here. On the contrary, the HUSB reactor slightly decreases the efficiency of the system due to the reduced nature of its effluents. Moreover, when properly operated, the application of a HUSB reactor can significantly reduce the amount of solids entering a wetland (30% fewer suspended solids than conventional settlers) and therefore help prevent or delay clogging processes.

A treatment wetland system composed of horizontal subsurface flow wetland operated with filling-resting-drain phases (as in the batch line) offers higher efficiencies in contaminant removal than a normally operated system (as in the control line) (between 5% and 10% more efficient in COD and ammonium removal, respectively). However, differences in contaminant removal (especially in terms of ammonium) are season-dependent. In this regard, differences in ammonium removal between a system operated under batch conditions and a system operated continuously are maximized in winter (up to 50% higher than a continuously fed system).

The results obtained in this study should be given due consideration in the design and operation of horizontal SSF CWs. After a long experimentation period, we have provided sufficient data and clearly demonstrated that the efficiency of shallow horizontal SSF CWs can be increased, particularly in cold months, by using a batch operation strategy.

7

Clogging development in experimental subsurface flow constructed wetlands assessed by different indicators[5]

This study aims at evaluating the effect of two types of primary treatment (hydrolitic upflow sludge blanket (HUSB) reactor and conventional settling) as well as two flow regimes (batch and continuous) on clogging development in subsurface flow constructed wetlands (SSF CWs). Clogging was assessed during three years by means of different clogging indicators in an experimental plant with 3 treatment lines. Results show that hydraulic parameters are suitable indirect methodologies for clogging assessment since accumulated solids (including sludge and roots) presented high correlations with the saturated hydraulic conductivity and drainable porosity reduction over time (74.5% and 89.2% respectively). Although no significant differences between treatment lines were found in regards to hydraulic conductivity or porosity reduction after three years of operation SSF CW implemented with a HUSB reactor as primary treatment accumulates ca. 30% lower sludge (1.9 kg DM/m2) than a planted system implemented with a conventional settler (2.5 - 2.8 kg DM/m2). The presence of a well developed root system may notably contribute to the reduction of both hydraulic conductivity and porosity in SSF CW. Accordingly, planted wetlands show between 30-40% and 10% lower hydraulic conductivity and porosity reduction, respectively, than a non-planted wetland.

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[5] This chapter is based on the article:

Pedescoll, A., Corzo, A., Álarez, E., Puigagut, J., García, J. The effect of primary treatment and flow regime on clogging development in horizontal subsurface flow constructed wetlands: an experimental evaluation. Bioresource Technology, submitted.

7.1 Introduction

Subsurface flow constructed wetlands (SSF CWs) are extensive systems widely used for wastewater treatment in small communities (Rousseau et al., 2005; Puigagut et al., 2007; Vymazal and Kröpfelová, 2009). Low energy requirements and non specialized manpower for plant management are among the most important advantages of SSF CWs in comparison to conventional alternatives such as the activated sludge process (Wallace and Knight, 2006). There is a broad consensus in the fact that clogging of the wetlands as the worst operational problem of such technology (Cooper et al., 2005; Wallace and Knight, 2006; knowles et al., 2010b). Clogging are complex phenomena that involves biological, chemical and physical processes. Accordingly, retention of inorganic and organic influent particles, biofilm and plant biomass development and decay, and deposition and accumulation of chemical precipitates are among the most important factors promoting a progressive obstruction of the filter media (Knowles et al., 2010b). Clogging limits the lifespan of the systems (Caselles-Osorio et al; 2007) and can have negative impacts on treatment efficiency (Platzer and Mauch, 1997; Rousseau et al., 2005). Because of the drawbacks that clogging may arise on SSF CWs (both in treatment and management costs terms) there is great interest in studies aimed at assessing, understanding and preventing development of clogging processes (Tanner et al., 1998; Muñoz et al., 2006; Suliman et al., 2006a; Caselles-Osorio et al., 2007; Knowles et al., 2010a).

The quantification of accumulated solids in SSF CWs constitutes a direct measure of clogging, since solids clog the pore space of the filter media (Tanner et al., 1998; Caselles-Osorio et al., 2007). However, this clogging indicator demands an extremely exhaustive sampling effort due to the highly heterogeneity of the systems in both accumulation and solids nature terms (Tanner and Sukias, 1995; Caselles-Osorio et al., 2007; Llorens et al., 2009). Therefore, indirect clogging indicators have been also widely used, such as tracer tests hydrodynamics (Muñoz et al., 2006) and hydraulic conductivity measurements (Suliman et al., 2006a; Knowles et al., 2010a). Furthermore, correlations between clogging indicators has scarcely been investigated in current literature and generally provides confuse results (Tanner and Sukias, 1998; Caselles-Osorio et al., 2007).

Moreover, since loading rates have a significant effect on clogging development (Tanner and Sukias, 1995), implementation of improved primary treatments is essential to decrease the solids loading rate (Tchobanoglous, 2003). Conventional primary treatments such as Imhoff or septic tanks commonly coupled to SSF CWs usually have suspended solids removal efficiencies ranging from 50 to 70% (Tchobanoglous, 2003; Brix and Arias, 2005; Puigagut et al., 2007). The use of other types of primary treatments in the context of wetland technology (such as low rate anaerobic digesters), though to provide promising results, has been scarcely investigated (Álvarez et al. 2008a; Barros et al., 2008). Hydrolytic up-flow sludge blanket reactors (HUSB) are anaerobic reactors where wastewater suspended solids are trapped within a sludge blanket. In HUSB reactors methanogenesis is not reached due to a low hydraulic retention time (from 2 to 5 hours), and trapped solids undergo hydrolysis and acid fermentation (Álvarez et al., 2008b; Ligero et al., 2001). On the other hand, biomass retention time in HUSB reactors is maintained high (usually more than 15 days) in order to allow a continuous growth of acid fermenting bacteria (Álvarez et al. 2008). When a HUSB reactor is implemented as primary treatment for SSF CWs, solubilisation of trapped solids could help to delay the gradual clogging of the granular medium.

In addition to the use of improved primary treatments, suitable operation strategies coupled with improved design criteria might be of use to avoid rapid clogging (Nguyen, 2000; Langergraber et al., 2003; Zhao et al., 2006). Traditionally, horizontal SSF CWs remain water saturated and thus physical oxygen transfer rates from the air to the bulk water are low (< 1 g $O2/m^2d$) (Tyroller et al., 2010). Although macrophytes actively transport oxygen from the atmosphere to the bulk water, direct measurements of the amount which is actually released by plant roots has been described to be very low (i.e. 0.001 to 0.004 $gO2/m^2d$ (Bezbaruah and Zhang, 2005)). Therefore, oxygen availability in horizontal SSF CWs (operating at a normal organic loading of 4-6 BOD/m^2d (Faulwetter et al., 2009)) might by a limiting factor for the removal or degradation of solids that gradually accumulate in the granular medium. Discontinuous or batch feeding strategies have been described to enhance oxygen transfer to wetlands, and therefore it is supposed that these strategies promote the aerobic degradation of accumulated solids (Chazarenc et al., 2009).

The main aim of this study is to comparatively evaluate the effect of two types of primary treatment (HUSB reactor and conventional settling) as well as two flow regimes (batch and continuous) on clogging development in SSF CWs. To this end, several commonly used indicators to assess clogging in SSF CWs, such as solids accumulation, drainable porosity, saturated hydraulic conductivity and effective volume have been measured and contrasted. The tested hypothesis is that of the use of a HUSB reactor as primary treatment and alternating batch-unsaturated and permanently saturated phases (batch feeding mode) could delay the gradual clogging process of SSF CWs. Results here presented belong to the first 3 years of operation of an experimental plant specially designed to study the clogging phenomena in SSF CWs. Although clogging is a long term phenomena, studies devoted to clogging assessment have described high amounts of organic matter accumulated during the first 1-2 years of operation (Tanner and Sukias, 1995). Moreover, evidences of systems in advance state of clogging have been also described in literature after just 4 years of operation (Caselles-Osorio et al., 2007). Therefore, authors consider that the time scale devoted to clogging assessment presented in this study (three years) is long enough to address the stated research objectives.

7.2 Methods

7.2.1 Experimental plant

The experimental plant used in this study is emplaced at the experimental facilities of the Department of Hydraulic, Maritime and Environmental Engineering of the Technical University of Catalonia (Barcelona, Spain) (Figure 7.1). The plant was set in operation in February 2007 and treats a small fraction of the urban wastewater generated in the neighbourhood for experimental purposes (wastewater is pumped directly from a near municipal sewer pipe). Pre-treatment consists of coarse screening. After pre-treatment the wastewater is stored in a plastic reservoir of 1.2 m^3, which is continuously stirred in order to avoid solids sedimentation. From the storage tank the wastewater is conveyed to 3 different treatment lines, which for reasons of comprehension have been named under batch, control and anaerobic lines. Control line consists of a conventional settler as primary treatment followed by a HSSF CW system permanently saturated and intermittently fed. Anaerobic line differs from control line in the primary treatment (HUSB reactor) and batch line differs from control line in the operation strategy (with cycles of saturated and unsaturated phases in the first two wetlands of the system).

Secondary treatment for the three experimental line consists of two small wetlands in parallel (0.65 m^2 each) and one bigger in series (1.65 m^2) planted with common reed (*Phragmites australis*). Each line also includes one unplanted small wetland which effluent is directly discharged back to the municipal sewer (Figure 7.1). Clogging assessment will be addressed by comparing planted and unplanted small wetlands of each treatment without considering the bigger wetlands downstream. The two small wetlands have a joint surface area (1.3 m^2), which is approximately 45% of the total surface area of the treatment line (2.95 m^2). Note that the appearance of clogging in this type of wetlands is more evident in the inlet zone of the wetlands (Caselles-Osorio et al., 2007), and this is why the total surface of the wetland area was split in two (one big and two small wetlands). The two small parallel wetlands were necessary for the operation of the batch line. This system was also adopted in the other two lines for comparative purposes.

Small wetlands (both planted and unplanted) were operated at the same hydraulic loading rate (64.5 mm/d) considering a four-day cycle for the batch line (the HLR for each treatment line, considering the planted small and big wetlands was 28.5 mm/d). Wetlands were planted in April 2007 with developed rhizomes of common reeds and by July 2007 plants were well established and covered the entire surface of the wetlands. The uniform gravel layer (D_{60} = 7.3 mm, Cu= 0.83, 40% initial porosity) was 0.3 m deep and the water level was kept 0.05 m below the gravel surface to give a water depth of 0.25 m (shallow horizontal SSF CWs).

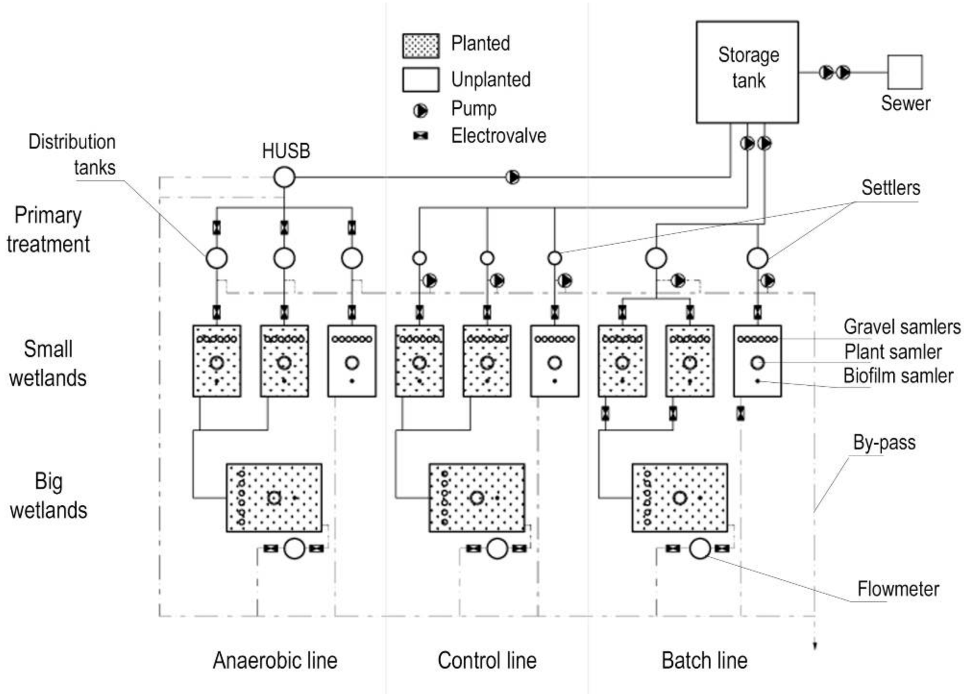

Figure 7.1 Schematic diagram of the experimental plant.

Small wetlands of the anaerobic and control lines were fed 6 times a day and remained permanently saturated. On the contrary, small wetlands of the batch line operated alternating batch-unsaturated and permanently saturated phases following a four-days-cycle:

- 2 days of filling and discharging to the big wetland or to the sewer according whether it is a planted or an unplanted system. Effluent discharge is carried out when the water in the wetlands reaches a level of 25 cm. During this phase the wetlands remain mostly saturated.
- 1 day saturated and resting. During this phase small wetlands do not receive influent for one day. At the end of this phase the wetlands are completely emptied by means of an electro valve).
- 1 day unsaturated and resting. During this phase wetlands do not receive wastewater.

Every 4 days each small wetland of the batch line has received the same amount of water than the small wetlands of the anaerobic or the control line but concentrated within the first 2 days of the 4-day-cycle (Figure 7.2).

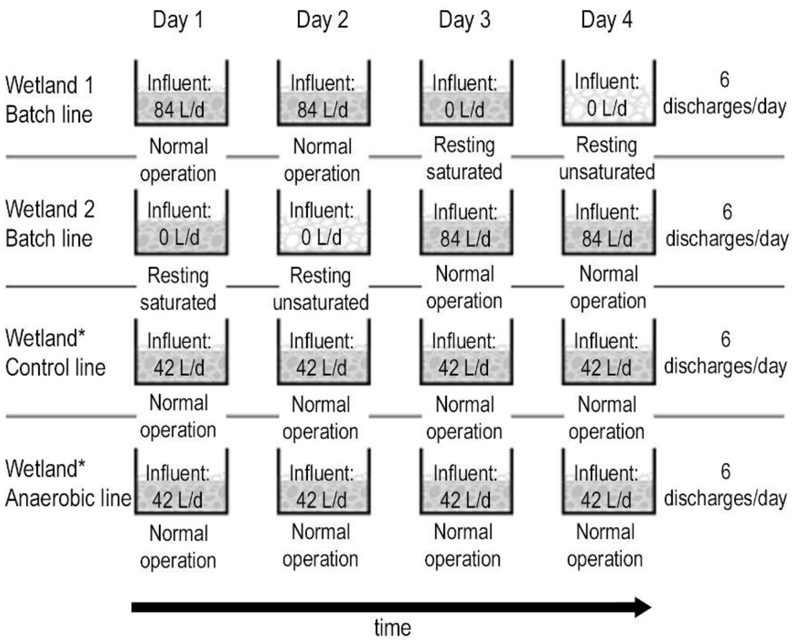

Figure 7.2 Schematic diagram of an operation cycle (4 days) of the batch line compared to the control and the anaerobic line (small wetlands). * For control and anaerobic lines only the operation of one of the two small wetlands is shown.

Results of water quality discussed in this paper consist of a three-year-data set split in two experimental periods according to the operational conditions of the HUSB reactor. Accordingly, the HUSB reactor was operated at 3 hours of hydraulic retention time (HRT) during the first experimental period (from February 2007 to December 2008) whereas during the second period it was operated at 5 hours of HRT (from January 2009 to December 2009).

7.2.2 Samples and analyses

Clogging factors were evaluated during the entire study period (3 years). A total of six sampling campaigns (with a duration ranging from 1 to 3 months each) were carried out in order to assess clogging development at the experimental SSF CWs. Table 1 summarizes the factors evaluated at each sampling campaign.

Table 7.1 Sampling campaigns and analyses

Parameters	Campaign*	Small wetlands sample points
Solids accumulation	2, 3, 4, 6	Gravel samplers
Drainable porosity	3, 4, 5, 6	The whole wetland in depth fractions of 5 cm
Saturated hydraulic conductivity	1, 2, 3, 4, 6	2 points at inlet and outlet
Tracer test (effective volume)	6	Outlet

Campaign 1: February 2007 (month 0)
Campaign 2: September 2007 (month 7)
Campaign 3: May – July 2008 (month 15)
Campaign 4: October – December 2008 (month 20)
Campaign 5: August – September 2009 (month 30)
Campaign 6: December 2009 – January 2010 (month 34)

Solids accumulation

In each wetland there were three different types of samplers (gravel, plant and microbial samplers). However, for the purposes of this study only gravel samplers will be considered (Figure 7.3). Gravel samplers consist of two tubes fitted one inside the other, with a diameter of 7 and 8.5 cm, respectively. Tubes are made out of coiled galvanized stainless steel mesh and were inserted into the granular medium to the bottom during construction of the wetlands. The mesh has a sieve size of 5 mm and therefore is able to contain the gravel. The inner tube was filled with gravel and left in the wetland until it was taken out for sampling. Each wetland has six gravel samplers located along the whole width of the wetlands and near the inlet (approximately 15 cm from the water distributor) (Figure 7.3).

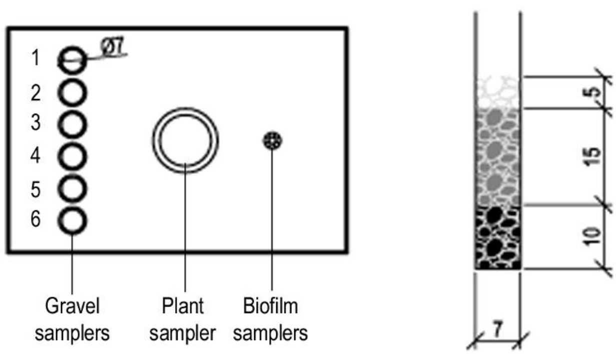

Figure 7.3 Schematic plant view of the wetlands and location of the gravel, root and biofilm samplers, and schematic diagram of the gravel samplers and the different sections (right). Note that plant and biofilm samplers were not used for the purposes of the present study.

From one to two gravel samplers were taken out at each sampling campaign. Samplers were taken out very carefully after draining completely the wetlands. Gravel samplers were immediately replaced by new ones in order to avoid any disturbance of the filter media (note that only the original samplers were analysed for solids accumulation).

Samplers were immediately transported to the laboratory and cut in 3 sections for accumulated solids analyses: 0.05 m above water level (which was not analyzed because it corresponds to the non-wetted media), top (0.15 m) and bottom (0,10 m) (Figure 7.3). Afterwards, gravel of the considered sections were taken out and processed. First it was removed by hand all recognisable alive and dead roots. Gravel and interstitial solids (sludge) were separated by means of washing (hand shaking) with 1 L of distilled water and water resulting from the cleaning process was further filtered through a 1 mm metal mesh according to the procedure described by Nguyen (2000). In this study solids strongly adhered to the gravel have not been considered because previous studies indicated that they represent < 1% of the total sludge in SSF CWs (Caselles-Osorio et al., 2007). Water used for washing containing the sludge was analyzed for total solids (TS) and volatile solids (VS) according to APHA- AWWA-WPCF (2001), obtaining the amount of sludge in terms of dry matter. Plant belowground biomass separated during manual screening was dried at 105ºC for 24 hours and weighted. Note that only roots of 1-5 mm of diameter are considered, since the sieve to wash the gravel had 1 mm pore space and sampler mesh had 5 mm.

Drainable porosity

Drainable porosity represents the pore volume that freely drains in a gravitational field at atmospheric pressure, thus representing the pore volume readily available for wastewater flow. In order to measure the drainable porosity the effluent pipe was lowered at intervals of 0.05 m and the water volume discharged was measured (Rowe et al, 2000). Water volume measured at each interval was compared with the total initial volume of the wetland (13.05 L considering an initial porosity of 40%).

Saturated hydraulic conductivity

A modified version of the *onsite* application of the falling head method (FHM) (Chapters 3, 4 and 5) was used for the measurement of saturated hydraulic conductivity of wetlands' filter medium. FHM consist of measuring the time a column of water that takes to leave a permeameter cell (a tub inserted into the gravel medium). The modification of the FHM described in Chapters 3 - 5 consisted of providing the water column to the permeameter cell by a balloon. Accordingly, the balloon was filled with water until it exploded inside the permeameter cell. This modified procedure was applied in order to avoid any flooding of the wetland due to its high initial conductivity (484 ±142 m/d).

For each wetland two points at the inlet and two at the outlet were considered for hydraulic conductivity measurements. Hydraulic conductivity was measured before the start up of the experimental plant, (initial hydraulic conductivity of the wetlands), and approximately every 6 months at the same locations.

Effective volume

In order to calculate the effective volume after three years of operation of the experimental plant a tracer test (with potassium bromide) was conducted. The tracer test was conducted using the step procedure (Levenspiel, 1999). Accordingly, the tracer compound was continuously supplied (coinciding with influent water discharges) and samples of wastewater at the outlet were taken every 4 hours (coinciding with discharges time) and analysed for bromide concentration by ionic chromatography (Dionex ICS-1000) until bromide concentration was stable (between 80 and 132 hours). With the obtained sigmoidal curve the actual mean hydraulic retention time (HRT) was obtained by calculating the area between the curve and the axis of the normalized bromide concentration (dividing each concentration by the concentration at the asymptote). The mean effective volume was then calculated for each wetland as the HRT times the flow rate.

Deviation from plug flow behaviour (D/uL) was calculated by plotting the experimental data on a probability graph paper:

$$\frac{D}{uL} = \frac{1}{2}\left(\frac{t_{84} - t_{16}}{2\tau}\right)^2 \tag{7.1}$$

Where,

D/u L is the longitudinal dispersion, dimensionless

t_{84} and t_{16} are the time at 84 and 16 percentile of the normalized bromide concentration plotted on a probability graph paper, in hours.

τ is the mean HRT, in hours

It must be noted that in order to obtain comparable values of the tracer tests between experimental lines, the small wetlands of the batch line were fed at the same regime than the control line (7L every 4 hours) and without cycles of drainage.

Physical-chemical analyses

For the evaluation of the efficiency of each of three lines, water samples of the influent (storage tank), primary treatments (settlers and HUSB reactor) and effluent (big wetlands) were taken 3 to 4 times per month from April 2007 to July 2009. From April to July 2007, samples were analysed weekly for pH, redox potential, turbidity, COD, TSS, ammonium and sulphates. In addition, from October 2007 to July 2009, BOD_5, dissolved COD, nitrites, nitrates, TKN and total phosphorus were also analysed approximately once a month. Redox potential values were corrected for the potential of the hydrogen electrode. Analyses were carried out following the methods described in APHA-AWWA-WPCF (2001). Note that the effectiveness of each line was assessed on mass balance basis, because influent and effluent flows were known. It must be mentioned that detailed discussion of water quality parameters at the effluent of each treatment line is out of the scope of the present paper. Accordingly, although for the sake of discussion comprehension authors will give some details on plant performance (note that plant performance is evaluated at the end of the whole treatment line not at the effluent of the small wetlands), the detailed influence of primary treatment and hydraulic regime on pollutants removal is discussed elsewhere (Chapter 6).

Statistical analysis

Differences between lines, planted and unplanted wetlands and temporal changes were assessed with two factors ANOVA test without replication. Pearson correlations between solids accumulation and porosity and hydraulic conductivity were conducted in order to find relations between clogging indicators. Statistical analyses were conducted using the SPSS 17.0 software package. Differences were considered significant at $p<0.05$.

7.3 Results

7.3.1 Influent characterisation and general efectiveness

The HUSB reactor produced effluents with significantly lower redox potential than the settlers (172 ± 103 and -103 ± 100 mV for settlers and HUSB reactor for the second period (HUSB operated at 5 hours of HRT), respectively). TSS clearly decreased from raw wastewater to the effluents of the primary treatments, although significant differences between primary treatments were only observed during the second period (38 and 60% of TSS removal efficiency for settlers and HUSB reactor, respectively). Hydrolysis and solubilisation of particulate matter in the HUSB reactor was clearly favoured during the second period when dissolved COD was significantly higher (dissolved COD/total COD ratio was 75%) than in raw wastewater or settler effluent (50 and 58%, respectively).

Average surface organic loads during the whole period were 8.2±3.3 g COD/m^2d and 4.7±1.4 g BOD/m^2d for the control and batch lines, respectively (same primary treatment), and 8.8±3.7 g COD/m^2d and 4.7±1.5 g BOD/m^2d for the anaerobic line (with the HUSB reactor as primary treatment). Average surface solids load was 2.88±1.35 g TSS/m^2d for the control and batch lines and 2.69±1.61 g TSS/m^2d for anaerobic line during the first period (HUSB reactor operated at 3 hours of HRT). During the second experimental period (HUSB operated at 5 hours HRT) surface solids load was 2.73±1.38 g TSS/m^2d for control and batch lines and 1.77±0.85 g TSS/m^2d for the anaerobic line.

The three treatment lines presented good global removal efficiencies (estimated in mass removal), around 80% for COD and ammonium and 90% for TSS and BOD_5. For COD and ammonium a seasonal pattern was found, with significant higher removal efficiencies in warm periods. Differences between batch and control line were found, mainly in cold periods. Accordingly, the batch line presented 50% higher removal of ammonium than the control line in winter time. Anaerobic line presented the lowest removal efficiencies, mainly due to the reduced state of HUSB's effluent. A detailed analysis of wetlands performances is out of the scope of the present study and it is accurately described in Chapter 6.

7.3.2 Accumulated solids

Accumulated sludge (interstitial solids of the gravel) and belowground plant biomass experienced and important increase through the experimental period, though plant biomass accumulation was especially important during the last year of operation, regardless the experimental line (Figure 7.4B and 7.4C). Roots accumulation occurred mainly in the last 10 months at the bottom of the wetlands, while roots in the upper layer (the top 15 cm of wetted gravel) remained essentially constant through the experimental period (results not shown). Overall, temporal changes on total accumulated solids (considering both sludge and roots) showed a progressive increase over time for the 3 treatment lines (Figure 7.4C).

After three years of operation, unplanted wetlands of the anaerobic and the batch line accumulated significantly less sludge (ca. 2 kg DM/m^2) than unplanted wetlands of the control line (ca. 5 kg DM/m^2) (Figure 7.4A). For planted wetlands accumulated sludge was of similar extent for the anaerobic and control lines (ca. 2 and ca. 2.5 kg DM/m^2, respectively), whereas the batch line accumulated slightly higher sludge with ca. 4 kg DM/m^2) (Figure 7.4A). Furthermore, accumulated sludge was slightly higher at the bottom of the wetlands (representing from 47% to

67% of the total), regardless the experimental line (Figure 7.5). Moreover, volatile solids represented, in all cases, less than 50% of total solids without differences between treatment lines, presence of plants or depth. However, the percentage of VS increased from the first to last sampling campaign in 12±8% (data not shown).

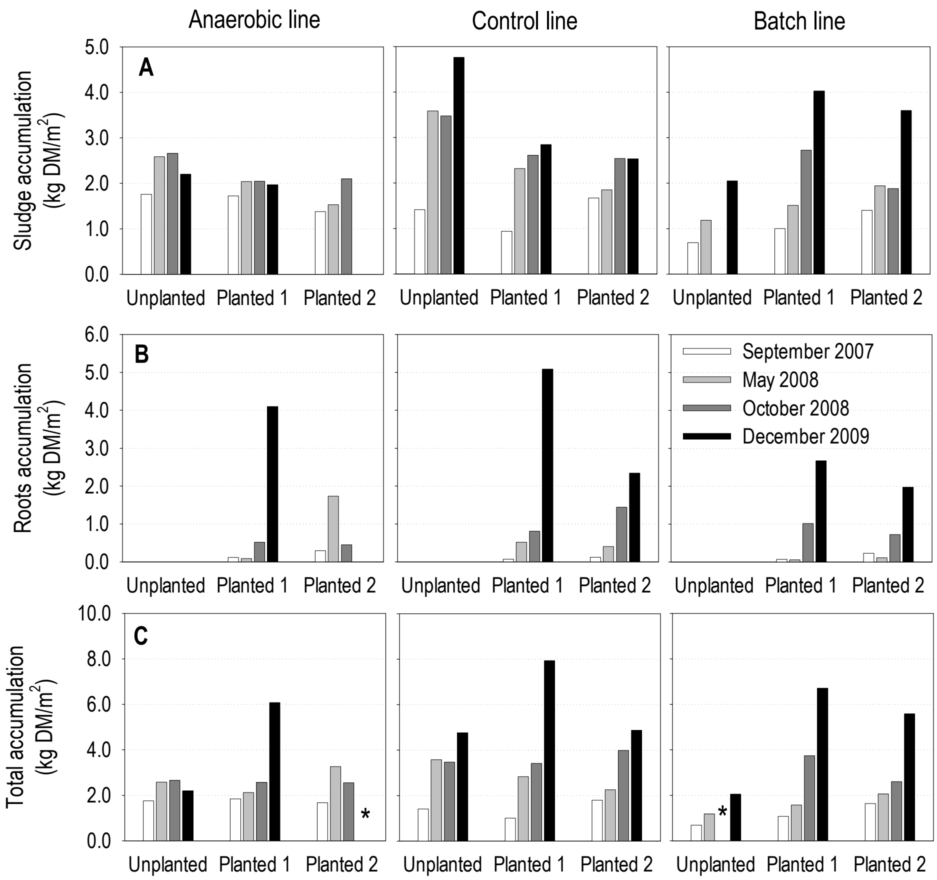

Figure 7.4 Temporal changes on accumulated solids; a) sludge, b) roots and c) its addition, in the small wetlands of the experimental plant. * Data not available due to the lack of sample.

Roots accumulation was significantly higher at the bottom of the wetlands, regardless the experimental line (Figure 7.5). Accordingly, roots at the bottom of the wetland (last 10 cm) represented between the 80% and the 90% of the total roots weight. It is important to point out that, even though only roots from 1 to 5 mm are considered, belowground biomass represented a great extent of the total solids accumulated within the wetland. More precisely, root biomass represented the 35% of the total solids of the batch line and around 50-70% of the accumulated solids in control and anaerobic lines (Figure 7.5).

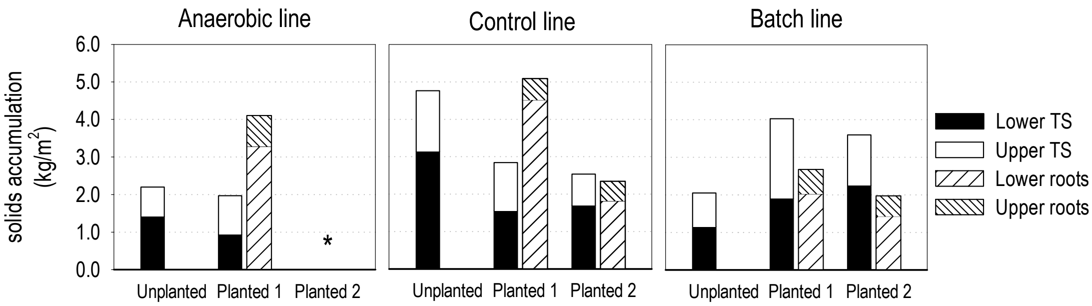

Figure 7.5 Sludge and root accumulation after three years. * Data not available due to the lack of sample.

7.3.3 Drainable porosity

A progressive reduction of porosity was observed over time (Figure 7.6A) in all the wetlands but with no significant differences between experimental lines. Note that Figure 7.6A shows the percentage of porosity reduction considering an initial drainable porosity of 40% (this corresponds to the 100% in the figure). After 3 years of operation wetlands filter media presented a porosity reduction ranging from 15 to 20% for unplanted systems and ca. 30% for planted wetlands, regardless the type of primary treatment or flow regime. Therefore, reduction of drainable porosity was significantly more important in planted wetlands (between 10-15% higher porosity than unplanted wetlands – Figure 7.6A).

7.3.4 Saturated hydraulic conductivity

Initial hydraulic conductivity was that of 484±142 m/d for all wetlands. Wetlands filter medium experienced an important hydraulic conductivity decrease (especially marked during the first 2 years of operation). More precisely, hydraulic conductivity in planted wetlands after three years of operation was 44±14 m/d, 72±34 m/d and 43±19 m/d in the inlet zone, and 122±55 m/d 216±65 m/d and 98±23 m/d in the inlet zone of control, batch and anaerobic lines, respectively. After three years of operation, although hydraulic conductivity final values in the outlet zone were higher than in the inlet (regardless the presence of plants or the treatment line), significant differences were only registered for the batch line (Figure 7.6C). Moreover, hydraulic conductivity in the inlet decreased up in a 60% and a 90% for unplanted and planted wetlands, respectively, regardless type of primary treatment or flow regime. At the outlet, values decreased in a 30% and between 60-80% for unplanted and planted beds, respectively. Therefore, hydraulic conductivity was significantly lower for planted than for unplanted wetlands (ca. 30 and ca. 50% lower in the inlet and outlet zones of the systems, respectively), regardless the type of primary treatment or hydraulic regime.

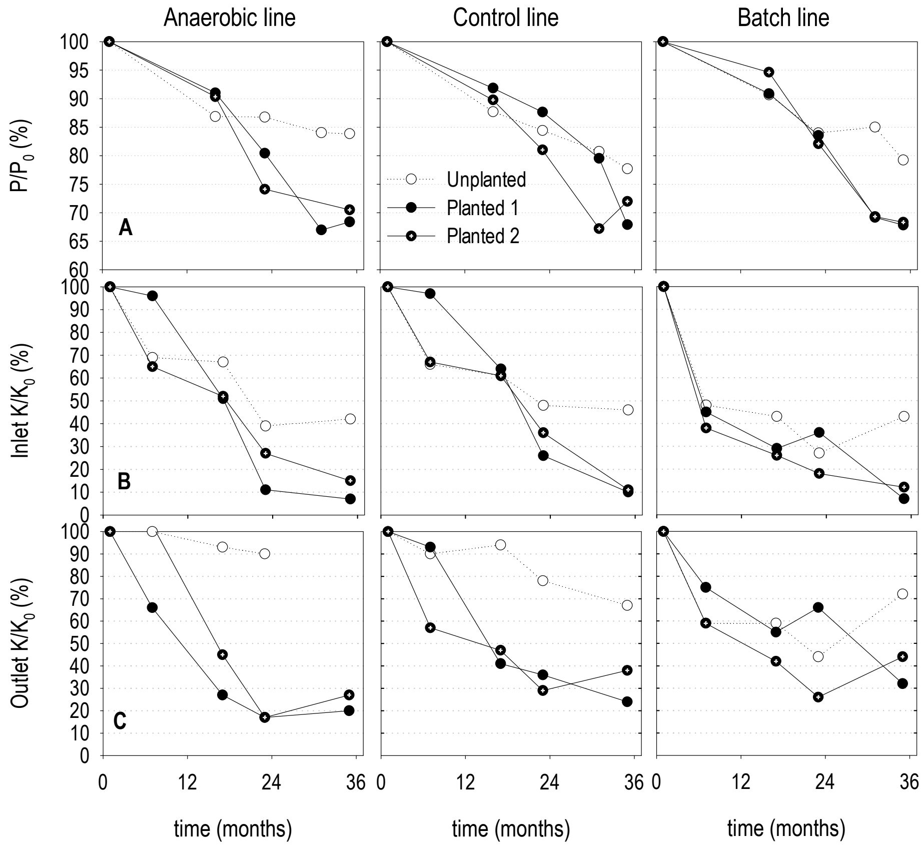

Figure 7.6 Temporal changes on a) porosity and hydraulic conductivity at inlet (b) and outlet (c) zones in the small wetlands of each treatment line. Values are plotted as a percentage of the initial ones (P/P0 and K/K0 for porosity and hydraulic conductivity, respectively).

7.3.5 Effective volume

Tracer test allows calculating the mean effective volume of water in a reactor from the mean hydraulic retention time (HRT) and the flow rate applied to the wetlands during the experiment. The most important decrease in HRT, and consequently in the effective volume, was for the batch line, although differences between experimental lines were not statistically different. In spite of the lack of defined trends observed from tracer tests, HRT reduction in the wetlands generally ranged from ca. 10% to ca. 30%, regardless the presence of macrophytes or treatment line (Table 7.2).

Table 7.2 Hydraulic characteristics calculated from the tracer test for the small wetlands of the experimental plant after three years of operation

		HRT (h)	Mean effective volume (L)	Difference to the theoretical volume[a] (%)	D/uL
Unplanted	Control	31.2	58.6	-9.9	0.19
	Batch	21.9	41.0	-36.9	0.27
	Anaerobic	32.0	60.1	-7.6	0.22
Planted 1	Control	45.7	85.6	31.7	0.09
	Batch	21.4	40.1	-38.3	0.68
	Anaerobic	32.3	60.5	-6.9	0.14
Planted 2	Control	34.7	65.1	0.2	0.22
	Batch	25.1	47.1	-27.5	0.20
	Anaerobic	41.6	78.0	20.1	0.13

[a] Negative signs indicate loss of volume

In two planted wetlands (planted 1 of control line and planted 2 of anaerobic) the HRT found was higher than the theoretical (32 and 20% higher, respectively). Consequently, the resulting effective volume was higher than expected (Table 7.2). Despite these odd results, dispersion factor (D/uL) for the majority of the small wetlands was within the range of moderate dispersion with regards to plug flow.

7.4 Discussion

In the first part of this discussion section indirect measures of clogging (hydraulic conductivity and porosity) will be compared to accumulated solids in order to assess the suspected relationships between clogging indicators. Furthermore, the effect of the type of primary treatment and hydraulic regime on clogging development will be addressed in the second and third part of this discussion section, respectively, by comparing planted wetlands of anaerobic and batch line with the control line. Finally, the detailed role of macrophytes on clogging will be specifically addressed in the last section of the discussion.

7.4.1 Clogging indicators

Although tracer tests have been widely used to assess HRT reduction due to filter media porosity decrease in constructed wetlands, direct correlation with organic matter accumulation is not straight forward (Tanner et al., 1998). Contrary to what we have found concerning solids accumulation and hydraulic and porosity measures, the effective volume and HRT calculations according to tracer tests evidence no general pattern for clogging (Table 7.2). It is necessary to point out that in some planted wetlands (control and anaerobic lines) HRT resulted in higher values than the theoretical (calculated according to initial media porosity). Higher values of HRT time in planted wetlands have been also described in literature (Drizo et al. (2000)). Although the exact reason for this result remains still uncertain it has been related to a water retention effect by the rizhosphere (Chazarenc et al. (2007). Overall, tracer tests, though to be a good tool to understand wetlands hydraulic behaviour, turns out to be a less sensitive method for indirect clogging assessment (at least under the condition here applied). Therefore, the results obtained from tracer tests will not be further discussed while addressing the clogging development of the experimental plant.

Generally, no direct correlations between clogging indicators are observed, due to the pattern distribution of accumulated solids and its quality characteristics (Tanner et al., 1998; Caselles-Osorio et al., 2007; Knowles et al., 2010a). In contrast, Pearson correlation coefficient between clogging indicators measured over time in this study revealed a significant direct (and negative) correlation (Table 7.3). However, accumulated sludge correlates better with both hydraulic conductivity and porosity than the accumulated roots.

Table 7.3 Pearson correlation coefficients between the clogging indicators measured in this study (n=36). Negative values indicate negative correlation.

	Hydraulic conductivity	Porosity
Total accumulated solids	-0.745*	-0.892*
Accumulated sludge	-0.730*	-0.781*
Accumulated roots	-0.538*	-0.720*

* Significant correlation α = 0.01

The lack of a better correlation between either hydraulic conductivity or porosity and accumulated sludge here presented might be, at least partially explained, by the nature of accumulated solids. Accordingly, density properties of accumulated solids in SSF CWs (solids nature) can vary to a great extent according to sampling location (Llorens et al., 2009). Sludge density and packing properties variation depending on labile and stable organic matter, in turn, may have a certain influence on both hydraulic conductivity and porosity values, regardless the amount of accumulated sludge (Nguyen et al., 2000).

On the other hand, accumulated roots presented a poor correlation with hydraulic conductivity, which could be due to the fact that only roots with diameters between 1-5 mm were analysed whereas plant rhizomes could represent more than 50% of the total belowground plant biomass (Edwards et al., 2006). In fact, there is a difference of a 10% in porosity reduction and a 30% in hydraulic conductivity reduction (Figure 7.6) between planted and unplanted wetlands that confirms the effect of belowground reeds in clogging processes. Furthermore, drainable porosity takes into account the whole wetland while hydraulic conductivity was measured in discrete points and at 20 cm depth. This could be the reason for better correlation between porosity and accumulated roots, which increased dramatically during the last year (and mainly at the bottom of the wetlands).

Despite the good correlation between hydraulic measurements (porosity and hydraulic conductivity) and solids accumulated, no differences were found between treatment lines with regards to reduction of both porosity and hydraulic conductivity. Drainable porosity might be considered the best strategy for indirect clogging assessment in experimental SSF CWs (namely because its correlation with accumulated solids is better). However, the procedure is of difficult applicability in full-scale facilities (namely because of the problems coupled to a complete drain of a full-scale wetland).Therefore, and as has been previously described (Chapters 3 - 5), in full-scale facilities the measurement of hydraulic conductivity seems to be a more suitable procedure for indirect clogging assessment.

7.4.2 Effect of primary treatment on clogging development

HUSB reactors have been described to significantly reduce the solids loading when treating domestic wastewater (Ligero et al., 2001; Ruíz et al., 2010). Our results are in agreement with this since the wetlands of the anaerobic line received ca. 20% and ca. 35% less solids than the control line for the whole experimental period and the second experimental period (HUSB reactor operated at 5 hours HRT), respectively. A reduction of the solids load (promoted by the HUSB reactor in this study) leads to lower sludge accumulation (roots not included) in the wetlands of the anaerobic line, which is in accordance to that described by Caselles-Osorio et al. (2007). Accordingly, in spite of the huge variation recorded in literature concerning solids accumulation of systems operated at similar solids loading rates (Table 7.4), solids accumulation rates in the anaerobic line were ca. 30% lower than those of the control line and roughly between 2 and 10 times lower than those generally reported for systems implemented with conventional primary treatment systems (septic tank) (Table 7.4).

Concerning the analysis of hydraulic measurements, no significant differences were recorded between treatment lines. Accordingly, the hydraulic conductivity method, though to be a reliable tool for on-site clogging assessment (Chapter 3 and 4), is not able to discriminate between small hydraulic conductivity differences (close clogging scenarios) (Chapter 5). Therefore, differences between treatment lines on accumulated solids (ca. 2 kg DM/m^2) might be still small (even after three years of operation) to be detected by both hydraulic conductivity method and drainable porosity (at least under the operational conditions tested).

7.4.3 Effect of operation strategy on clogging development

Authors hypothesise that alternating batch-unsaturated and permanently saturated phases may delay clogging due to an increase of oxygen transfer rates and subsequent enhancement of aerobic degradation of accumulated solids (Platzer and Mauch, 1997; Langergraber et al., 2003), which is supported by the higher performances on ammonium removal for the batch line. However, authors cannot confirm this hypothesis since, not only the amount of accumulated solids was of similar extent for the batch line, but accumulated sludge (no roots considered) was slightly higher (Figure 7.4a and 7.4c for the accumulated sludge and solids, respectively). Accordingly, one plausible explanation is that of, as has been already pointed out, higher oxygen transfer rates may enhance biofilm growth in SSF CWs (Chazarenc et al., 2009) which, in turn, may contribute to a higher sludge accumulation rather than favouring the aerobic degradation of accumulated solids. However, Chazarenc et al. (2009) found lower accumulation rates in unplanted and artificially aerated beds (Table·7.4) than in non-aerated. Therefore, whether batch operation mode may contribute to clogging delay in SSF CWs due to a greater oxygenation of filter media (initial research hypothesis) remains still to be proved (note that in batch strategy a passive aeration is conducted).

Although hydraulic conductivity values after 3 years of operation are not statistically different between treatment lines, the batch line present slightly higher values than the control at the outlet of the system (ca. 10% higher) (Figure 7.6C). This result is probably due to a lower root system development for the batch line (Figure 7.4B) as a consequence of the hydric stress caused by the alternation of saturated and unsaturated phases (Tanner et al., 1999).

7.4.4 Effect of vegetation on clogging development

The root system biomass after three years of operation ranged from 2 to 5 kg DM/m^2, regardless the type of primary treatment or the flow regime. These values are slightly higher than root biomasses recorded for common reed in gravel-based wetlands (Adcock and Ganf, 1994; Edwards et al., 2006). Furthermore, it has been described that the higher the belowground biomass of common reed, the closer to its optimal plant growth (Parr, 1990; Adcock and Ganf, 1994). Therefore, results suggest that macrophytes were in a very healthy state, regardless the experimental line considered. Roots mainly developed at the bottom of the systems (roots biomass within the last 10 cm accounted for 80% to 90% of the total root biomass in the wetland). Furthermore, roots accumulation in planted wetlands represented up to the 70% of the total accumulated solids at the end of the study period (35%, 50% and 70% for the batch, anaerobic and control line, respectively) (Figure 7.5). Although the contribution of plant detritus to solids accumulation in SSF CWs has

Table 7.4 TSS surface loading rates, accumulated solids, and solids accumulation rates in different SSF CWs studies.

Study	System	Primary treatment	Solids loading rate	Time	Accum. Solids[1]	Solids accum. Rate[1]
			g TSS/m² d	months	kg DM/m²	kg DM/m² year
Tanner and Sukias (1995)			2.2-7.3	22	1.9-6.5	1.5-4.5
Tanner et al. (1998)			2.2-7.3	60	8.5-18.6	1.3-3.0
Caselles-Osorio et al. (2007)	Verdú 1	Septic tank	2.8-4.5	48	2.8-12.8	0.7-3.2
	Verdú 2	Septic tank		48	2.3-11.9	0.6-2.9
	Alfés	Septic tank	3.2-4.9	36	2.6-35.1	0.6-8.8
	Corbins	Imhoff tank	6.5-10	48	6.0-57.3	1.5-14.3
	Almatret north	Septic tank	4.5-6.2	36	2.3-9.6	0.8-3.2
	Almatret south	Septic tank	2.6-8.0	36	2.8-20.3	0.9-6.8
Chazarenc et al. (2009)	Planted aerated		3.2	60	36.0-44.0	7.2-8.8
	Planted non aerated		3.2	60	44.0	8.2
	Unplanted aerated		3.2	60	43.0-44.0	8.6-8.8
	Unplanted non aerated		3.2	60	55.5-66.0	11.1-13.2
Ruiz et al. (2010)	Santiago de Compostela	UASB	5.9±6.5	36	3.2	1.07
This Study[2]	Batch Line	Settler	0.60-7.43	36	0.90-2.13 1.15-2.25 2.05-4.02	0.32-0.75 0.41-0.79 0.72-1.42
	Control Line	Settler	0.60-7.43	36	0.83-1.63 1.55-3.13 2.53-4.76	0.29-0.57 0.55-1.11 0.89-1.68
	Anaerobic Line	HUSB reactor	0.67-5.03	36	0.79-1.03 0.93-1.41 1.97-2.20	0.28-0.36 0.33-0.50 0.69-0.78

[1] Interstitial solids (sludge) are considered

[2] Planted and unplanted wetlands are considered, thus solids accumulation rates in this study correspond to the first third of the entire planted wetlands. For each line, sludge from the top 15 cm, the bottom 10 cm and the total depth are shown (by order).

been described to be important (Tanner et al., 1998; Nguyen, 2000), no significant differences on sludge accumulation (roots not considered) between planted and unplanted wetlands were recorded (regardless treatment line) (Figure 7.4A). This is probably due to the fact that aboveground biomass was harvested after plant senescence took place. Furthermore, the batch line showed a less developed root system when compared to the control line. Accordingly, it has been described that alternating saturated and unsaturated phases (batch operation) may cause hydric stress on macrophytes (Tanner et al., 1999) that, eventually, may lead to a lower development of both above and belowground biomass (Sasikala et al., 2009). Although macrophytes production has been related to nutrient uptake (Brix, 1997), we have no empirical evidence that a less developed root system in the batch line is coupled to a reduction of treatment efficiency. In fact, the batch line performed significantly better than the rest of treatment lines for most of analysed pollutants (specially for ammonium).

On the other hand, contrary to earlier believe, the presence of and macrophytes do not increase hydraulic conductivity in soil-based constructed wetlands (Brix, 1997) but they can even contribute to a notable extent to the reduction of the effective retention time in reed beds (Tanner and Sukias, 1995). Our results confirm the role of plants (belowground biomass) on hydraulic conductivity decrease since root biomass represent the 10% and 30% of the variation of drainable porosity and hydraulic conductivity, respectively (comparing the reduction between planted and unplanted beds). Therefore, the presence of well developed rizhomes (larger than 5 mm) may cause even a higher reduction of hydraulic conductivity or porosity in SSF CWs. Unfortunately, we cannot determine the extent of the contribution of large roots to hydraulic conductivity or porosity reduction, since only roots smaller than 5 mm are considered in this study. Edwards (1992) estimated the volume occupied by bulsrush roots (*Scirpus validus*) in a 5% of the total substrate of a wetland (46 cm depth), in which the majority of roots were developed in the first 12-15 cm depth and the maximum depth of penetration was ranged from 26 to 37 cm. In the planted wetlands of this study (30 cm depth) the belowground plant covered the total depth of the wetlands (Figure 7.5). Thus, in this case the percentage of volume occupied by belowground biomass (roots and rhizomes) should be clearly more important. Therefore, the extent of the effect of the root system on hydraulic conductivity reduction in SSF CWs must be further investigated, with even longer study periods. Furthermore, whether considering large roots (>5 mm) in studies addressing the clogging phenomena in SSF CWs will contribute to better explain the relation between accumulated solids and hydraulic conductivity or porosity remain still to be determined.

7.5 Conclusions

Hydraulic conductivity and drainable porosity are indirect measures of clogging highly correlated to solids accumulated in SSF CWs. More precisely, accumulated solids (roots and sludge included) represent the 74.5% and 89.2% of the hydraulic conductivity and drainable porosity variation, respectively. Accumulated sludge correlates better with hydraulic measurements than accumulated roots, probably due to the fact that rhizomes and roots larger than 5 mm could play an important role in the decrease of both hydraulic conductivity and drainable porosity.

Therefore, hydraulic conductivity and drainable porosity are suitable techniques to measure the extent of clogging in CWs, although in SSF CWs implemented in this study are of similar extent after three years of operation. However, hydraulic conductivity seems to be a more suitable procedure for indirect clogging assessment due to its applicability in full-scale CWs.

Despite the lack of differences in hydraulic parameters between treatment lines, a SSF CW implemented with a HUSB reactor as primary treatment accumulates ca. 30% lower sludge than a system implemented with a conventional settler. Therefore, the implementation of HUSB reactor is recommended in order to delay solids accumulation in SSF CWs.

A SSF CW operated under batch regime accumulates slightly higher sludge than a SSF CW operated under continuous hydraulic regime, probably as a consequence of biofilm growth stimulation. However, because root system in a batch operated systems is less developed than in a continuous operated system (probably as a consequence of great hydric stress on macrophytes), overall differences on solids accumulation between both feeding strategies are not significantly different after three years of operation.

Macrophytes greatly contribute to clogging phenomena in SSF CW. Accordingly, roots accumulation in planted wetlands represented up to the 70% of the total accumulated solids at the end of the study period, (ca. 35% for the batch line and ca. 70% for the anaerobic and control line). Furthermore, hydraulic conductivity and porosity were lower in planted than in unplanted wetlands (30% and 10% lower to hydraulic conductivity and drainable porosity, respectively), regardless the type of primary treatment or flow regime.

8

Mineralogy and resistance properties of gravel as a factor affecting clogging in subsurface flow wetlands[6]

Gravel constitutes the filter medium in subsurface flow constructed wetlands (SSF CWs) and its porosity and hydraulic conductivity decrease over time (clogging) limit the lifespan of the systems. Using gravel of poor quality may contribute to accelerate clogging in wetlands. In this study, gravel samples from six different wetland systems were compared with regards to its mineral composition and mechanical resistance properties. Results showed that both mineralogy and texture are related to mechanical resistance. Accordingly, gravel with high content of quartz (>80%) showed a lower percentage of broken medium (0.18-1.03%) than those with lower content of quartz (2.42-4.56% media broken). Although granite is formed by high durability minerals, its non-uniform texture results in a lower resistance to abrasion (ca. 10% less resistance than calcareous gravel). Therefore, it is recommended to use gravels composed basically by quartz or, when it is not available, limestone gravels (rounded and uniform) are recommended instead. The resistance to abrasion (LAA test) seems to be a good indicator to determine the mechanical properties of gravels used in CWs. It is recommended to use materials with LAA values below 25% or at least not exceeding 30%.

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[6] This chapter is based on the article:

Pedescoll, A., Passos, F., Alba, E., Puigagut, J., García, J. Mechanical resistance properties of gravel used in subsurface flow constructed wetlands: implications for clogging. Water Science and Technology, submitted.

8.1 Introduction

It is widely recognized that clogging is the main operational problem in of subsurface flow constructed wetlands systems (SSF CWs) (Knowles et al., 2010b). Solids accumulation, both from external (wastewater suspended solids) and internal sources (chemical precipitation, plant detritus accumulation and biofilm growth) results in porosity and hydraulic conductivity decrease among time (Platzer and Mauch, 1997; Caselles-Osorio et al., 2007; Kadlec and Wallace, 2009). In early stages, clogging develops at the inlet of the wetland and results in water ponding, preferential flow paths and hydraulic shortcircuits. Ultimately, clogging processes can cause the failure of the system leading to a decrease in pollutant removal efficiency (Rousseau et al., 2005).

Since there are just few control factors once these systems are operational, it is essential to apply for the best design criteria. Accordingly, many design factors are (considered of capital importance, such as organic and solids load (Wallace and Knight, 2006), upstream treatment processes (Cooper et al., 2005; Barros et al., 2008; Griffin et al., 2008), dimensioning and width to length ratio of the bed (García et al., 2005), inlet flow distributors (Griffin et al., 2008; Knowles et al., 2010a), operational strategies (Behrends et al., 2001; Caselles-Osorio and García, 2007a), vegetation choice and its maintenance (Bécares, 2004) or filter media characteristics (Chapter 4).

SSF CWs are biological reactors in which filter media (namely gravel) constitutes the support for plant and biofilm growth. Filter media's hydraulic conductivity is directly related to particle size and granulometry distribution (Kadlec and Knight, 1996), although the shape of the particles is also important. Therefore, it is recommended to use washed gravels with median sizes of 6-11 mm as much uniform as possible in order to avoid the presence of large quantities of fine materials (Griffin et al., 2008). Furthermore, rounded gravels are recommended over irregular gravels to maintain initial hydraulic properties.

Construction and building materials should meet certain requirements for its use. Aggregates used in Portland cement concrete should be among others clean, hard, strong, uniform and resistant to abrasion (Ugur et al., 2010). Gravels are exposed to mechanical and chemical degrading forces during transportation and construction of CWs and also during the operation time, since the filter media is in contact with wastewater and microorganisms, thus exposed to biological reactions. Accordingly, wetlands filter medium should have similar properties than those required for civil construction.

Granitic gravel is generally recommended for the construction of wetlands (García and Corzo, 2008). In practice, however, and due to economical reasons, the gravel employed for wetlands construction is the one that can be obtained from the nearest supplier, regardless its mineral composition. Furthermore, in Chapter 4 it was found that sludge accumulated in two different wetlands had the same mineral composition than that of the gravel and hypothesized that gravel would dissolve or crumble, exacerbating the clogging process. This study aims at evaluating the mechanical and chemical resistance of different types of gravel. Results obtained will contribute to assess best design criteria to minimize clogging derived from media disintegration in subsurface-flow constructed wetlands.

8.2 Methods

Filter media samples (consisting in gravel) from six different wetlands treating urban sewage were obtained in order to conduct this study (Figure 8.1). Wetlands are located at Corbins, Verdú, Arnes, Gualba (Catalonia, Spain), Moreton Morrell (Warwickshire, UK) and an experimental plant located at the Technical University of Catalonia (Barcelona, Spain) (NEWWET's gravel). In all these wetlands, several parameters related to clogging were previously measured, such as hydraulic conductivity and solids accumulation. For Corbins and Verdú data are available in Pedescoll et al. (2009) (Chapter 4), for Arnes and Gualba data are partially available (hydraulic conductivity) in Samsó et al. (2009) (Chapter 3) and for Moreton Morrell in Knowles et al. (2009) (Chapter 5), whereas the rest is yet to be published. Main wetland's characteristics related to operation, clogging and gravel properties are shown in Table 8.1. Samples were taken in different sampling campaigns. For Verdú and Corbins samples were taken in april 2007; Arnes and Gualba in march 2009; Moreton Morrell in February 2009 and NEWWET's gravel was the original material used as filter medium of a experimental plant (Chapters 6 and 7).

Table 8.1 Characteristics related to clogging measured at the moment of gravel samples collection.

Wetland characteristics							Gravel characteristics	
System	Time of operation	Type of wetland	Surface TSS load	Hydraulic conductivity	Solids accumulated	VS/TS (of accumulated solids)	D_{60}	C_u
	months		g/m^{2}d	m/d	kg DM/m^2	%	mm	
Corbins	60	Primary	9.3 – 80.9	0.2 – 811	1.7 – 10	15 – 26	9.2	1.8
Verdú	60	Primary	3.4 – 10.4	0.5 – 701	1.2 – 14	14 – 33	9.0	1.8
Arnes		Primary	10.5 – 29.5	1.3 – 42.1	1.8 – 5.8	7 – 47	7.8	0.6
Gualba	12	Primary	26.5 – 89.9	1.7 – 39.9	1.8 – 18	2 – 37	8.1	0.5
Moreton Morrell	180	Secondary	n.a.	0.01 – 4.0	2.4 – 9.4	20 – 43	6.9	0.6
NEWWET	36	Primary	0.6 – 7.4	19 – 270	6.2 – 15.2	30 – 45	7.3	0.8

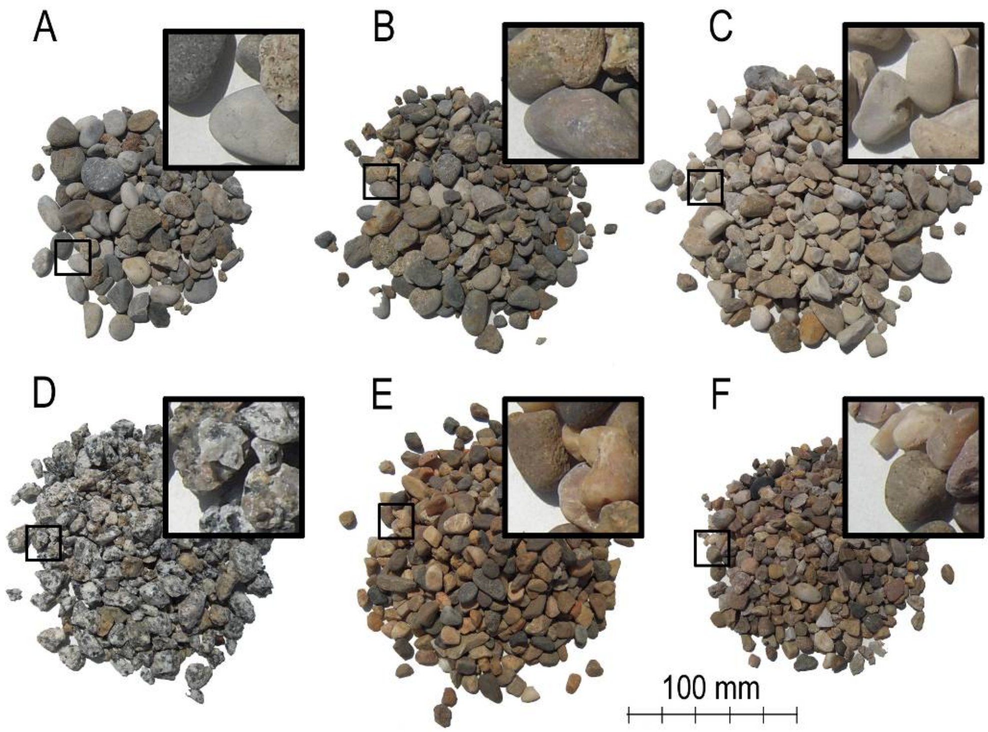

Figure 8.1 Gravel samples physical appearance. A Corbins, B Verdú, C Arnes, D Gualba, E Moreton Morrell, F NEWWET.

8.2.1 Mineral composition

The six gravel samples were processed according to the EN-933-1 and EN-932-2 standards (AENOR, 1999, 1998a). Samples were milled to a particle diameter of 0.063 mm and were subjected to X-ray diffraction analyses using a Siemens D-500 diffractometer equipped with a copper pipe. Diffractometry was carried out from 4º to 70º with steps of 0.05º every 3 s. Difractograms were interpreted by means of Eva software. From the standards selected during the difractograms interpretation a semi-quantitative analysis was also conducted.

8.2.2 Mechanical resistance

Gravel mechanical resistance was assessed by the application of three different standard tests commonly used for soil science. Accordingly, the tests applied were the Standard Proctor test

(UNE 103500 (AENOR, 1994), which corresponds to ASTM D-698:1978) (breakability), Los Angeles abrasion (LAA) test (UNE-EN 1097-2 (AENOR, 1998b), which corresponds to ASTM C-131:1965) and the Slake durability test (NLT-251/91 (CEDEX, 1991), which corresponds to ASTM D-4644-87:1998) (to assess the durability of the media under cycles of moisture and dryness).

The Standard Proctor test consists of subjecting a sample to 26 impacts of a metal hammer (2.5 kg). In soil science results are expressed in terms of soil compacting degree; however, we modified the outcome of the test to express the degree of broken media. To this end, a granulometric test was performed before and after the Proctor test was applied to each gravel, and the percentage of broken media was calculated as:

$$B = \left[\frac{D_{60o} - D_{60f}}{D_{60o}} \right] \times 100 \tag{8.1}$$

where:

B is the amount of broken media, in %; D_{60o} is the D_{60} value before Proctor test was applied (in mm); D_{60f} is the D_{60} value after Proctor test was applied (in mm). For each gravel sample the test was performed in triplicate.

LAA test consists of subjecting 5 kg of a sieved fraction of gravel sample (with grain size between 14-10 mm) to abrasion and attrition forces by means of introducing the gravel sample and 9 steel balls in a standardized drum. After 500 laps at constant 31-33 rpm, the sample is sieved again (between 14-10 mm) and weighted. Finally LAA index is expressed as the percentage of material loss calculated from weights. Moreton morrell's and NEWWET's gravels were not subjected to this procedure due to the lack of sufficient amount of sample with grain size specified in the standard procedure. Due to the lack of sample amount the test was performed once.

Finally, slake durability test consists of evaluating the weight loss of a gravel sample ($\sim$0.5 kg) subjected to cycles of dry and moisture. The test consists of introducing the gravel sample in a drum (made of metal mesh) submerged in 20 mm of tap water (water level is below the drum shaft). The drum containing the sample is subjected to 40$\pm$1 rpm for 30 minutes. After that, the sample is dried and weighted again and the weight loss calculated. The procedure was conducted twice in order to obtain the I_{d2} (index after two cycles of drying-moisture). Each test was conducted in triplicate.

8.3 Results and discussion

8.3.1 Gravel composition

Results from X-ray diffraction showed that, except for the Arnes' gravel (calcareous nature with dolomite (~ 65%) and calcite (~35%)), analysed gravels were of granitic nature as recommended by García and Corzo (2008) (Table 8.2). Moreton Morrell's and NEWWET's gravels are composed basically by quartz. Corbins and Verdú gravels presented a combination of quartz and calcite (ca. 60% of the total composition) and Gualba's gravel presents typical granite composition (quartz, mica and feldspar). Therefore, with the exception of Corbins and Verdú, all the gravels are fairly different between them with regards to its mineralogy. Note that Corbins and Verdú treatment plants were constructed and are operated by the same company, and are emplaced within the same geographical area, thus probably their gravel comes from the same quarry.

Table 8.2 Relative percentages of minerals in gravel samples obtained from the semi-quantitative x-ray diffractometry.

		Corbins	Verdú	Arnes	Gualba	Moreton morrell	NEWWET
Quartz		30	30	1	45	99	99
Mica	Biotite				10		
	Muskovite	20	15				
Feldspar	Albite	10	15		30		
	Microcline				15		
Limestone	Calcite	35	30	33			
	Dolomite			66			
Other		5	10				

Quartz SiO_2
Biotite $K(Mg,Fe)_3(Al,Fe)Si_3O_{10}(OH,F)_2$
Muskovite $KAl_2(Al,Si)_4O_{10}(OH,F)_2$
Albite $NaAlSi_3O_8$
Microcline $KAlSi_3O_8$
Calcite $CaCO_3$
Dolomite $CaMg(CO_3)_2$

8.3.2 Mechanical resistance

Values from LAA test obtained for the different gravels (Table 8.3) are in the range of those previously observed by other authors (Pacheco Torgal and Castro-Gomes, 2006; Gonilho Pereira et al., 2009; Ugur et al., 2010). However, Gualba's gravel (showing a 33% of weight loss by abrasion) is not within the allowed limits established for media devoted to construction purposes in Spain (BOE, 2008) and presents also a higher degree of abrasion than that generally described for granitic gravels (22-27%) (Pacheco Torgal and Castro-Gomes, 2006).

The resistance of gravel to alternate periods of moisture and dryness provide very similar results for all tested gravels, though Gualba's gravel showed also the lowest value (Table 8.3). According to Johnson and DeGraff (1988) the output of the test allows classifying all tested gravels into the very high durability group. It is important to point out, however, that in this study Slake durability test was carried out under neutral conditions (water pH around 7). Gravel in SSF CWs is exposed to both temporal and spatial pH variations (Kaseva, 2004; Wießner et al., 2005), and pH has been described to affect the slake durability (Gupta and Ahmed, 2007). Therefore, further research should be carried out to determine the extent of the effect of the pH on mechanical resistance of gravels placed in constructed wetlands.

Table 8.3 Results of mechanical and chemical resistance for the gravel samples analysed.

	Resistance to breakage B [a] (%)	Resistance to abrasion LAA [b] (%)	Dry an moisture resistance $I_{d\,2}$ [c] (%)
Corbins	3.05±0.84	19	99.3
Verdú	2.49±1.36	17	99.3
Arnes	2.42±0.53	23	99.1
Gualba	4.56±2.49	33	98.3
Moreton Morrell	1.03±0.57	-	99.3
NEWWET	0.18±0.15	-	99.5

[a] Broken media calculated from the Proctor test
[b] Los Angeles Abrasion
[c] Index for Slake durability test after two cycles of drying-moisture.

Similarly, proctor test showed the highest percentage of broken media for Gualba's gravel whereas Moreton Morrell and NEWWET's gravel were the most resistant materials (Table 8.3). Mechanical resistance tests showed quite good correlations between them (Figure 8.2). The use of any of the applied tests may lead to a similar conclusion in regards to mechanical resistance properties of gravel. However, LAA test offered a greater variation in the results obtained and therefore, LAA test allows to describe differences between samples with a greater resolution than the other two tests applied in this study.

The results of this study indicate that higher mechanical resistance generally corresponds to gravel in which quartz is the main mineral compound, except for Gualba's gravel. Gravel durability is related not only to its mineralogical composition, but also to its texture (Dhakal et al., 2002). Accordingly, the non-homogenous texture of Gualba's gravel (Figure 8.1) seems to be the reason why it presents a lower mechanical resistance in spite of its dominant granitic composition. This hypothesis is consistent with the fact that Gualba's wetland, though to be a young system (its gravel was almost completely replaced one year before sampling), presents not only a solids accumulation in the range of older wetlands (such as Corbins or Verdú systems) but also a high degree of inorganic material accumulated (Table 8.1).

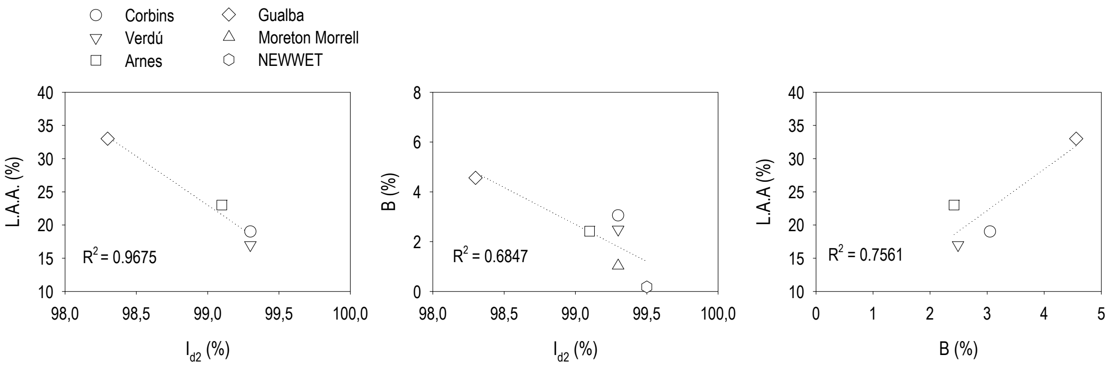

Figure 8.2 Correlations between resistance tests.

When comparing the distribution of accumulated solids from Gualba and Arnes's beds (Figure 8.3) it could be seen that Gualba presented higher accumulations of solids (basically inorganic solids) at the end of the wetland while at Arnes the accumulation was more uniform along the length of the cell. The distribution of solids (both, total and volatile solids) leads in a logical distribution of hydraulic conductivity at Arnes (increasing the values from the inlet to the outlet) whereas at Gualba the hydraulic conductivity distribution showed high heterogeneity, probably promoted by the aspect ratio of the wetland (Samsó et al., 2009) and high degree of clogging due to the greater surface organic load (Table 8.1). Furthermore, the clogging distribution seems to be related to the nature of accumulated solids (Llorens et al., 2009). In this sense inorganic solids occupied less volume than organic matter, which is consistent with the results obtained in Arnes' cell. Although, at the outlet of Gualba's bed the amounts of inorganic solids was twice the Arnes' wetland in only one year of operation since the gravel was replaced in Gualba. This situation could be promoted by the poor quality of its gravel.

In order to control clogging in SSF CWs it is recommended to remove plant debris (Turon et al., 2009) and harvest the vegetation yearly (in Mediterranean regions) to avoid the necromass accumulation at the surface of the wetland (Bécares, 2004). When possible, this action is carried out using heavy machinery. Therefore, gravel would be exposed to mechanical degrading forces during transportation (abrasion attack) from the quarry to the facility and maintenance tasks (compaction forces). Thus, gravel should be mechanically resistant to avoid the crumbling of the grains. From the results obtained in this study is not possible to predict how much will contribute the gravel matrix to clogging development since it depends, in a greater extent, on the operation of the wetland such as the use of heavy machinery in maintenance tasks.

Furthermore, it is necessary to conduct resistance tests exposing the gravel to different pH conditions. However, the resistance to abrasion (LAA test) seems to be a good indicator to determine the quality of gravel. In this sense, values below 25% for this test showed good performances in the wetlands studied while for values above 30% (the case of Gualba) the hydraulic behavior in the bed could be jeopardized by the accumulation of inorganic solids from gravel crumbling.

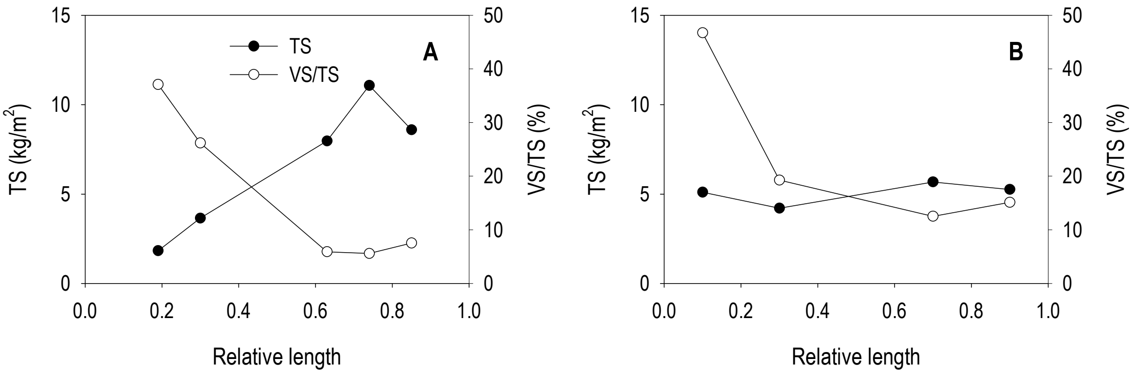

Figure 8.3 Total solids (TS) and volatile solids fraction (VS/TS) distribution along the length of two wetlands, A) Gualba and B) Arnes

8.4 Conclusions

This study has addressed the relationship between gravel quality (in terms of mineral composition and mechanical resistance properties) and clogging in SSF CWs. Accordingly, gravels composed basically by quartz are harder than those containing calcareous materials, thus less susceptible to disaggregate under mechanical stress. Furthermore, minerals composing granite gravel are durable; however, its non-homogeneous texture is more susceptible to be disaggregated by any mechanical action. Therefore, it is preferable to use limestone pebbles (rounded and uniform) than harder materials but not homogeneous when quartz-rich gravel is not available. LAA test is presented as the most adequate indicator to determine the mechanical properties of the gravel. From the results it is recommended to use gravels with resistance to abrasion below 25% or at least that not exceed 30%, regardless its mineral composition.

9

Discusión general

En este Capítulo se discutirán de manera general los distintos aspectos tratados a lo largo de los Capítulos anteriores y que conducen a responder a los objetivos formulados en la tesis. Por ello la discusión se estructura en cuatro puntos que pretenden dar respuesta a los seis objetivos específicos expuestos en el segundo Capítulo.

9.1 El permeámetro de carga variable para la medida *in situ* de la conductividad hidráulica de humedales construidos de flujo subsuperficial horizontal

En esta sección se discuten los puntos concernientes al primer objetivo de la tesis y que giran entorno a la idoneidad del permeámetro de carga variable como técnica para la medida indirecta del estado de colmatación de un humedal.

La conductividad hidráulica resulta un parámetro muy útil para conocer el estado de colmatación de un humedal. La acumulación de sólidos en el humedal a lo largo del tiempo disminuye la porosidad del medio filtrante (Brix, 1997; Tanner et al., 1998; Nguyen, 2000; Wallace and Knight, 2006; García et al., 2007) y por lo tanto la capacidad del suelo de permitir la circulación del agua a través de él (conductividad hidráulica) y aumenta la resistencia al flujo de agua en el interior del lecho. Este parámetro constituye una importante propiedad de los suelos y se aplica frecuentemente en actividades relacionadas con la agricultura y geotecnia (Reynolds et al., 2000).

El mayor problema al que nos enfrentamos al medir la conductividad hidráulica es la imposibilidad de obtener un testigo inalterado y representativo del medio para analizarlo mediante técnicas estandarizadas como el permeámetro de carga constante *ex situ*, ya que el medio filtrante de los humedales subsuperficiales es un material no cohesivo. Por eso es necesario desarrollar técnicas para la medición *in situ* de este parámetro. Uno de los métodos para estimar la conductividad hidráulica en humedales se basa en medir la pérdida de carga o el gradiente hidráulico entre dos puntos del lecho aplicando un caudal conocido de agua (Sanford et al., 1995; Rodgers and Mulqueen, 2006; Suliman et al., 2006a). Otras técnicas disponibles son el permeámetro de carga constante y el de carga variable (Wilson et al., 2000). Todas ellas se basan en la ley de Darcy para el cálculo de la conductividad hidráulica.

En esta tesis se ha comparado un método de medición de la conductividad hidráulica para la evaluación *in situ* de la colmatación basado en el permeámetro de carga variable descrito en NAVFAC (1986) con el permeámetro de carga constante tanto en condiciones de laboratorio (Capítulo 3) como en un humedal construido a escala real (Capítulo 5). El permeámetro de carga variable aplicado en este estudio se desarrolló para medir la conductividad hidráulica en materiales altamente conductivos y se aplicó con éxito en humedales construidos (Caselles-Osorio et al., 2007). Se basa en medir el tiempo que tarda una columna de agua en atravesar el material contenido en una zona de sondeo.

Esta técnica es fácilmente aplicable para medir la conductividad hidráulica en puntos discretos de un humedal, por un lado porque requiere un equipo sencillo y relativamente económico (un tubo de acero, una sonda de presión hidrostática y un sistema de recogida de datos). El equipo usado en este estudio cuesta en total unos 2500 € (aunque dependiendo del modelo de datalogger y la sonda de presión el coste puede ser inferior). A efectos prácticos la técnica se puede aplicar sin usar este equipo, utilizando un cronómetro para medir el tiempo inicial y final del experimento, aunque esto influye negativamente en la precisión de la medida de la conductividad hidráulica. Por otro lado se trata de un método bastante intuitivo para medir la velocidad de infiltración del agua en el lecho. Además, como se vio en los Capítulos 3, 4 y 5, la interpolación entre puntos de medición nos ofrece información acerca de la distribución horizontal de la colmatación y demuestra la existencia de caminos preferenciales en el interior del lecho, tal y como se había descrito previamente en la literatura utilizando otros métodos de conductividad hidráulica (Knowles and Davies, 2009).

Algunas técnicas disponibles para medir la velocidad de infiltración, como el permeámetro de Guelph (Langergraber et al., 2003) o el infiltrómetro de tensión (Youngs et al., 1995; Reynolds et al., 2000), están estandarizadas para medir en rangos de conductividad bajos, propios de suelos para el cultivo, arcillas o arenas muy finas (entre 0 y 10 m/d). Sin embargo en un humedal, generalmente compuesto de gravas medias y finas, el rango de conductividad hidráulica es mucho más amplio, entre 0 m/d en casos de colmatación completa y más de 500 m/d en un medio granular limpio (Knowles and Davies, 2009). El permeámetro de carga variable utilizado en este estudio permite medir en todo este rango. Además, como se vió en el Capítulo 3, el permeámetro de carga variable es un método preciso, cuyas medidas no dependen de la distancia del punto de medición a la salida del lecho y presenta buena repetibilidad (Tabla 9.1).

Tabla 9.1 Estadísticos básicos para las conductividades hidráulicas (m/d) medidas en arena y grava con los permeámetros de carga constante y variable en condiciones de laboratorio.

	Arena		**Grava**	
	CH	FH	CH	FH
N	19	18	6	18
Mínimo	28.87	3.31	176.73	284.83
Máximo	48.48	5.38	363.60	397.99
Media	38.65	4.35	364.94	346.49
CV (%)	**12.59**	**14.59**	**29.08**	**11.34**

CH: permeámetro de carga constante
FH: permeámetro de carga variable
N: número de ensayos realizados
CV: coeficiente de variación

Existen, no obstante, discrepancias en las medidas tomadas con este método al compararlas con un permeámetro de carga constante bajo condiciones de laboratorio (Capítulo 3). En rangos de conductividad bajos (0-30 m/d), medidos en arena (D_{50} = 0.9 mm) el permeámetro de carga variable (FH) presentó valores de aproximadamente un 80% inferiores que los obtenidos con el permeámetro de carga constante (CH) (Tabla 9.1). En cambio, para rangos superiores de conductividad hidráulica medidos en grava (D_{50} = 7.1 mm) la diferencia entre ambos métodos fue del 20%, con valores superiores para el permeámetro de carga variable.

Igualmente se encontraron diferencias entre el permeámetro de carga variable y el de carga constante aplicando ambos métodos *in situ*. Se observó una relación potencial entre ambos métodos ($FH = CH^{0.7821}$) de modo que a medida que los valores aumentan para el permeámetro de carga constante, aquellos obtenidos con el de carga variable crecen más lentamente. Esto indica que el permeámetro de carga variable es un método menos sensible a cambios en la conductividad hidráulica en rangos bajos (entre 0-70 m/d) (Capítulo 5). Sin embargo estas diferencias se pueden considerar aceptables (Elrick and Reynolds, 1992; Reynolds and Zebchuk, 1996; Elrick et al., 2002), por lo que el método propuesto en esta tesis se considera aplicable para medir la distribución espacial de la colmatación en humedales construidos. Estas diferencias fueron del mismo orden comparando ambos métodos en condiciones de laboratorio e *in situ* (Figura 9.1).

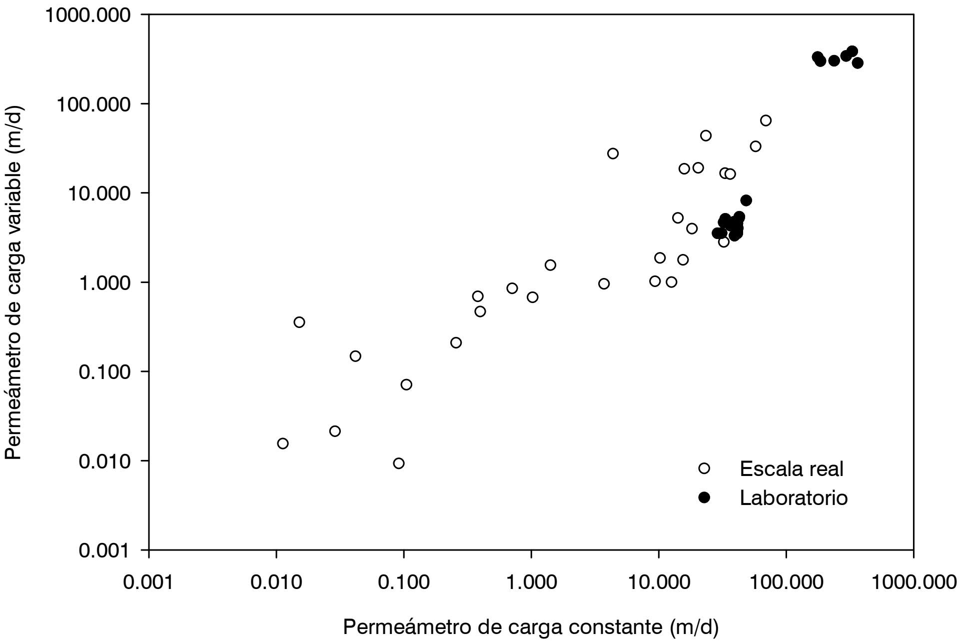

Figura 9.1 Nube de puntos entre valores experimentales de conductividad hidráulica tomados con el permeámetro de carga variable y el permeámetro de carga constante. Se diferencian los ensayos en condiciones de laboratorio y los puntos medidos en un humedal horizontal a escal real (n=53). Los ejes se representan en escala logarítmica.

La medida de la conductividad hidráulica mediante el permeámetro de carga variable es una herramienta útil para el seguimiento de la colmatación de un humedal y la detección de factores de diseño y operacionales relacionados con el desarrollo de este fenómeno (como se discutirá más adelante). Así, esta técnica ofrece una valiosa información acerca del funcionamiento hidráulico de un lecho permitiendo al explotador de una planta de tratamiento de agua basada en humedales construidos realizar las actuaciones necesarias para mejorar la hidráulica del sistema y optimizar el presupuesto de estas actuaciones (por ejemplo en el caso de una reposición parcial de la grava).

9.2 Indicadores de la colmatación en humedales construidos de flujo subsuperficial

La colmatación es el principal problema operacional de humedales construidos de flujo subsuperficial y su desarrollo en el tiempo limita la vida útil del mismo (Cooper et al., 2005; Cooper, 2009; Knowles et al., 2010b). Su detección y la determinación de los parámetros que afectan su desarrollo deben conducir a estrategias y diseños más efectivos que permitan retrasar la aparición de la colmatación y alargar, por lo tanto, la vida del humedal. Por ello resulta esencial medir el grado de colmatación en sistemas de humedales de manera fiable. En esta sección se discute sobre los distintos indicadores de la colmatación, lo que atañe al segundo objetivo de la tesis.

La colmatación tiene lugar por la acumulación de sólidos de diversa naturaleza y que provienen de distintas fuentes, por lo que su medida no es trivial (Tanner et al., 1998; Caselles-Osorio and García, 2006, 2007b; Llorens et al., 2009). Disponemos de distintos indicadores de esta colmatación, como son la acumulación de sólidos, la conductividad hidráulica, la porosidad del medio o el tiempo de retención hidráulico (HRT). En el Capítulo 7 se compararon estos indicadores a fin de determinar su grado de adecuación para evaluar la colmatación de humedales.

Se evaluó la porosidad drenable, la conductividad hidráulica, la acumulación de sólidos y el volumen efectivo (calculado a partir de un ensayo de trazadores) en una planta piloto, construida y operada durante tres años para estudiar el efecto de dos tratamientos primarios y dos modos de operación en el desarrollo de la colmatación. El tiempo de retención hidráulico (y por lo tanto el volumen efectivo del lecho), que *a priori* debería disminuir al desarrollarse la colmatación (Tanner et al, 1998), no presentó los resultados esperados. Algunos de los humedales plantados estudiados presentaron volúmenes efectivos mayores que el teórico (Figura 9.2).

El ensayo con trazadores es una buena herramienta para conocer el tipo de flujo de un reactor y nos informa sobre caminos preferenciales, zonas muertas y cortocircuitos en el flujo del agua (Batchelor and Loots, 1997; Kadlec and Wallace, 2009; Knowles et al, 2010a). En la tabla 9.2 se resumen algunos parámetros obtenidos de las curvas de distribución del trazador a la salida en los humedales estudiados en la planta piloto. No se observa patrón alguno atribuible a la presencia o ausencia de plantas o al tipo de tratamiento primario. Sin embargo, la línea batch sería la más ineficiente desde el punto de vista hidráulico. El grado de dispersión del trazador en el reactor se puede expresar como el número de tanques de mezcla completa de que se compone el sistema. Un solo tanque indica que el reactor es de mezcla completa mien-

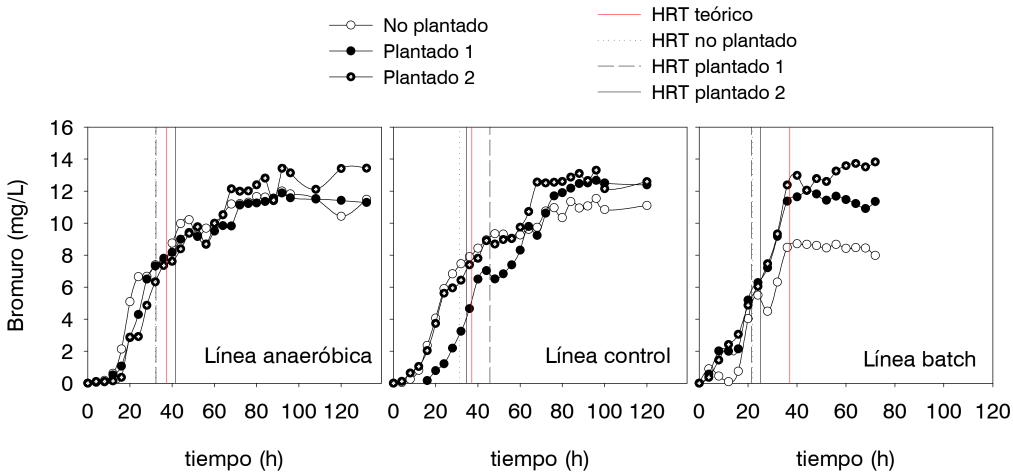

Figura 9.2 Curvas de trazador a la salida de los humedales y tiempos de retención (HRT) calculados a partir de éstas. En rojo se representa el HRT teórico calculado como HRT=V_{poros}/Q donde $V_{poros}=V_{grava\ saturada}$ porosidad

tras que para un valor >100 se considera flujo pistón (Chazarenc et al., 2003). En todos los casos existe dispersión del flujo aunque en la línea batch ésta es mayor. Del mismo modo, el volumen efectivo en esta línea es menor y el porcentaje de zonas muertas (léase colmatadas) mayor. En cambio no se encontraron evidencias de esto con otros indicadores como la conductividad hidráulica o la porosidad drenable. Y es que el flujo de agua en un humedal, lejos de comportarse idealmente, puede discurrir a través de pequeños cortocircuitos de modo que no existe un único tiempo de retención en el reactor sinó una distribución de HRT (Jenkins and Greenway, 2005). Por esta razón el cálculo del tiempo de retención hidráulico (o del volumen efectivo que se deduce) tal como se ha aplicado en este estudio no es un buen indicador del estado de colmatación de un lecho. Sin embargo, la distribución espacial de la colmatación puede evaluarse mediante un ensayo de trazadores analizando la curva del trazador en una matriz de puntos en el humedal. Knowles et al. (2010a) determinaron que el modelo de flujo obtenido de este ensayo se complementa con la conductividad hidráulica medida en los mismos puntos.

Tabla 9.2 Parámetros hidráulicos obtenidos a partir de la curva de distribución del trazador a la salida de los humedales estudiados

		HRT teórico (h)	HRT observado (h)	Número tanques en serie*	Volumen efectivo** (%)	% zonas muertas
Control	No plantado	34.7	31.2	3.59	90.14	9.86
	Plantado 1	34.7	45.7	7.66	131.73	0.00
	Plantado 2	34.7	34.7	3.17	100.20	0.00
Batch	No plantado	34.7	21.9	2.63	63.15	36.85
	Plantado 1	34.7	21.4	1.49	61.65	38.35
	Plantado 2	34.7	25.1	3.47	72.52	27.48
Anaeróbica	No plantado	34.7	32.0	2.70	92.45	7.55
	Plantado 1	34.7	32.3	4.63	93.07	6.93
	Plantado 2	34.7	41.6	4.80	120.06	0.00

* El número de tanques de mezcla completa en serie se ha calculado como la inversa de la varianza de la distribución del trazador a la salida (Levenspiel, 1999). Cuanto más se aleja este valor de 1 más se asemeja a un flujo en pistón.
** Volumen efectivo = HRT observado/HRT teórico

El desarrollo de las raíces y rizomas de las plantas pueden, por un lado promover flujos preferenciales del agua en el fondo del humedal y, por otro retener el agua retrasando su salida y aumentando el tiempo de retención en el humedal (Bowmer 1987; Drizo et al., 2000). Esto último conduce a mayores volúmenes efectivos del lecho que no tienen que ver con la acumulación de sólidos (Chazarenc et al., 2007). De hecho, únicamente en la línea batch se observa la disminución del HRT para los tres lechos. Sin embargo tanto la pérdida de conductividad hidráulica como de porosidad fue aproximadamente igual en las tres líneas (comparando los humedales plantados por un lado y los no plantados por otro). En cambio sí se observaron diferencias entre humedales plantados y no plantados (las pérdidas de conductividad hidráulica en humedales plantados fueron un 30% superiores que en humedales no plantados), hecho que no queda reflejado en el tiempo de retención hidráulico.

Respecto al resto de indicadores estudiados se hallaron buenas correlaciones entre la acumulación de sólidos y la pérdida de conductividad hidráulica y porosidad en los lechos estudiados (Capítulo 7). En la Figura 9.3 se representa la pérdida de conductividad hidráulica y de porosidad en función de los sólidos acumulados (entendiendo por sólidos acumulados tanto la acumulación de lodo como la de raíces). La conductividad hidráulica y la porosidad presentan una tendencia exponencial negativa frente al aumento de sólidos. La acumulación de lodo acarrea una pérdida de conductividad hidráulica de hasta un 60% (humedales no plantados) y una disminución en la porosidad de hasta un 20%. En los humedales plantados los valores iniciales de conductividad disminuyen en un 90% y los de porosidad en un 35% debido a la presencia de raíces. En un estudio realizado por Tanner et al. (1998), el balance de masas entre la carga orgánica aplicada en un humedal y la materia orgánica acumulada en el mismo conducía a concluir que una parte importante de la acumulación de materia orgánica en el lecho se debía a la deposición de detritos de plantas. En el presente estudio ambos indicadores (conductividad hidráulica y porosidad drenable) son suficientemente sensibles a estas diferencias en los sólidos acumulados, si bien la conductividad hidráulica, por tener un rango de medidas más amplio evidencia mejor estos cambios.

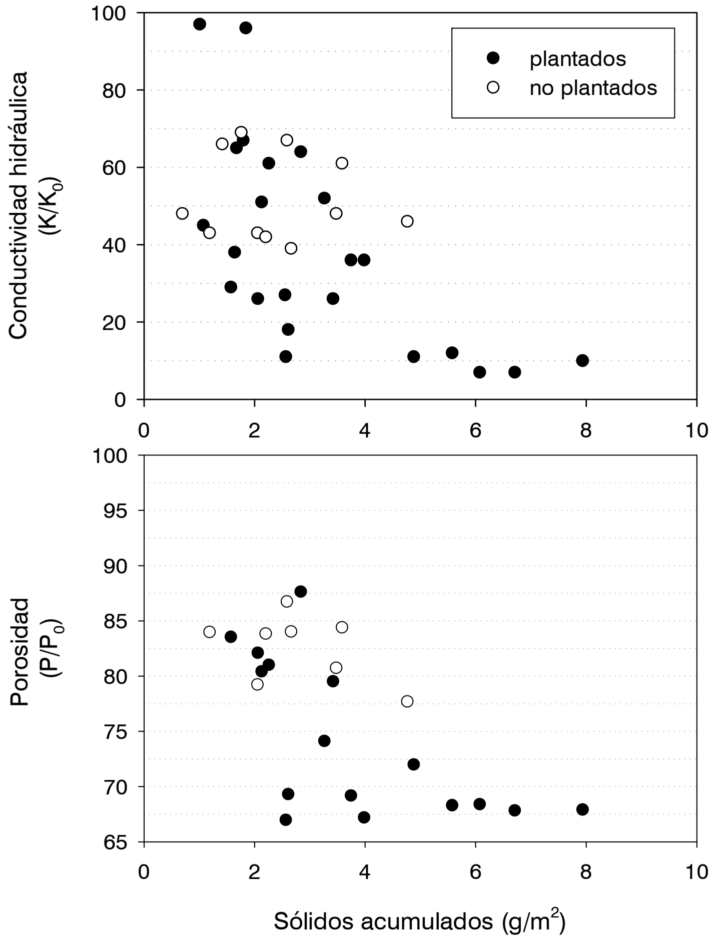

Figura 9.3 Conductividad hidráulica y porosidad drenable (en % respecto a los valores iniciales) en la entrada de un sistema de humedales a escala experimental en función de la acumulación de sólidos en la misma zona.

Aunque no sólo la cantidad de sólidos influye en la pérdida de conductividad hidráulica. El lodo acumulado presenta propiedades distintas en función de su naturaleza (Llorens et al., 2009). Como se vio en el Capítulo 4, los sólidos acumulados a la salida del humedal, aun siendo en cantidad similares a la entrada (en el caso del humedal de Corbins la acumulación de sólidos totales fue de 3.0-10 y 2.0-9.0 kg/m^2 en la entrada y la salida del humedal respectivamente), presentaban mayor decantabilidad. Así pues, en la entrada del humedal se acumulan preferentemente sólidos de carácter orgánico (del agua residual y correspondientes al biofilm) que producen un lodo gelatinoso que obtura más fácilmente los espacios intersticiales de la grava, dando lugar a mayor colmatación (y consecuentemente menor conductividad hidráulica, entre 0 y 20 m/d en la entrada del humedal frente a 30-70 m/d en la salida del lecho estudiado en Corbins) y a la aparición de charcos (Knowles et al., 2010b).

La porosidad drenable y la conductividad hidráulica aportan información acerca del comportamiento hidráulico del humedal y además se relacionan adecuadamente con la acumulación de sólidos (los coeficientes de correlación entre sólidos acumulados y porosidad, y sólidos y conductividad son 0.892 y 0.745 respectivamente). Sin embargo, la conductividad hidráulica permite evaluar de manera localizada la colmatación y por lo tanto nos describe la distribución de ésta en el humedal al realizar un mapaje (Knowles et al., 2010a; Capítulos 3 – 5). Por otro lado, la porosidad drenable no permite evaluar la distribución horizontal de la colmatación si bien se pueden realizar perfiles verticales. En la Figura 9.4 se muestra el perfil vertical de porosidad de los tres humedales de una de las líneas de la planta piloto de los Capítulos 6 y 7, y la distribución vertical de los sólidos acumulados, donde se observa que la pérdida de porosidad, más acusada en el fondo del humedal se debe fundamentalmente al crecimiento de las raíces en esta zona. En los humedales plantados, la acumulación de raíces en la parte inferior (entre 2-3 veces mayor que en la parte superior) se corresponde con una porosidad de aproximadamente un 25% (la porosidad inicial era del 40%). Esto es un 5% menos que la porosidad de la parte superior y la porosidad del humedal no plantado. En estos sistemas se observó que la penetración de las raíces era más importante que lo descrito generalmente en la literatura para sistemas a gran escala (Bowmer, 1987; Edwards, 1992; Brix, 1997), debido a que se trata de humedales someros (25 cm de lámina de agua frente a 60-70 cm en configuraciones convencionales).

Sin embargo, el ensayo de la porosidad implica el vaciado del humedal y la medida del volumen de drenaje, por lo que resulta de difícil aplicación en humedales construidos a gran escala. La conductividad hidráulica, de aplicación sencilla (como se ha discutido en la sección 9.1) y en distintas zonas se presenta como el indicador más adecuado para evaluar la colmatación en humedales a nivel práctico, mientras que la acumulación de sólidos nos ayuda a comprender más detalladamente las causas de este fenómeno.

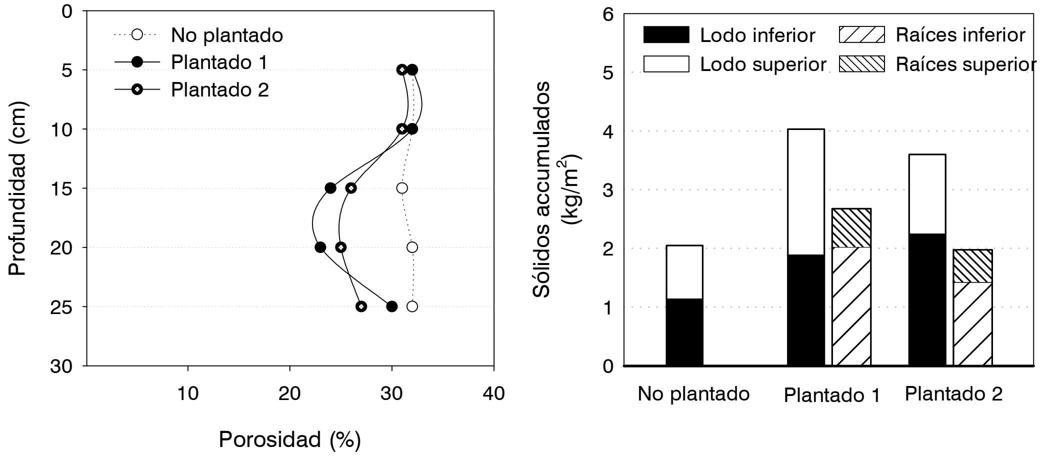

Figura 9.4 Perfil vertical de porosidad en los humedales de una línea en la planta experimental y los sólidos acumulados en la misma línea de tratamiento. La parte inferior de las raíces y lodo corresponde a los 10 cm del fondo, mientras que la parte superior corresponde a los 15 cm por encima.

9.3 Factores de diseño y operación de humedales construidos que afectan a la colmatación

A lo largo de los distinto Capítulos de la tesis han ido apareciendo factores ligados al diseño y operación de humedales que pueden afectar el desarrollo de la colmatación. En esta sección se discute acerca de estos factores, dando respuesta a los objetivos tercero y sexto del Capítulo 2.

Durante las diferentes campañas de campo realizadas en distintos humedales construidos a escala real (Capítulos 3, 4 y 5) se constataron patrones de conductividad hidráulica atribuibles al diseño y operación de los humedales. Se ha corroborado la relación entre la carga de sólidos y la acumulación de éstos en el medio granular, especialmente en la entrada del humedal (Figura 9.5). Así, la planta de tratamiento de Gualba (Capítulo 5), que no cuenta con un tratamiento primario del agua residual (sólo tiene un desbaste) presentaba un estado de colmatación más avanzado que Arnes, que lleva en funcionamiento 10 veces más tiempo pero cuenta con un tanque Imhoff como tratamiento primario. Es imprescindible, pues, tener en cuenta la carga orgánica y de sólidos en el diseño del humedal para evitar una sobrecarga que provoque la colmatación temprana del sistema (Tanner and Sukias, 1995; Tanner et al, 1998; Caselles-Osorio et al., 2007). La USEPA (2000) recomienda una carga superficial de sólidos que no exceda los 20 g/m^2d basándose en la eliminación de sólidos en suspensión. Sin embargo existen evidencias de colmatación en humedales operados a cargas inferiores, entre 2.6-10 g SST/m^2d (Caselles-Osorio et al., 2007) y 14 g/m^2d (Grismer et al., 2003). De acuerdo con los resultados obtenidos en este estudio y los que se hallan en la literatura (Tanner et al., 1998; Ruiz et al., 2010) parece suficiente operar los humedales horizontales con una carga de sólidos en la entrada inferior a 10 g/m^2d para limitar la tasa de acumulación de sólidos a unos 2 kg/m^2año.

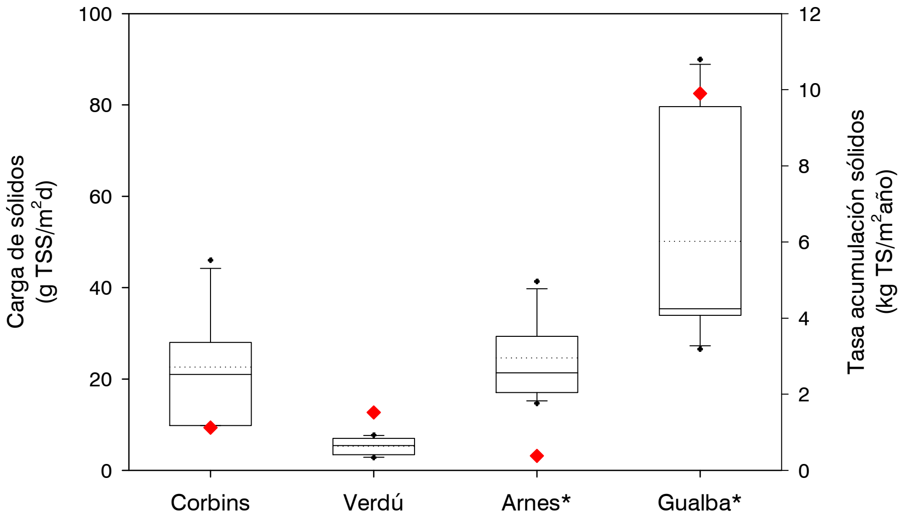

Figura 9.5 Carga superficial de sólidos aplicadas en distintos humedales de Catalunya (box plot) y tasa media de acumulación de sólidos (en rojo) (n≥ 6). * La carga de sólidos para los humedales de Arnes y Gualba se ha calculado en base al caudal de diseño (se desconoce el caudal real tratado).

Se ha observado la importancia de otros parámetros de diseño y operación en estos sistemas que afectan a la colmatación. Por un lado el sistema de distribución en la entrada del humedal. Algunos estudios han señalado el efecto de la posición tanto del distribuidor del agua como del colector a la salida en la aparición de caminos preferenciales del agua y zonas muertas dentro del sistema (Chazarenc et al., 2003; Suliman et al., 2006b), aunque con mayor influencia de la distribución en la entrada del humedal. Esto se vio tanto en el humedal de Gualba (Capítulo 3) como en Fenny Compton y Moreton Morrell (Capítulo 5). El agua toma el camino más directo entre el punto de entrada del agua y el de recogida, si se considera que las zonas de menor conductividad hidráulica son las que concentran la mayor parte del flujo de agua (Figura 9.6).

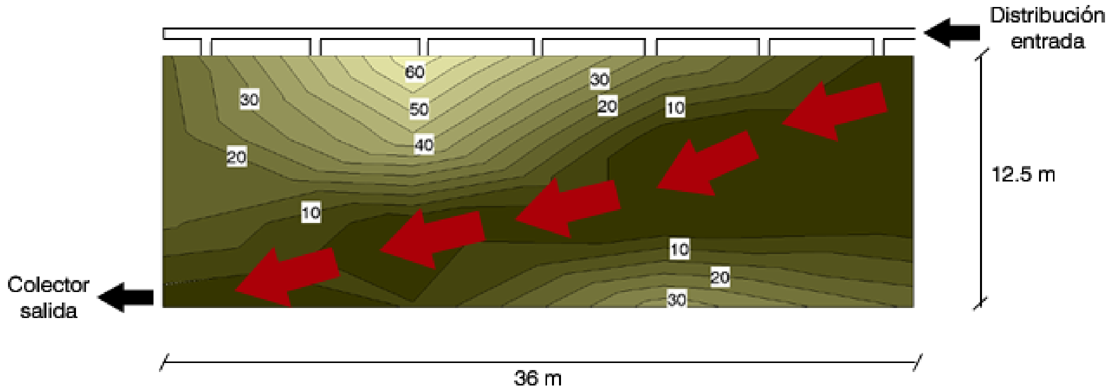

Figura 9.6 Distribución de la conductividad hidráulica medida con el permeámetro de carga variable (m/d) en un humedal con relación largo/ancho inferior a 1 y entrada y salida situadas en esquinas opuestas (Fenny Compton). Las flechas rojas indican el flujo preferencial del agua durante gran parte del funcionamiento.

Existen distintos tipos de sistemas de distribución usados en humedales horizontales. Las tuberías perforadas, dispuestas directamente sobre la superficie del lecho en la entrada, se asentan diferencialmente en el terreno, lo que repercute en zonas de mayor afluencia de agua, en las que la colmatación es más acusada. Un claro ejemplo lo encontramos en Arnes (Figura 9.7) donde la tubería de distribución del agua (con un punto de entrada por parcela) se ha asentado de modo que las celdas más alejadas del punto de entrada reciben un mayor caudal de agua. Consecuentemente, a medida que nos alejamos de éste, el avance de la colmatación es más importante. Tuberías verticales, como la que alimenta el humedal de Moreton Morrell, en Reino Unido (Capítulo 5) necesitan un mantenimiento regular para evitar la completa obturación de éstas. En ellas, la deposición de sólidos tanto del agua residual como de las plantas puede bloquear completamente la entrada del agua en el humedal. De hecho, en aquel lecho se observó que de los cuatro puntos de entrada de afluente sólo uno estaba operativo. En cambio, los canales de distribución, que además precisan de un mantenimiento más sencillo, son mucho más eficaces en la repartición del afluente a lo ancho del humedal (Griffin et al., 2008). Este tipo de distribución se está implementando tanto en Reino Unido como en Catalunya. Actualmente, los humedales de Corbins, Gualba y Verdú cuentan con canales para la distribución del afluente. Concretamente en Gualba (Capítulo 3) se observó, a pesar de los caminos preferenciales y el estado de colmatación avanzado, que la distribución del agua residual era visiblemente más uniforme que en sistemas con otros tipos de repartición. Conforme a lo estudiado en los Capítulos 3, 4 y 5, se recomienda el uso de canales para garantizar la buena distribución del agua en el ancho del humedal y prevenir, así, una colmatación diferencial que promueve la aparición de caminos preferenciales.

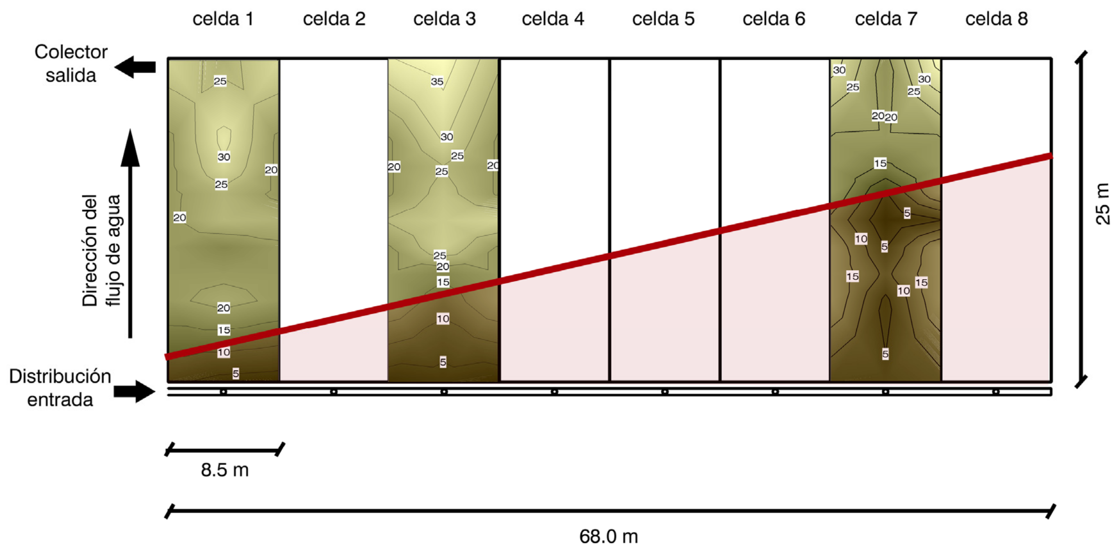

Figura 9.7 Distribución horizontal de la conductividad hidráulica (m/d) en tres parcelas de un humedal con distribución de agua mediante una tubería perforada (Arnes). En rojo se indica el avance de la colmatación en todo el lecho.

Por otro lado, se observó que la relación entre la anchura y la longitud del humedal tiene mucha influencia en la distribución espacial de la colmatación. Así, relaciones largo/ancho inferiores a 1 estarían desaconsejadas de acuerdo con los resultados obtenidos en el estudio de la conductividad hidraúlica en Gualba y Arnes (Capítulo 3) y Fenny Compton y Moreton Morrell (Capítulo 5). Eso se debe a que el agua fluye por caminos preferenciales (Figura 9.6) a través de las zonas de menor resistencia al flujo. Éstas coinciden con zonas donde el medio es más conductivo, ya sea por el tipo de grava o por la ausencia de raíces (Bowmer, 1987; Suliman et al., 2006b; Knowles et al, 2010a). Al aumentar la relación largo/ancho del lecho el agua se ve obligada a discurrir con un flujo más próximo al pistón, reduciéndose la dispersión del flujo y aumentando la eficiencia hidráulica (Jenkins and Greenway, 2005), y la distribución es más uniforme.

Relaciones largo/ancho superiores a 1, como las parcelas del humedal de Arnes (con una relación 3:1) obligan al agua a circular por el sistema ocupando todo el ancho del lecho (Figura 9.7). En el humedal de Arnes, a pesar de la pobre distribución del agua en la entrada (al considerar el humedal en su totalidad), la conductividad hidráulica aumenta desde la entrada a la salida de cada una de las celdas analizadas, como cabe esperar. Aparentemente el flujo en estas celdas no presenta caminos preferenciales o cortocircuitos transversales. Por ello, en los humedales en forma de canal (claramente más largos que anchos), por ser hidraúlicamente más eficientes, la colmatación se desarrolla de manera lógica desde la entrada hacia la salida más uniformemente. Esto supone una ventaja para el explotador en el caso de tener que realizar actuaciones para solventar un problema de colmatación, ya que puede determinar más fácilmente la extensión del problema y por lo tanto minimizar el presupuesto para remediarlo.

Finalmente un aspecto al que frecuentemente no se da especial importancia en el diseño de humedales construidos es el tipo de medio granular (García and Corzo, 2008). En el Capítulo 8 se comprobó que la composición mineral de la grava usada en el lecho está directamente relacionada con las propiedades de resistencia mecánica que ofrece. Así, el uso de gravas graníticas, cuya composición es básicamente cuarzo, mica y feldespato, dan como resultado gravas de grano anguloso (que disminuye en sí la conductividad hidráulica del medio – Griffin et al., 2008) y, según los porcentajes de cada uno de estos minerales, con tendencia a disgregarse. Las gravas más resistentes son aquellas compuestas eminentemente por cuarzo, si bien gravas calcáreas pero uniformes y redondeadas son preferibles al granito.

La calidad de la grava puede ser un factor importante en el desarrollo de la colmatación si no se escoge adecuadamente. Gravas como la del humedal de Gualba (Capítulos 3 y 8) pueden exacerbar el desarrollo de la colmatación. Durante el transporte y colocación de la grava en la fase de construcción del humedal se produce un desgaste abrasivo. Además, aunque el mantenimiento del humedal es sencillo, en muchos sistemas (al menos en España) se cosechan anualmente las plantas, usando maquinaria para facilitar la tarea (Turon et al., 2009). Esto puede suponer un desgaste mecánico en el medio granular. Teniendo en cuenta, además, que la grava está permanentemente en contacto con agua residual y expuesta a variaciones espaciales y temporales de pH (Kaseva, 2004; Wieβner et al., 2005), que pueden afectar a la durabilidad del material (Gupta and Ahmed, 2007), el uso de gravas fácilmente disgregables suponen un agente más de colmatación. En los humedales de Verdú y Corbins se analizó la composición mineral de la grava y del lodo acumulado en ambos humedales (Capítulo 4). Se comprobó que la composición mineral del lodo coincidía con la de la grava, lo que puede ser un síntoma de elevadas cantidades de finos en la grava (Caselles-Osorio et al, 2007) o bien de la desintegración de ésta. El caso de Gualba fue mucho más evidente. En este humedal la grava había sido parcialmente repuesta un año antes del análisis de la conductividad hidráulica. Los primeros 8 metros se repusieron en su totalidad mientras que en el resto del lecho se reemplazó la parte superficial (unos 20 cm). A pesar de la reposición de la grava el humedal presentó conductividades hidráulicas del mismo orden de magnitud que el humedal de Arnes (que llevaba en operación 10 años). Además el porcentaje de sólidos volátiles de muestras tomadas en algunos puntos del humedal fue muy bajo (entre 2-37%). Estos porcentajes difícilmente se pueden atribuir exclusivamente a la mineralización de la materia orgánica (Nguyen, 2000; Chazarenc and Merlin, 2005). La grava de este humedal parece contribuir de manera significativa al desarrollo de la colmatación prematura.

9.4 Nuevas configuraciones de humedales construidos en el desarrollo de la colmatación

Finalmente, en esta sección se discute acerca de la eficiencia de eliminación de contaminantes y el desarrollo y evolución de la colmatación en una planta piloto, dando respuesta a los objetivos cuarto y quinto del Capítulo 2.

En los Capítulos 6 y 7 se estudiaron dos nuevas configuraciones de sistemas de humedales, desde el punto de vista de la eficiencia de eliminación de contaminantes y del desarrollo de la colmatación. Para ello se construyó una planta experimental con tres líneas de tratamiento: 1) línea control (decantación – humedales construidos permanentemente inundados), 2) línea anaeróbica (HUSB - humedales construidos permanentemente inundados) y 3) línea batch (decantación – humedales construidos con alternancia de fases saturadas e insaturadas). Las tres líneas se operaron con la misma carga hidráulica (28 mm/d). El sistema de humedales tenía una profundidad de 30 cm (25 cm de columna de agua) y se alimentaron intermitentemente (14 L cada 4 horas). Se ha demostrado con anterioridad que estas dos características de diseño favorecen ambientes menos reducidos en el lecho, permitiendo alcanzar mayores eficiencias de eliminación que con diseños convencionales (humedales más profundos y operados en continuo, García et al., 2005; Caselles-Osorio and García, 2007).

La comparación entre las líneas control y anaeróbica permite determinar si un reactor HUSB resulta una buena alternativa a un tratamiento primario convencional (decantación), mientras que entre las líneas control y batch se evalúa un régimen hidráulico distinto al tradicional en humedales horizontales. Las hipótesis testadas se basan en que 1) un reactor HUSB como tratamiento primario elimina más eficientemente los sólidos en suspensión que una decantación (Álvarez et al. 2008b; Barros et al., 2008), y por lo tanto se reduce la carga superficial de sólidos aplicada en el humedal, retrasando los síntomas de colmatación; y 2) la operación de humedales combinando fases saturadas e insaturadas favorecería la hidrólisis de los sólidos acumulados gracias a una mayor aireación del lecho (Platzer and Mauch, 1997; Langergraber et al., 2003; Chazarenc et al., 2009).

Los humedales de las tres líneas de tratamiento recibieron una carga orgánica ligeramente menor a la de diseño, de 6 g DBO/m^2d, pero en el rango generalmente recomendado para este tipo de sistemas (Kadlec and Knight, 1996; García et al., 2005; Akratos and Tsihrintzis, 2007). Mientras que la eliminación de materia orgánica se mantuvo en las eficiencias esperadas (eficiencia de eliminación de DQO del 83%, 80% y 88% para las líneas control, anaeróbica y batch respectivamente), la eliminación de amonio (80%, 73% y 87% para las líneas control, anaeróbica y batch respectivamente) fue superior a los valores descritos generalmente para humedales horizontales (Austin and Nivala, 2009; Vymazal and Kröpfelová, 2009). Esto confirma que humedales someros alimentados intermitentemente eliminan la materia orgánica y el amonio más eficientemente.

La primera de las configuraciones estudiadas combina un reactor hidrolítico de flujo ascendente (HUSB) como tratamiento primario del agua residual y un sistema de humedales construidos de flujo subsuperficial horizontal operados de manera convencional (esto es, permanentemente inundados aunque con alimentación intermitente). Esta configuración ofrece ventajas a la tradicional (con tratamientos primarios basados en tratamientos físicos, como la decantación), puesto que permite reducir la carga de sólidos en la entrada del humedal (Ruíz et al., 2010). Los humedales experimentales de la línea con un tratamiento primario alternativo recibieron entre un 20 y un 35% menos carga de sólidos en suspensión que aquellos en que el agua pasaba por una mera decantación (Figura 9.8). Este factor, como hemos visto en secciones anteriores es uno de las principales variables causantes de colmatación en un lecho (Caselles-Osorio et al., 2007).

El uso de un reactor HUSB como tratamiento primario del agua, aunque no presentó diferencias en cuanto a reducción de porosidad drenable y conductividad hidraúlica con la línea de tratamiento control (operada de la misma manera pero con una decantación como tratamiento primario), acumuló un 30% menos de sólidos (Figura 9.8). A pesar de ser la conductividad hidráulica un buen indicador a nivel práctico, en estos sistemas las diferencias de sólidos acumulados (tanto cuantitativas como cualitativas) no fueron suficientemente importantes como para ser detectadas por estos indicadores hidráulicos.

Por lo tanto, limitando la carga superficial de sólidos mediante tratamientos primarios más eficaces en su eliminación se podría mitigar el desarrollo de la colmatación a largo plazo. Sin embargo, al tratarse de un reactor que opera en condiciones anaeróbicas, proporciona una agua más reducida que un decantador, lo que limita la eficiencia de eliminación de materia orgánica. Aún así, la profundidad del humedal puede, en parte, contrarrestar los efectos negativos del tratamiento primario. Como se ha comentado anteriormente la construcción de humedales someros (alrededor de 0.3 m de profundidad) permite mayores eliminaciones de contaminantes puesto que favorecen ambientes menos reducidos en el interior del lecho (García et al., 2004 y 2005). Por ello, la implantación de reactores anaeróbicos como tratamiento primario del agua residual constituye una alternativa a tener en cuenta para retrasar los efectos de la colmatación.

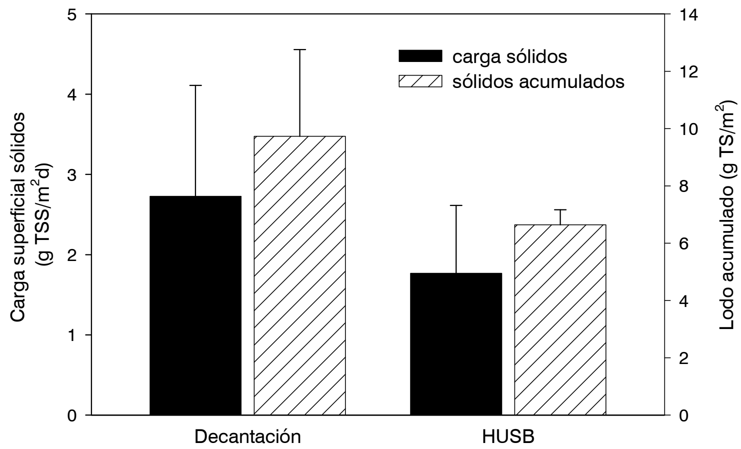

Figura 9.8 Carga superficial de sólidos en suspensión aplicada en humedales con tratamientos primarios distintos (decantación y reactor HUSB) y lodo acumulado en 3 años de operación.

La segunda configuración estudiada se basa en la implantación de un modo de operación distinto del comúnmente usado en humedales horizontales y que afecta al régimen hidráulico de éstos (línea batch). El uso de diferentes celdas operadas alternando fases saturadas e insaturadas del lecho permitió una mayor eficiencia en la eliminación de contaminantes, similar a la alcanzada por sistemas de humedales de flujo vertical (Brix and Arias, 2005). Mayor eficiencia especialmente acusada durante los meses fríos (un 50% más eficiente en la eliminación de amonio que la línea control), en los que estos sistemas reducen su eficiencia debido a las bajas temperaturas y a la ausencia de plantas activas (Akratos and Tsihrintzis, 2007; Hijosa-Valsero et al., 2010).

Sin embargo, en contra de lo previsto, esta configuración parece acumular más lodo en los espacios intersticiales de la grava. La hipótesis inicial de que la alternancia de fases saturadas e insaturadas puede favorecer la eliminación de sólidos acumulados debido a una mayor presencia de oxígeno (Platzer and Mauch, 1997; Langergraber et al., 2003) no se observó en la línea batch de la planta experimental (Figura 9.9). Al contrario, una ligera mayor acumulación de lodo en la línea plantada con la estrategia de operación alternativa puede ser una consecuencia de un mayor crecimiento de biofilm (Chazarenc et al, 2009). En cambio, en el humedal no plantado la acumulación de sólidos era menor que en su análogo permanentemente inundado. Esta situción puede estar provocada por el arrastre de sólidos durante las fases de drenaje del humedal.

A pesar de esta mayor acumulación de sólidos en la línea con alternancia de fases saturadas e insaturadas, el sistema radicular de las plantas se vio afectado, probablemente por estrés hídrico durante las fases insaturadas (Tanner et al., 1999), por lo que la acumulación total de sólidos (lodo y raíces) no fue significativamente distinta de la línea control.

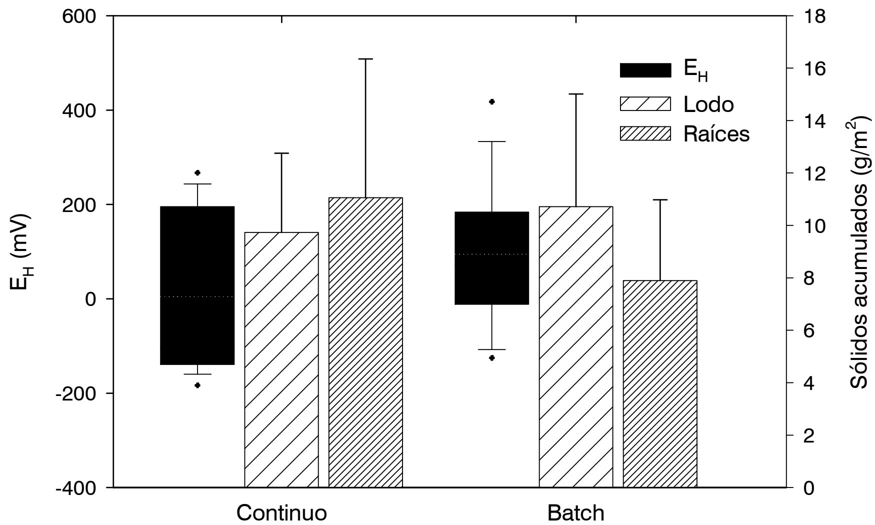

Figura 9.9 Potencial redox en el efluente de humedales con distinto régimen hidráulico (continuo y alternancia de fases saturadas e insaturadas) y sólidos acumulados (lodo y raíces) en la entrada de los sistemas después de 3 años de operación. Los datos de potencial redox corresponden a medidas cada 4h durante un mes en invierno y un mes en primavera.

Como ya se discutió en secciones anteriores, un agente que parece tener una no menospreciable contribución en la disminución de la conductividad hidráulica y de la porosidad del medio granular es el crecimiento de las plantas (Gersberg et al., 1986; Tanner, 1996; Brix, 1997; Muñoz et al., 2006). En ese sentido la operación alternando fases saturadas e insaturadas resulta ser una alternativa a la operación convencional de humedales horizontales puesto que aunque a corto plazo no se evidencian diferencias en la colmatación, sí lo hace en eficiencia de eliminación de contaminantes.

A la vista de los resultados obtenidos en la comparación de estas nuevas configuraciones de humedales parece lógico pensar que la combinación de un reactor HUSB como tratamiento primario del agua residual y humedales horizontales operados en batch podría ofrecer ventajas a las configuraciones estudiadas en esta tesis. Por un lado, porque el reactor HUSB disminuiría la carga superficial de sólidos aplicada al sistema de humedales, retrasando la colmatación. Por otro, porque la alternancia de fases saturadas e insaturadas permitiría aumentar la eficiencia de eliminación de contaminantes (fundamentalmente nitrógeno) al aumentar el potencial redox del sistema gracias a la aireación del medio. Sin embargo, el reactor HUSB produce efluentes más fácilmente biodegradables (con relaciones DBO_5/COD de 53% y 64% para los efluentes del decantador y el reactor HUSB, respectivamente) gracias a una mayor solubilización de la materia orgánica derivada de la actividad microbiológica del reactor. Si en el sistema de humedales operados en batch la acumulación de sólidos se debe, en parte, al crecimiento del biofilm, afluentes fácilmente biodegradables sumados a condiciones más aeróbicas en el lecho podrían favorecer este crecimiento de microorganismos y acelerar, por lo tanto, el desarrollo de la colmatación. Aún así, sería necesario operar estos sistemas durante un periodo de tiempo mayor para evaluar el efecto de estas configuraciones sobre la colmatación a largo plazo.

10

Conclusiones

Las conclusiones de este trabajo se sintetizan a continuación:

1 El permeámetro de carga variable para medir la conductividad hidráulica de un humedal de flujo subsuperficial horizontal es un método de sencilla aplicación que permite medir de manera fiable en el rango de valores que podemos encontrar en un humedal (0-500 m/d).

2 Se trata de un método preciso aunque presenta variaciones en los valores obtenidos respecto al permeámetro de carga constante (tanto en el laboratorio como en observaciones de campo *in situ*). En el rango de conductividades medidas *in situ* (0-70 m/d) se ha encontrado una relación potencial entre ambos métodos aplicados *in situ* (carga variable = carga constante$^{0.7821}$). Para rangos altos medidos en el laboratorio (>300 m/d) el permeámetro de carga variable ofrece valores un 20% superiores que el de carga constante. Estas discrepancias son aceptables en el rango de conductividades a medir.

3 Las discrepancias entre valores tomados en mediciones *in situ* con el permeámetro de carga variable y el de carga constante indican que el primero es un método menos sensible. Sin embargo la facilidad de aplicación lo hace adecuado para fines prácticos.

4 La medida de la conductividad hidráulica mediante el permeámetro de carga variable

permite obtener resultados fiables acerca del grado de colmatación en que se encuentra un humedal de flujo subsuperficial y de la distribución espacial de la misma. Esto permite identificar aspectos del diseño del humedal susceptibles de producir caminos preferenciales del agua residual en el lecho.

5 De los distintos indicadores utilizados para evaluar el grado de colmatación de un humedal (porosidad drenable, conductividad hidráulica, acumulación de sólidos y volumen efectivo mediante ensayos de trazadores), la conductividad hidráulica es el que ofrece más ventajas. En primer lugar porque aporta información acerca del comportamiento hidráulico en el interior del lecho y se correlaciona bien con la acumulación de sólidos (correlación de -0.745). En segundo lugar, porque su aplicación en humedales a gran escala es mucho más sencilla que la medida de la porosidad drenable. El uso de trazadores para determinar el volumen efectivo del humedal no es una buena opción para evaluar la colmatación, si bien es un buen indicador para determinar el flujo de agua en el interior del humedal.

6 El agua residual tiende a tomar el camino que menor resistencia ofrece al paso y que suele coincidir con el más corto entre el distribuidor de la entrada y el colector de salida del humedal. Humedales con relación largo/ancho menores a 1 desarrollan con facilidad caminos preferenciales y pueden presentar zonas muertas. El diseño de lechos con relaciones mayores a 1 fuerzan a que el agua circule por el humedal ocupando todo el ancho, por lo que se consigue una mejoría en el comportamiento hidráulico y consecuentemente una distribución más uniforme de la colmatación.

7 Del mismo modo afecta el diseño del sistema de distribución del agua en la entrada del humedal. Éste debe permitir una distribución homogénea en todo el ancho del humedal, por lo que se aconsejan canales de distribución y no el uso tuberías perforadas. Éstas, al asentarse diferencialmente en el terreno provocan zonas de mayor afluencia de agua, luego mayor colmatación. Además el mantenimiento de tuberías perforadas también es más difícil.

8 El tipo de grava utilizada en los humedales también puede ser un factor relacionado con el desarrollo de colmatación, sobre todo si se usan gravas fácilmente disgregables. Los áridos de naturaleza calcárea resultan menos resistentes a la abrasión y al desmoronamiento que aquellos de naturaleza granítica (con porcentajes de desmoronamiento del 0.18-1.03 y 2.42-4.56% para materiales calcáreos y graníticos, respectivamente). Sin embargo, siendo el grano uniforme y redondeado, son preferibles al granito, cuya com-

posición forma estructuras angulosas y fácilmente disgregables. La grava más recomendable es la compuesta eminentemente de cuarzo (> 80%) por sus cualidades de resistencia mecánica.

9 El test de abrasión de los Ángeles (LAA) es un buen indicador para determinar las propiedades mecánicas de la grava. De acuerdo con los valores obtenidos de este test se recomienda el uso de gravas con valores de LAA menores al 25%, o al menos que no excedan el 30%, independientemente de su composición mineral.

10 La carga superficial de sólidos se relaciona con la acumulación de éstos en el humedal, el uso de un reactor hidrolítico de flujo ascendente (HUSB) permite eliminar más eficazmente (un 30% más) la materia particulada del agua residual que tratamientos primarios convencionales (decantación). El uso de un reactor HUSB como tratamiento primario produce efluentes claramente más reducidos (-103±100 mV y 172±103 mV para los efluentes del reactor HUSB y el decantador respectivamente) y puede suponer una pequeña pérdida en la eficiencia de eliminación de contaminantes en el sistema (un 3% en eliminación de DQO y un 7% en eliminación de DBO_5 y amonio). A pesar de esta desventaja los humedales alimentados con el efluente del reactor HUSB presentaron una menor acumulación de sólidos en la entrada del humedal (un 30% menos de sólidos que el tratamiento control). Por lo tanto la configuración HUSB-humedal resulta una alternativa a tener en cuenta en el diseño de sistemas de tratamiento de agua basados en humedales construidos para retrasar la colmatación.

11 La alternancia de fases saturadas e insaturadas en la operación de un humedal permiten obtener mayores eficiencias de eliminación de contaminantes (un 5 y un 10% más eficiente en eliminación de DQO y amonio respectivamente). Esto se ve favorecido especialmente en los meses fríos, cuando la eficiencia de eliminación de materia orgánica y amonio puede llegar a ser un 50% mayor que en los humedales operados convencionalmente. Esta estrategia de operación produce una mayor acumulación de lodo en el lecho (un 10% más lodo acumulado en la línea batch que en la operación convencional) aunque más uniforme. Sin embargo la mayor acumulación de lodo se ve compensada con un menor crecimiento de plantas, y por lo tanto raíces, pudiendo mantener conductividades hidráulicas similares al tratamiento control.

12 Sería conveniente estudiar la combinación de un reactor HUSB como tratamiento primario del agua residual con un sistema de humedales operados con fases saturadas e insaturadas para evaluar el efecto cruzado de ambas configuraciones sobre la colmata-

ción.

A partir de las conclusiones obtenidas en esta tesis se pueden extraer las siguientes recomendaciones técnicas:

1 Es muy recomendable el uso de tratamientos primarios del agua residual eficaces para disminuir la carga de sólidos aplicada al humedal (limitándola a unos 10 g SST/m^2d), lo que puede retrasar en gran medida el desarrollo de la colmatación.

2 El sistema de distribución del afluente más adecuado es aquel en forma de canal puesto que permite una distribución más uniforme en la entrada, evitando la aparición de caminos preferenciales.

3 Se recomienda construir los humedales horizontales con relaciones largo/ancho superiores a 1 para aumentar la eficiencia hidráulica del sistema y conseguir así un flujo más próximo al pistón.

4 Es importante utilizar un tipo de grava uniforme, de grano redondeado y mecánicamente resistente. En este sentido se recomienda el uso de gravas con un porcentaje de cuarzo superior al 80%, o en su defecto, con valores por debajo del 25% en el test de abrasión de Los Ángeles.

5 La construcción de humedales someros aumenta la eficiencia de eliminación de contaminantes, así como el uso de estrategias de operación que favorezcan la aireación del medio granular. En este sentido es necesario ahondar en estrategias como la operación alternando fases saturadas e insaturadas para maximizar los rendimientos de eliminación sin acarrear mayores costes de operación.

6 El mantenimiento de los sistemas de tratamiento basados en humedales construidos es de suma importancia para retrasar los síntomas de colmatación en los lechos. Por ello se recomienda mantener los sistemas de distribución afluente y de recogida del efluente en buen estado y cosechar las plantas anualmente y eliminar los detritos de plantas de la superficie del humedal.

11

Referencias bibliográficas

Adcock, P.W. and Ganf, G.G. (1994). Growth characteristics of three macrophyte species growing in a natural and constructed wetland system. Water Science and Technology, 29(4), 95-102.

Admon S., Tarre S., Sabbah I., Lahav O., Green M., 2002. Treatment of pre-settled municipal wastewater using a passively aerated vertial bed. Mbwette T.S.A. (ed.). Proceedings of the 8th International Conference on Wetland Systems for Water Pollution Control, 16-19 September 2002; IWA Publishing and University of Dar Es Salaam: Arusha, Tanzania, pp. 615-626.

AENOR (1999). Test for general properties of aggregates. Part 2: Methods for reducing laboratory samples. UNE-EN 932-2. Madrid (in Spanish).

AENOR (1998a). Test for geometrical properties of aggregates. Part 1: determination of particle size distribution. Sieving method. UNE-EN 933-1. Madrid (in Spanish).

AENOR (1998b). Test for chemical properties of aggregates. Part 1: Chemical analysis. UNE-EN 1744-1. Madrid (in Spanish).

AENOR (1998c). Tests for mechanical and physical properties of aggregates. Part 2: Methods for the determination of resistance to fragmentation. UNE-EN 1097-2. Madrid (in Spanish).

AENOR (1994). Geotechnique. Compactation test. Standard Proctor. UNE-EN 103500. Madrid (in Spanish).

Agència Catalana de l'Aigua (ACA), 2010. L'Agència. Memòria 2009. http://aca-web.gencat.cat/aca/documents/ca/publicacions/memoria2009/memoria_aca_2009.pdf (consultado en junio 2010).

Akratos, C.S., Tsihrintzis, V.A., 2007. Effect of temperature, HRT, vegetation and porous media on removal efficiency of pilot-scale horizontal subsurfaceflow constructed wetlands. Ecological Engineering 29(2), 173–191.

Albuquerque, A., Oliveira, J., Semitela, S., Amaral, L., 2009. Influence of bed media characteristics on ammonia and nitrate removal in shallow horizontal subsurface flow constructed wetlands. Bioresource Technology 100, 6269-6277.

Álvarez, J.A., Ruíz, I., Soto, M., 2008a. Anaerobic digesters as a pretreatment for constructed wetlands. Ecological engineering 33(1), 54-67.

Álvarez, J.A., Armstrong, E., Gómez, M., Soto, M., 2008b. Anaerobic treatment of low-strength municipal wastewater by a two-stage pilot plant under psychrophilic conditions. Bioresource Technology 99(15), 7051-7062.

APHA-AWWA-WPCF, 2001. *Standard Methods For The Examination of Water and Wastewater (20th ed.)*. American Public Health Association,Washington DC, USA.

Arias, C.A., Brix, H., 2005. Phosphorus removal in constructed wetlands: can suitable alternative media be identified? Water Science and Technology 51(9), 267-273.

Arnold, P.W., 1995. *Manual of Soil Laboratory Testing. Vol. 2. Permeability, Shear strength and Compressibility tests, 2nd ed*. Pentech Press, London, 1994. 440 pp.

Austin, D., Nivala, J., 2009. Energy requirements for nitrification and biological nitrogen removal in engineered wetlands. Ecological Engineering 35(2), 184-192.

Bagarello, V., Elrick, D.E., Iovino, M., Sgroi, A., 2006. A laboratory analysis of falling head infiltration procedures for estimating the hydraulic conductivity of soils. Geoderma 135, 322-334.

Bagarello, V., Iovino, M., Elrick, D.E., 2004. A simplified falling-head technique for rapid determination of field-saturated hydraulic conductivity. Soil Science Society of America Journal 68(1), 66–73.

Barros, P., Ruíz, I., Soto, M., 2008. Performance of an anaerobic digester-constructed wetland system for a small community. Ecological engineering 33(2), 142-149.

Batchelor A., Loots P., 1997. A critical evaluation of a pilot scale subsurface flow wetland: Ten years after commissioning. Water Science and Technology 35(5): 337-343.

Bavor H.J., Schulz T.J., 1993. Sustainable suspended solids and nutrient removal in large-scale, solid-matrix, constructed wetland systems. In: *Constructed Wetlands for Water Quality Improvement*, Moshiri G.A. (ed.), Lewis Publishers, Boca Raton, Florida, pp. 646-656.

Bécares, E., 2004. Effect of vegetation and design processes in horizontal subsurface flow and free water surface constructed wetlands. In: García, J., Morató, J. Bayona, J.M. (Eds) *New Criteria for Design and Operation of Constructed Wetlands: A Low-Cost Alternative for Wastewater Treatment*. Barcelona Ed. CPET, Barcelona, Spain. pp. 7–16 (in Spanish).

Behrends, L.L., Bailey, E., Houke, L., Jansen, P., Smith, S., 2006. Non-invasive methods for treating and removing sludge from subsurface flow constructed wetlands II. In: 10th International Conference on Wetland systems for Water Pollution Control, Lisbon, Portugal, pp. 1271–1281.

Behrends, L.L., Houke, L., Bailey, E., Jansen, P., Brown, D., 2001. Reciprocating constructed wetlands for treating industrial, municipal and agricultural wastewater. Water science and Technology 44(11-12), 399-405.

Behrends, L.L., Coonrod, H.S., Bailey, E. and Bulls, M.J., 1993. Oxygen diffusion rates in reciprocating rock biofilters: potential applications for subsurface constructed wetlands. In: Proceedings Subsurface Flow Constructed Wetlands Conference. August 16-17. University of Texas at El Paso.

Bezbaruah, A.N., Zhang, T.C., 2005. Quantification of oxygen release by bulrush (*Scirpus* *validus*) in a 457 constructed treatment wetland. Biotechnology and Bioengineering 89(3), 308-318.

Blazejewski, R., Murat-Blazejewska, S., 1997. Soil clogging phenomena in constructed wetlands with subsurface flow. Water Science and Technology 35(5), 183–188.

Boletín Oficial del Estado (BOE) (2008). Structural concrete instruction. RD1247/2008, BOE num. 203, Madrid (in Spanish).

Bowmer, K.H., 1987. Nutrient removal from effluents by artificial wetland: Influence of rhizosphere aeration and preferential flow studied using bromide and dye tracers. Water research 21(5), 591-599.

Breen, P.F., 1997. The performance of vertical flow experimental wetlands under a range of operational formats and experimental conditions. Water Science and Technology 35(5), 167-174.

Brix, H., Arias, C.A., 2005. The use of vertical flow constructed wetlands for on-site treatment of domestic wastewater: New Danish guidelines. Ecological Engineering 25(5), 491-500.

Brix, H., Arias, C.A., Del Bubba, M., 2001. Media selection for sustainable phosphorus removal in subsurface flow constructed wetlands. Water Science and Technology 44(11-12), 47-54.

Brix, H., 1997. Do macrophytes play a role in constructed treatment wetlands? Water Science and Technology 35(5), 11-17.

Calheiros, C.S.C., Duque, A.F., Moura, A., Henriques, I.S., Correia, A., Rangel, A.O.S.S., Castro, P.M.L., 2009. Substrate effect on bacterial communities from constructed wetlands planted with Typha latifolia treating industrial wastewater. Ecological Engineering 35(5), 744-753.

Carter, M.R., Ball, B.C., 1993. Soil porosity. In: Carter, M.R. (Ed.) *Soil sampling and methods of analysis*. CRC Press. Boca Raton, Fl., USA, pp. 581-588.

Caselles-Osorio, A., Puigagut, J., Segú, E., Vaello, N., Granés, F., García, D., García, J., 2007. Solids accumulation in six full-scale subsurface flow constructed wetlands. Water Research 41(6), 1388–1398.

Caselles-Osorio, A., García, J., 2007a. Impact of continuous and intermittent feeding strategies on the performance of shallow horizontal subsurface-flow constructed wetlands. Science of the total Environment 378, 253-262.

Caselles-Osorio, A., García, J., 2007b. Effect of physico-chemical pre-treatment on the removal efficiency of experimental horizontal subsurface-flow constructed wetlands. Environmental Pollution 146, 55-63.

Caselles-Osorio, A., García, J., 2006. Performance of experimental horizontal subsurface flow constructed wetlands fed with dissolved or particulate organic matter. Water Research 40(19), 3603-3611.

Centro de Estudios y Experimentación de Obras Públicas (CEDEX) (1991). Determination of durability to collapse of soft rocks. NTL-251/91. Madrid (in Spanish).

Chazarenc, F., Gagnon, V., Comeau, Y., Brisson, J., 2009. Effect of plant and artificial aeration on solids accumulation and biological activities in constructed wetlands. Ecological Engineering 35, 1005-1010.

Chazarenc F., Maltais-Landry G., Troesch S., Comeau Y., Brisson J., 2007. Effect of loading rate on performance of constructed wetlands treating an anaerobic supernatant. Water Science and Technology 56(3): 23-29.

Chazarenc F., Merlin G., 2005. Influence of surface layer on hydrology and biology of gravel bed vertical flow constructed wetlands. Water Science and Technology 51(9): 91-97.

Chazarenc, F., Merlin, G., Gonthier, Y., 2003. Hydrodynamics of horizontal subsurface flow constructed wetlands. Ecological Engineering 21, 165-173.

Cheug, K.C., Venkitachalam, T.H., 2000. Improving phosphate removal of sand infiltration system using alkaline fly ash. Chemosphere 41, 243-249.

Cooper, P., 2009. What can we learn from old wetlands? Lessons that have been learned and some that may have been forgotten over the past 20 years. Desalination 246, 11-26.

Cooper D., Griffin P., Cooper P.F., 2008. Factors affecting the longevity of sub-surface horizontal flow systems operating as tertiary treatment for sewage effluent. In: *Wastewater Treatment, Plant Dynamics and Management in Constructed and Natural*

Wetlands, Vymazal J. (ed.), Springer, Dordrecht, Netherlands, pp. 191-198.

Cooper, P., 2005. The performance of vertical flow constructed wetland systems with special reference to the significance of oxygen transfer and hydraulic loading rates. Water Science and Technology 51(9), 81-90.

Cooper, D.J., Griffin, P., Cooper, P.F., 2005. Factors affecting the longevity of subsurface horizontal flow systems operating as tertiary treatment for sewage effluent. Water Science and Technology 51(9), 127-135.

Cooper P.F., 2003. Sizing vertical flow and hybrid constructed wetland systems. En: The Use of Aquatic Macrophytes for Wastewater Treatment in constructed Wetlands, 1st International Seminar. Dias V. y Vymazal J. (eds.), Instituto Nacional da Água, Lisbon, Portugal, pp. 195-218.

Council of the European Communities, 1991. Council Directive of 21 May 1991 concerning urban wastewater treatment (91/271/EEC). Official Journal of the European Communities, No. L 135/40-52 (30 May).

Crites, R., Tchobanoglous, G., 1998. *Small and decentralized wastewater menagement systems*. McGraw-Hill Inc. Boston, USA, 1084 pp.

Davies, L.C., Pedro, I.S, Novais, J.M, Martins-Dias, S., 2006. Aerobic degradation of acid orange 7 in a vertical constructed wetland. Water Research 40(10), 2055-2063.

Dhakal, G., Yoneda, T., Kato, M., Kaneko, K., 2002. Slake durability and mineralogical properties of some pyroclastic and sedimentary rocks. Engineering Geology 65, 31-45.

Díaz M.A., Ruiz I., García J., 2008. Solids hydrolysis and accumulation in an anaerobic digetser-constrcuted wetland system. Proceedings of the 11th International Conference on Wetland Systems for Water Pollution Control, 1-7 November 2008; Institute of Environment Management and Plant Sciences, Vikram University and IWA: Indore, India, pp. 205-211.

Dornelas F.L., Machado M.B., von Sperling M., 2008. Performance evaluation of planted and unplanted subsurface-flow constructed wetlands for the post-treatment of UASB reactor effluents. Proceedings of the 11th International Conference on Wetland Systems for Water Pollution Control, 1-7 November 2008; Institute of Environment Management and

Plant Sciences, Vikram University and IWA: Indore, India, pp. 400-407.

Drizo, A., Frost, C. A., Grace, J., Smith, K. A., 2000. Phosphate and ammonium distribution in a pilot-scale constructed wetland with horizontal subsurface flow using shale as a substrate. Water Research 34(9), 2483-2490.

Drury, W.J., Mainzhausen, K., 2000. Hydraulic Characteristics of Subsurface Flow Wetlands. Billings Land Reclamation Symposium. Billings, MT March 20–24.

Dušek, J., Picek, T., Čížková, H., 2008. Redox potential dynamics in a horizontal subsurface flow constructed wetland for wastewater treatment: Diel, seasonal and spatial fluctuations. Ecological Engineering 34(3), 223-232.

Edwards, K.R., Čižková, H., Zemanová, K., Šantrůčková, H., 2006. Plant growth and microbial processes in a constructed wetland planted with Phalaris arundinacea. Ecological Engineering 27(2), 153-165.

Edwards, G.S., 1992. Root distribution of soft-stem bulrush (Scirpus validus) in a constructed wetland. Ecological Engineering 1(3), 239–243.

Elrick, D.E., Angulo-Jaramillo, R., Fallow, D.J., Reynolds, W.D., Parkin, G.W., 2002. Analysis of infiltration under constant head and falling head conditions. P. 47-53. In: P.A.C. Raats et al. (ed.) *Environmental mechanics: water, mass and energy transfer in the biosphere*. Geophysical Monograph Series, vol. 129, AGU, Washington, DC.

Elrick, D.E., Reynolds, W.D., 1992. Methods of analyzing constant-head well permeameter data. Soil Scinece Society of America Journal 56, 320-323.

Faulwetter, J.L., Gagnon, V., Sundberg, C., Chazarenc, F., Burr, M.D., Brisson, J., Camper, A.K., Stein, O.R., 2009. Microbial processes influencing performance of treatment wetlands: A review. Ecological Engineering 35(6), 987-1004.

Fisher P.J., 1990. Hydraulic characteristics of constructed wetlands at Richmond, New South Wales, Australia. In: *Constructed Wetlands in Water Pollution Control*, Cooper P.F., Findlater B.C. (eds.), Pergamon Press, Oxford, United Kingdom, pp. 21-32.

García, J., 2004. Constructed wetlands for pollution control: perspective on a technology in expansion. In: García, J., Morató, J., Bayona, J.M. (Eds.), *New Criteria for Design and*

Operation of Constructed Wetlands: A Low-Cost Alternative for Wastewater Treatment. Ediciones CPET, Barcelona, Spain, pp. 7–16 (in Spanish).

García, J., Corzo, A., 2008. *Water treatment with constructed wetlands: Handbook of design, construction and operation of subsurface flow wetlands Systems*. Universitat Politècnica de Catalunya, Barcelona, Spain. http://upcommons.upc.edu/e-prints/bitstream/2117/2474/1/JGarcia_and_ACorzo.pdf (accessed 15 May 2010) (in Spanish).

García, J., Domingo, V., 2006. Reutilización de aguas depuradas para usos ambientales en Granollers. Tecnología del Agua 263, 62-68.

García; J., Rousseau, D.P.L., Morató, J. Lesage, E., Matamoros, V.; Bayona, J.M., 2010. Contaminant Removal Processes in Subsurface-Flow Constructed Wetlands: A Review. Critical Reviews in Environmental Science and Technology 40: 7, 561 — 661

Garcia, J., Rousseau, D., Caselles-Osorio, A., Story, A., De Pauw, N., Vanrolleghem, P., 2007. Impact of prior physico-chemical treatment on the clogging process of subsurface flow constructed wetlands: Model-based evaluation. Water, Air and Soil Pollution 185(1-4), 101-109.

García, J., Aguirre, P., Barragán, J.,Mujeriego, R., Matamoros, V., Bayona, J.M., 2005. Effect of key design parameters on the efficiency of horizontal subsurface flow constructed wetlands: long-term performance pilot study. Ecological Engineering 25(4), 405–418.

García, J., Aguirre, P., Mujeriego, R., Huang, Y., Ortiz, L., Bayona, J.M., 2004. Initial contaminant removal performance factors in horizontal flow reed beds used for treating urban wastewater. Water Research 38(7), 1669–1678.

García, J., Ojeda, E., Sales, E., Chico, F., Píriz, T., Aguirre, P., Mujeriego, R., 2003. Spatial variations of temperature, redox potential, and contaminants in horizontal flow reed beds. Ecological Engineering 21(2-3), 129–142.

Gersberg, R.M., Elkins, B.V., Lyon, S.R., Goldman, C.R., 1986. Role of aquatic plants in wastewater treatment by artificial wetlands. Water Research 20(3), 363-368.

Gonilho Pereira, C., Castro-Gomes, J., Pereira de Oliveira, L., 2009. Influence of natural coarse aggregate size, mineralogy and water content on the permeability of structural concrete.

Construction and Building Materials 23, 602-608.

Griffin, P., Wilson, L., Cooper, D., 2008. Changes in the use, operation and design of sub-srface flow constructed wetlands in a major UK water utility. 11th International Conference on Wetland Systems for Water Pollution Control. Indore, India, 419-426.

Grismer, M.E., Carr, M.A., Shepherd, H.L., 2003. Evaluation of constructed wetland treatment performance for winery wastewater. Water Environment Research 75(5), 412-421.

Grubb, D.G., Guimaraes, M.S., Valencia, R., 2000. Phosphate immobilisation using an acidic type fly ash. Journal of Hazardous Materials 73(2-3), 217-236.

Gupta, V., Ahmed, I., 2007. The effect of pH of water and mineralogical properties on the slake durability (degradability) of different Rocks from the Lesse Himalaya, India. Engineering Geology 95, 79-87.

Hijosa-Valsero, M., Matamoros, V., Sidrach-Cardona, R., Martín-Villacorta, J., Bécares, E., Bayona, J.M., 2010. Comprehensive assessment of the design configuration of constructed wetlands for the removal of pharmaceuticals and personal care products from urban wastewaters. Water Research 44(12), 3669-3678.

Huang, Y., Ortiz, L., Aguirre, P., García, J., Mujeriego, R., Bayona, J.M., 2005. Effect of design parameters in horizontal flow constructed wetland on the behaviour of volatile fatty acids and volatile alkylsulfides. Chemosphere 59, 769-777.

Huang, Y., Latorre, A., Barceló, D., García, J., Aguirre, P., Mujeriego, R., Bayona, J.M., 2004. Factors affecting linear alkylbenzene removal in subsurface flow constructed wetlands. Environmental Science and Technology 38, 2657-2663.

Jenkins, G.A., Greenway, M., 2005. The hydraulic efficiency of fringing versus banded vegetation in constructed wetlands. Ecological Engineering 25, 61-72.

Jiménez Salas, J.A., De Justo Alpañes, J.L., 1975. *Geotécnia y Cimientos I, 2ª Ed.*, Editorial Rueda, Madrid.

Johnson, R.B., DeGraff, J.V., 1988. *Principles of engineering geology*, Wiley, New York, 497 pp.

Kadlec, R.H., 2009. Comparison of free water and horizontal subsurface treatment wetlands.

Ecological Engineering 35(2), 159-174.

Kadlec, R.H., Wallace, S.D., 2009. *Treatment Wetlands, Second Edition*. CRC Press. Boca Raton, Florida, USA, pp. 1016.

Kadlec R.H., Knight R.L., Vymazal J., Brix H., Cooper P. y Haberl, R., 2000. *Constructed Wetlans for Pollution Control: Processes, Performance, Design and Operation.* IWA Specialist Group on use of Macrophytes in Watwe Pollution Control, IWA Publishing, 156 pp.

Kadlec, R.H., Knight, R.L., 1996. *Treatment Wetlands*. CRC Press Boca Raton, 893 pp

Kadlec, R.H., Watson, J.T., 1993. Hydraulics and solids accumulation in a gravel bed treatment wetland. In: Moshiri, G.A. (Ed.), *Constructed Wetlands For Water Quality Improvement*. Lewis Publishers, Boca Raton, Florida, pp. 227–235.

Kalkan, E., 2009. Influence of silica fume on the dessication cracks of compacted clayey soils. Applied Clay Science 43(3–4), 296–302.

Kaseva, M.E., 2004. Performance of a sub-surface flow constructed wetland in polishing pre-treated wastewater—a tropical case study. Water Research 38, 681-687.

Knowles, P.R., Griffin, P., Davies, P.A., 2010a. Complementary methods to investigate the development of clogging within a horizontal sub-surface flow tertiary treatment wetland. Water Research 44(1), 320-330.

Knowles, P., Dotro, G., Nivala, J., García, J., 2010b. Clogging in subsurface-flow treatment wetlands: occurrence, contributing factors and Management strategies. Ecological Engineering, submitted.

Knowles, P.R., Davies, P.A., 2009. A method for the in-situ determination of the hydraulic conductivity of gravels as used in constructed wetlands for wastewater treatment. Desalination and Water Treatment 1(5), 257–266.

Knowles, P.R., Morris, R.H., Pedescoll, A., Newton, M.I., Davies, P.A., Bencsik, M., 2009. Analysis of the clog matter within a mature horizontal subsurface flow constructed wetland. Proceedings of the 3rd Wetland Pollutant Dynamics and Control (WETPOL2009). Barcelona, Spain, 131-132.

Knowles, P.R., Griffin, P., Davies, P.A., 2008. An investigation into development of heterogeneous hydraulic conductivity profiles in Constructed Wetlands for Wastewater treatment. In: 11th International Conference on Wetland Systems for Water Pollution Control, Indore, India, pp. 814–824.

Kuschk, P., Wieβner, A., Kappelmayer, U., Weiβbrodt, E., Kästner, M., Stottmeister, U., 2003. Annual cycle of nitrogen removal by a pilot-scale subsurface horizontal flow in a constructed wetland under moderate climate. Water Research 37, 4236-4242.

Langergraber, G., Haberl, R., Laber, J., Pressl, A., 2003. Evaluation of substrate clogging processes in vertical flow constructed wetlands. Water Science and Technology 48(5), 25-34.

Levenspiel, O., 1999. *Chemical reaction engineering. 3rd ed*. John Wiley and Sons, Inc. Hoboken, USA. 668pp.

Ligero, P., Vega, A., Soto, M., 2001. Pretreatment of urban wastewaters in a hydrolytic upflow digester. Water SA 27(3), 399-404.

Llorens, E., Puigagut, J., García, J., 2009. Distribution and biodegradability of sludge accumulated in full-scale horizontal subsurface-flow constructed wetland. Desalination and water treatment 4, 54-58.

Mastrorilli M., Ranieri E., Simeone V., 2001. Evaluation of hyraulic conductivity in a Phragmites wastewater treatment plant. Progress in Water Resources: 105-111.

Matamoros, V., García, J., Bayona, J.M., 2005. Behavior of selected pharmaceuticals in subsurface flow constructed wetlands. A pilot scale study. Environmental Science and Technology 39, 5449-5454.

Metcalf & Eddy., 2003. *Wastewater Engineering: Treatment and Reuse, 4th edition*. McGraw-Hill, Boston, USA.

Ministerio de medio Ambiente, Medio Rural y Marino (MARM), 2007. Plan Nacional de calidad de Agua: saneamiento y depuración 2007-2015. http://www.mma.es/secciones/medios_comunicacion/prensa/notas_pre/2007/06/PlanNacionalCalidadAguas.pdf (consultado en mayo de 2010).

Molle, P., Liénard, A., Boutin, C., Merlin, G., Iwema, A., 2005. How to treat raw sewage with constructed wetlands: an overview of the French systems. Water Science and Technology 51(9), 11-21.

Muñoz, P., Drizo, A., Cully Hession, W., 2006. Flow patterns of dairy wastewater constructed wetlands in a cold climate. Water research 40, 3209-3218.

Murphy C., Cooper D., 2009. The evolution of horizontal sub-surface flow reed bed design for tertiary treatment of sewage effluents in the U.K. Proceedings of the 7th Workshop on Nutrient Cycling and Retention in Natural and Constructed Wetlands, 21-24 April 2009; ENKI: Trebon, Czech Republic.

Murphy, C., Cooper, D., Williams, E., 2009. Reed bed refurbishment: A sustainable approach. 3rd International Symposium on Wetland Pollutant Dynamics and Control, WETPOL 2009, 20-24 September 2009. Barcelona, Spain.

NAVFAC, 1986. Soil Mechanics. Design Manual 7.01. Naval Facilities Engineering Command, Alexandria, Virginia, USA, 389 pp.

Nguyen, L.M., 2000. Organic matter composition, microbial biomass and microbial activity in gravel-bed constructed wetlands treating farm dairy wastewaters. Ecological Engineering 16(2), 199–221.

Nivala, J., Rosseau, D.P.L., 2009. Reversing clogging in subsurface flow constructed wetlands by hydrogen peroxide treatment: two case studies. Water Science and Technology 59(10), 2037-2046.

Nivala, J., Rosseau, D.P.L., 2008. Reversing clogging in subsurface flow constructed wetlands by hydrogen peroxide treatment: two case studies. In: 11th International Conference on Wetland Systems forWater Pollution Control, Indore, India, pp. 383–391.

O'Brien, R., Kent Keller, C., Strobridge, D.M., 2004. Plant-cover effects on hydrology and pedogenesis in a sandy vadose zone. Geoderma 118, 63–76.

Pacheco Torgal, F., Castro-Gomes, J.P., 2006. Influence of physical and geometrical properties of granite and limestone aggregates on the durability of a C20/25 strength class concrete. Construction and Building Materials 20, 1079-1088.

Parr, T.W. (1990). Factors affecting reed (phragmites australis) growth in UK reed bed treatment systems. In: Constructed wetlands and water pollution control. Proceedings of the international conference on the use of constructed wetlands in water pollution control. Adv. Wat. Pollut. Control no. 11., P.F. Cooper and B.C. Findlater (Eds). Pergamon Press, Oxford, pp. 383-389.

Pedescoll, A., Samsó, R., Romero, E., Puigagut, J., García, J., 2010. Reliability, accuracy and repeatability of the falling head method for hydraulic conductivity measurements under laboratory conditions. Ecological Engineering, doi: 10.1016/j.ecoleng.2010.06.032.

Pedescoll, A., Uggetti, E., Llorens, E., Granés, F., Garcia, D., García, J., 2009. Practical method based on saturated hydraulic conductivity used to assess clogging in subsurface flow constructed wetlands. Ecological Engineering, 35(8), 1216–1224.

Platzer, C., Mauch, K., 1997. Soil clogging in vertical flow reed beds—mechanisms, parameters, consequences and . . . solutions? Water Science and Technology 35(5), 175–181.

Puigagut, J., Villaseñor, J., Salas, J.J., Bécares, E., García, J., 2007. Subsurface-flow constructed wetlands in Spain for the sanitation of small communities: a comparative study. Ecological Engineering 30(4), 312-319.

Ragusa S.R., McNevin D., Qasem S., Mitchell C., 2004. Indicators of biofilm development and activity in constructed wetlands microcosms. Water Research 38: 2865-2873.

Ranieri, E., 2003. Hydraulics of sub-superficial flow constructed wetlands in semi arid climate conditions. Water Science and Technology 47(9), 49-55.

Reynolds, W.D., Bowman, B.T., Brunke, R.R., Drury, C.F., Tan, C.S., 2000. Comparison of tension infiltrometer, pressure infiltrometer, and soil core estimates of saturated hydraulic conductivity. Soil Science Society of America Journal 64, 478–484.

Reynolds, W.D., Zebchuk, W.D., 1996. Hydraulic conductivity in a clay soil: two measurement techniques and spatial characterization. Soil Science Society of America Journal 60, 1679-1685.

Robusté, J., 2004. Constructed wetlands in operation, experience in Catalonia. In: García, J., Morató, J., Bayona, J.M. (Eds.), *New Criteria for Design and Operation of Constructed*

Wetlands: A Low-Cost Alternative for Wastewater Treatment. Barcelona Ed. CPET, Barcelona, Spain. pp. 7–16 (in Spanish).

Rodgers, M., Mulqueen, J., 2006. Field-saturated hydraulic conductivity of unsaturated soils from falling-head well tests. Agricultural Water Management 79(2), 160–176.

Rousseau, D.P.L., Lesage, E., Story, A., Vanrolleghem, P.A., De Pauw, N., 2008. Constructed wetlands for water reclamation. Desalination 218(1-3), 181–189.

Rousseau, D.P.L., Horton, D., Vanrolleghem, P.A., De Pauw, N., 2005. Impact of operational maintenance on the asset life of storm reed beds. Water Science and Technology 51(9), 243-250.

Rousseau, D.P.L., Vanrolleghem, P.A., De Pauw, N., 2004. Model-based design of horizontal subsurface flow constructed treatment wetlands: a review. Water Research 38(6), 1484-1493.

Rowe, K.R., Armstrong, M.D., Cullimore, D.R., 2000. Particle size and clogging of granular media permeated with leachate. ASCE Journal of Geotechnical and Geoenvironmental Engineering 126(9), 775-786.

Ruiz, I., Díaz, M.A., Crujeiras, B., García, J., Soto, M., 2010. Solids hydrolysis and accumulation in a hybrid anaerobic digester-constructed wetlands system. Ecological Engineering 36(8), 1007-1016.

Ruiz, I., Alvarez, J.A, Diaz, M.A, Serrano, L., Soto, M., 2008. Municipal wastewater treatment in an anaerobic digester constructed wetland system. Environmental Technology 29(11), 1249 — 1256

Salas, J.J., Pidre, J.R., Sánchez, L., 2007.Manual de tecnologías no convencionales para la depuración de aguas residuales. Capítulo IV: Humedales artificiales. Coria Gráfica, Sevilla, Spain.

Samsó, R., Pedescoll, A., García, J., 2009. Clogging distribution in two horizontal subsurface flow constructed wetlands through measurements of hydraulic conductivity. Proceedings of the 3rd Wetland Pollutant Dynamics and Control (WETPOL2009). Barcelona, Spain, 127-128.

Sandford, W.E., Steenhuis, T.S., Parlange, J.Y., Surface, J.M., Peverly, J.H., 1995. Hydraulic conductivity of gravel sand as substrates in rock-reed filters. Ecological Engineering 4(4), 321–336.

Sasikala, S., Tanaka, N., Wah Wah, H.S.Y.,. Jinadasa, K.B.S.N., 2009. Effects of water level fluctuation on radial oxygen loss, root porosity, and nitrogen removal in subsurface vertical flow wetland mesocosms. Ecological Engineering 35(3), 410-417.

Seguí, L., Alfranca, O., García, J., 2009. Techno-economical evaluation of water reuse for wetland restoration: a case study in a Natural Parc in Catalonia, Northeastern Spain. Desalination 246, 179-189.

Serrano, L., De La Varga, D., Díaz, M.A., Ruiz, I., Bondelle, R., Soto, M., 2009. HUSB-CW treatment system for winery wastewater. Proceedings of the 5th International Specialized Conference on Sustainable Viticulture: Winery waste and Ecologic Impacts Management, 317-324. Trento and Verona, Italy. March 30th-April 3rd 2009.

Shepherd, H., Grismer, M.E., Tchobanoglous, G., 2001. Treatment of high-strength winery wastewater using a subsurface-flow constructed wetland. Water Environment Research 73(4), 394-403.

Singh, S., Haberl, R., Moog, O., Shrestha, R.R., Shrestha, P., Shrestha, R., 2009. Performance of an anaerobic baffled reactor and hybrid constructed wetland treating high-strength wastewater in Nepal—A model for DEWATS. Ecological Engineering 35(5), 654-660.

Song, H-L., Nakano, K., Taniguchi, T., Nomura, M., Nishimura, O., 2009. Estrogen removal from treated municipal effluent in small-scale constructed wetland with different depth. Bioresource Technology 100, 945-2951.

Stein, O.R., Hook, P.B., Biederman, J.A., Allen, W.C., Borden, D.J., 2003. Does batch operation enhance oxidation in subsurface constructed wetlands? Water Science and Technology 48(5), 149-156.

Suliman, F., French, H.K., Haugen, L.E., Svik, A.K., 2006a. Change in flow and transport patterns in horizontal subsurface flow constructed wetlands as a result of biological growth. Ecological Engineering 27(2), 124–133.

Suliman, F., Futsaether, C., Oxaal, U., Haugen, L.E., Jenssen, P., 2006b. Effect of the inlet –

outlet positions on the hydraulic performance of horizontal subsurface-flow wetlands constructed with heterogeneous porous media. Contaminant Hydrology 87, 22-36.

Tanner, C.C., Kadlec, R.H., Gibbs, M.M., Sukias, J.P.S., Nguyen, M.L., 2002. Nitrogen processing gradients in subsurface-flow treatment wetlands—influence of wastewater characteristics. Ecological Engineering 18(4), 499-520.

Tanner, C.C., 2001. Plants as ecosystem engineers in subsurface-flow treatment wetlands. Water Science and Technology 44(11-12), 9-17.

Tanner, C.C., D'Eugenio, J., McBride, G.B., Sukias, J.P.S., Thompson, K., 1999. Effect of water level fluctuation on nitrogen removal from constructed wetland mesocosms. Ecological Engineering 12(1-2), 67-92.

Tanner, C.C., Sukias, J.P.S., Upsdell, M.P., 1998. Organic matter accumulation and maturation of gravel-bed constructed wetlands treating dairy farm wastewaters. Water Research 32(10), 3046-3054.

Tanner, C.C., 1996. Plants for constructed wetland treatment systems — A comparison of the growth and nutrient uptake of eight emergent species. Ecological Engineering 7(1), 59-83.

Tanner, C.C, Sukias, J.P., 1995. Accumulation of organic solids in gravel bed constructed wetlands. Water Science and Technology 32(3), 229-239.

Tchobanoglous, G., 2003. Preliminary treatment in constructed wetlands. In: Proceedings of the 1st International Seminar on the Use of Aquatic Macrophytes for Wastewater Treatment in Constructed Wetlands, 8-10 May 2003, Dias V., Vymazal J. (eds.); Instituto da Conservacao da Natureza and Instituto da Agua: Lisbon, Portugal, pp. 13-33.

Tietz A., Kirschner A., Langergraber G., Sleytr K., Haberl R., 2007. Characterisation of microbial biocenosis in vertical subsurface flow constructed wetlands. Science of the Total Environment 380: 163-172.

Trang, N.T.D, Konnerup, D., Schierup, H.H., Chiem, N.H., Tuan, L.A., Brix, H., 2010. Kinetics of pollutant removal from domestic wastewater in a tropical horizontal subsurface flow constructed wetland system: Effects of hydraulic loading rate. Ecological Engineering 36(4), 527-535.

Turon, C., Comas, J., Poch, M., 2009. Constructed wetland clogging: a proposal for the integration and reuse of existing knowledge. Ecological Engineering 35, 1710-1718.

Tyroller, L., Rousseau, D.P.L., Santa, S., García, J., 2010. Application of the gas tracer method for measuring oxygen transfer rates in subsurface flow constructed wetlands. Water Research 44(14), 4217-4225.

U.S. EPA, 2000. Constructed wetlands treatment of municipal wastewaters, EPA 625/R-99/010, U.S. EPA Office of Research and Development: Washington D.C.

U.S. EPA, 1993. Design manual no. 74: Subsurface flow constructed wetland for wastewater treatment: A technology assessment, EPA 832/R-93/008, U.S. EPA Office of Water.

U.S. EPA, 1983. Design manual. Municipal wastewater stabilization ponds, EPA-625/1-83-015, Cincinnati, Ohio.

Ugur, I., Demirdag, S., Yavuz, H., 2010. Effect of rock properties on the Los Angeles abrasión and impact test characteristics of the aggregates. Materials characterization 61, 90-96.

Vohla, C., Alas, R., Nurk, K., Baatz, S., Mander, Ü., 2007. Dynamics of phosphorus, nitrogen and carbón removal in a horizontal subsurface flow constructed wetland. Science of the Total Environment 380, 66-74.

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

Vymazal, J., Kröpfelová, L., 2009. Removal of organics in constructed wetlands with horizontal sub-surface flow: A review of the field experience. Science of The Total Environment 407, 3911-3922.

Vymazal, J., 2005. Horizontal sub-surface flow and hybrid constructed wetlands systems for wastewater treatment. Ecological Engineering 25(5), 478–490.

Vymazal, J., Masa, M., 2003. Horizontal sub-surface flow constructed wetland with pulsing water level. Water Science and Technology 48(5), 143-148.

Vymazal, J., 2002. The use of sub-surface constructed wetlands for wastewater treatment in the Czech Republic: 10 years experience. Ecological Engineering 18(5), 633-646.

Wallace, S.D., Knight, R.L., 2006. Small-scale Constructed Wetland Treatment Systems. In: *Feasibility, Design Criteria and O&M Requirements. Final Report, Project 01-CTS-5*, Water Environment Research Foundation (WERF). Alexandria, USA.

Watson J.T., Choate K.D., 2001. Hydraulic conductivity of onsite constructed wetlands. Mancl K. (ed.). Proceedings of the Ninth National Symposium on Individual and Small Community Sewage Systems; American Society of Agricultural Engineers: St. Joseph, Michigan, pp. 632-649.

Wieβner, A., Kappelmeyer, U., Kuschk, P._ Kästner, M., 2005. Influence of the redox condition dynamics on the removal efficiency of a laboratory-scale constructed wetland. Water Research 39, 248-256.

Wilson, M.A., Hoff, W.D., Brown, R.J.E., Carter, M.A., 2000. A falling head permeameter for the measurement of the hydraulic conductivity of granular solids. Review of Scientific Instruments 71(10), 3942-3946.

Yagüe, A., Valls, S., Vázquez, E., Albareda, F., 2005. Durability of concrete with addition of dry sludge from wastewater treatment plants. Cement Concrete Res. 35(6), 1064–1073.

Youngs, E. G., Leeds-Harrison, P. B., Elrick, D. E., 1995. The hydraulic conductivity of low permeability wet soils used as landfill lining and capping material: analysis of pressure infiltrometer measurements. Soil Technology 8(2), 153-160.

Zhao Y.Q., Sun G., Lafferty C., Allen S.J., 2006. Optimising the performance of a lab-scale tidal flow reed bed system treating agricultural wastewater. Water Science and Technology 50(8): 65-72.

Zhao, J., 1998. Rock mass hydraulic conductivity of the Bukit Timah granite, Singapore. Eng. Geol. 50, 211–216.

Zhu, T., Jenssen, P.D., Mæhlum, T., Krogstad, T., 1997. Phosphorus sorption and chemical characteristics of Lightweight Aggregates (LWA) – potential filter media in treatment wetlands. Water Science and Technology 35(5), 103-108.

Agraïments

Ignorant. Així em sento avui, quina paradoxa. Després de quatre anys acumulant i processant dades se suposa que tinc més coneixements i, per contra només me n'adono de tot el que em falta per saber. És això? Com més n'aprens menys en saps? Aleshores, com més saps més ximple et tornes? Això no es el que havíem pactat. I el cas és que en aquest camí cap a la ignorància m'hi ha ajudat molta gent. Soc bilingüe, des de petita em vaig acostumar a canviar del català al castellà i viceversa segons amb qui parlava, així que a partir d'ara aquest text serà un caos idiomàtic, incapaç com sóc de dirigir-me a la gent en una altra llengua d'aquella en què els vaig conèixer (quasi sempre).

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Moltes gràcies!

Curriculum vitae

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Articles included in JRC

Ávila, C., Pedescoll, A., Matamoros, V., Bayona, J.M., García, J., 2010. Capacity of a horizontal subsurface flow constructed wetlands system for the removal of emerging pollutants: an injection experiment. Chemosphere, doi:10.1016/j.chemosphere.2010.08.006.

Pedescoll, A., Samsó, R., Romero, E., Puigagut, J., García, J., 2010. Reliability, accuracy and repeatability of the falling head method for hydraulic conductivity measurements under laboratory conditions. Ecological Engineering, doi: 10.1016/j.ecoleng.2010.06.32.

Pedescoll, A., Uggetti, E., Llorens, E., Granés, F., García, D., García, J., 2009. Practical method based on saturated hydraulic conductivity used to assess clogging in subsurface flow constructed wetlands. Ecological Engineering 35(8), 1216-1224.

Uggetti, E., Llorens, E., Pedescoll, A., Ferrer, I., Castellnou, R., García, J., 2009. Sludge drying reed beds: a cae study. Journal of Residual Science and Technology 6, 57-59.

Pedescoll, A., Knowles, P.R., Davies, P., García, J. and Puigagut, J. A comparison of in situ constant and falling head permeameter tests to assess the distribution of clogging within horizontal subsurface flow constructed wetlands. Science of the Total Environment, submitted.

Pedescoll, A., Corzo, A., Álvarez, E., Puigagut, J., García, J. Improving contaminant removal according to primary treatment and and operation strategy in horizontal subsurface flow constructed wetlands. Ecological Engineering, submitted.

Pedescoll, A., Corzo, A., Álarez, E., Puigagut, J., García, J. The effect of primary treatment and flow regime on clogging development in horizontal subsurface flow constructed wetlands: an experimental evaluation. Bioresource Technology, submitted.

Pedescoll, A., Passos, F., Alba, E., Puigagut, J., García, J. Mechanical resistance properties of gravel used in subsurface flow constructed wetlands: implications for clogging. Water Science and Technology, submitted.

Hijosa-Valsero, M., Matamoros, V., Pedescoll, A., Martín-Villacorta, J., Bécares, E., García, J., Bayona, J.M., 2010. Evaluation of primary treatment and loading regimes in the removal of pharmaceuticals and personal care products from urban wastewaters by subsurface-flow constructed wetlands. International Journal of Environmental Analytical Chemistry, under revision.

Articles published in technical journals

Pedescoll, A., Granés, F., García, D., Ramírez, G., García, J., 2009. Nivel de clmatación en dos humedales construidos de flujo subsuperficial a gran escala. Tecnología del Agua 307, 34-40.

Conference contributions

Pedescoll, A., Passos, F., Alba, E., Puigagut, J., García, J. Gravel composition related to clogging development in horizontal subsurface flow constructe wetlands. 12th IWA International Conference on Wetland Systems for Water Pollution Control. Venice, Italy, 4-9 October 2010

(oral communication).

Pedescoll, A., Corzo, A., Álvarez, E., Puigagut, J., García, J. Contaminant removal and clogging development in shallow subsurface flow constructed wetlands: effect of primary treatment and operating strategy. 12th IWA International Conference on Wetland Systems for Water Pollution Control. Venice, Italy, 4-9 October 2010 (oral communication).

Pedescoll, A., Puigagut, J., García, J. Batch operation to improve pollutants removal efficiency in subsurface flow constructed wetlands. IWA National Young Water Profesional Conference. Barcelona, Spain, 16-18 June 2010 (poster).

Pedescoll, A., Álvarez, E., García, J. Strategies to improve pollutants removal efficiency in shallow subsurface flow constructed wetlands. 3rd Wetland Pollutants Dynamics and Control (WETPOL2009). Barcelona, Spain, 20-24 September 2009 (oral communication).

Samsó, R., Pedescoll, A., García, J. Clogging distribution in two horizontal subsurface flow constructed wetlands through measurements of hydraulic conductivity. 3rd Wetland Pollutants Dynamics and Control (WETPOL2009). Barcelona, Spain, 20-24 September 2009 (oral communication).

Knowles, P.R., Morris, R.H., Pedescoll, A., Newton, M.I., Davies, P.A., Bencsik, M. Analysis of the clog matter within a mature horizontal subsurface flow constructed wetland. 3rd Wetland Pollutants Dynamics and Control (WETPOL2009). Barcelona, Spain, 20-24 September 2009 (oral communication).

Adrados, B., Bécares, E., Codony, F., Fittipaldi, M., Pedescoll, A., Morató, J. Microbial removal in different constructed wetlands configurations. 3rd Wetland Pollutants Dynamics and Control (WETPOL2009). Barcelona, Spain, 20-24 September 2009 (oral communication).

Ávila, C., Pedescoll, A., Matamoros, V. Bayona, J.M., García, J. Assessment of PPCPs removal in pilot-scale subsurface flow constructed wetland system: an injection experiment. 3rd Wetland Pollutants Dynamics and Control (WETPOL2009). Barcelona, Spain, 20-24 September 2009 (poster).

Pedescoll, A., Knowles, P.R.. Clogging in constructed wetlands. Measuring hydraulic conductivity *in situ*. Specialist workshop on Constructed Wetland Clogging. Birmingham, UK., 4-5 March 2009 (oral communication).

Pedescoll, A., Corzo, A., Álvarez, E., García, J. Tratamiento primario y modos de operación en la colmatación de humedales construidos de flujo subsuperficial. Reunión de la Mesa Española de Tratamiento de Aguas (META2008). Puerto de la Cruz, Spain, 3-6 December 2008 (oral communication).

Corzo, A., Pedescoll, A., Álvarez, E., García, J. Solids accumulation and drainable porosity in experimental subsurface flow constructed wetlands with different primary treatments and operating strategies. 11th International Conference on Wetland Systems Technology in Water Pollution – control. Indore – Ujjain, India, 1-7 November 2008 (oral communication).

Pedescoll, A., Corzo, A., Álvarez, E., García, J. Influence of pevious treatments and operating mode on the efficiency of shallow subsurface flow constructed wetlands. 11th International Conference on Wetland Systems Technology in Water Pollution – control. Indore – Ujjain, India, 1-7 November 2008 (poster).

Pedescoll, A., Uggetti, E., Llorens, E., Puigagut, J., García, J., García, D., Granés, F. Linking inirect measures to clogging phenomena in full-scale subsurface flow constructed wetlands. II International Congress on Wastewater Treatment in Small Communities (SMALLWAT2007). Sevilla, Spain, 11-15 November 2007 (oral communication).