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Effluent treatment in trout aquaculture, state of the art and further developments

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Effluent treatment in trout aquaculture, state of the art and further developments

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Zusammenfassung

Das Ziel der vorliegenden Arbeit ist einen Überblick über die Ablaufwasserbelastung aus Forellenteichanlagen zu geben sowie Möglichkeiten der Ablaufwasserreinigung aufzuzeigen. In einer Vorstudie in bayerischen Forellenteichanlagen wurden die Faktoren ermittelt, die die Nährstofffracht im Ablauf der Fischzuchten bedingen. Dabei wird die Nährstoffkonzentration im Ablaufwasser, neben der Zulaufkonzentration, durch die Fütterungsintensität, die Art der Haltungseinrichtungen sowie die Effektivität der Ablaufwasserreinigung bestimmt.

Aufbauend auf diesen ersten Ergebnissen, wurden in einer Literaturstudie potentielle Reinigungsmöglichkeiten evaluiert, die für Forellenteichanlagen geeignet sind. Anhand dieser Literaturdaten wurden Reinigungskonzepte für Forellenteichanlagen entwickelt, die an die jeweilige Art der Haltungseinrichtung und die Produktionsintensität angepasst sind. Die Konzepte wurden anschließend auf ihre Effektivität und Wirtschaftlichkeit hin überprüft. Pflanzenkläranlagen wurden dabei als die Reinigungsmethode mit dem höchsten Potential, allerdings auch mit dem größten Forschungsbedarf beurteilt.

Demzufolge wurden in einer ersten Versuchsanordnung sechs von acht identischen Absetzbecken zu vertikal durchströmten Pflanzenbeeten umgewandelt und für die Reinigung des gesamten Ablaufwassers einer extensiv betriebenen Fischzuchtanlage erprobt. Dabei wurde die Reinigungsleistung der Pflanzenkläranlage mit dem ursprünglichen Absetzbecken verglichen. Die Pflanzenkläranlage übertraf die Effektivität des Absetzbeckens sowohl während der Behandlung des Durchlaufwassers als auch während Phasen der Teichreinigung um ein Vielfaches. Es wurden mit der Pflanzenkläranlage Reinigungsleistungen von bis zu 87 % für gelöste und partikuläre Nährstoffe gemessen, während das Absetzbecken eine maximale Entnahme von bis zu 45 % aufwies.

Nach Abschluss dieses ersten sehr erfolgreichen Versuches, wurden die sechs Pflanzenbeete für die Behandlung von Ablaufwasser aus intensiver Forellenproduktion erprobt. Dabei wurde die Reinigungsleistung jedes einzelnen der sechs Pflanzenbeete, getrennt über einen Zeitraum von 12 Monaten betrachtet. Jeweils zwei der sechs Beete erhielten die gleichen Zulaufmengen, was einer jeweils identischen hydraulischen Belastung und Nährstofffracht entsprach. Durch diese Versuchsanordnung konnten die Faktoren, die die Reinigungsleistung der Pflanzenbeete beeinflussen statistisch ermittelt werden. Die wichtigsten Faktoren mit starken Effekten auf die Reinigungsleistung waren die Nährstoffkonzentration im Zulauf der Pflanzen-

beete, die hydraulische Belastung der Beete sowie die Anreicherung von abfiltrierbaren Stoffen im Wurzelraum der Pflanzenbeete. Dabei ist die Anreicherung von Partikeln im Wurzelraum der Beete der einzige zeitabhängige Faktor, der Einfluss auf die Reinigungsleistung hat. Andere Faktoren wie Abfischungen oder die Vegetationsperiode waren von untergeordneter Bedeutung für die Effektivität.

Drei Faktoren sind demnach für die Reinigungsleistung einer Pflanzenkläranlage verantwortlich: 1. die Zulaufkonzentration zur Pflanzenkläranlage, die vornehmlich durch die Produktionsintensität der Fischzucht bestimmt wird, 2. die hydraulische Belastung der Pflanzenkläranlage, die verantwortlich ist für den Flächenbedarf der Anlage und 3. die Akkumulation von Partikeln im Wurzelraum, die die Lebensdauer der Anlage bestimmt. Durch die Ermittlung dieser drei Faktoren können die Kosten für die erfolgreiche Ablaufwasserbehandlung in Pflanzenkläranlagen ermittelt werden.

Pflanzenkläranlagen sind als alleinige Reinigungstechnik nur zur Behandlung von Ablaufwasser aus extensiven Fischzuchtanlagen geeignet. Bei intensiver Fischproduktion sollte vor Pflanzenkläranlagen ein effektiver mechanischer Vorfilter, wie z. B. ein Trommelfilter vorgeschalten werden. Diese Kombination gewährleistet eine sehr effektive Entnahme von gelösten und partikulären Nährstoffen aus dem Ablaufwasser von Fischzuchtanlagen. Zudem verursacht diese Kombination Kosten von weniger als 10 % des Produktionspreises, eine Investition, die im Sinne einer nachhaltigen Fischproduktion angemessen ist.

Summary

The present work aimed to identify the actual effluent problems from flow-through trout aquaculture and to offer possible effluent treatment strategies. Starting from a preliminary study, the effluent treatment need and the factors influencing effluent nutrient concentration from trout farming were identified. The main factors turned out to be, beneath the farm inflow concentration, the production intensity, the type of rearing units used and the effluent treatment device applied.

With this preliminary information, the effluent treatment methods, potentially suitable for trout aquaculture were identified from the literature. The different methods were compared for their suitability, treatment efficiency and treatment costs. With this information an effluent treatment scheme for flow-through trout aquaculture was developed in dependence on the rearing units used and the production intensity applied. Additionally, suitable effluent treatment methods, where profound research is needed for a successful application in aquaculture, like constructed wetlands, were identified.

Thus subsurface flow (SSF) constructed wetlands were tested for the treatment of the whole trout farm effluent.

In a first study six identical sedimentation basins were transformed to SSF wetlands, applied for the treatment of the effluents from a low intensive trout farm. Treatment efficiency of the newly build wetlands was compared to the initial sedimentation basin, during “normal” operation and during pond cleaning situation. In both situations, the SSF wetland cells performed much better than the initial sedimentation basin and reached significant treatment efficiencies of up to 87 % for particulate and dissolved nutrient fractions, compared to maximum treatment efficiencies of only 45 % for the sedimentation basin.

Subsequently, the six SSF wetland cells were used for the treatment of the effluent from intensive trout farming. The treatment efficiency of each cell of this already established wetland was surveyed for 12 month under a constant production intensity. Always two cells received the same hydraulic load, thus the factors influencing wetland treatment efficiency under intensive production conditions could be identified. The most important factors were the nutrient concentration in the wetland inflow, the hydraulic load on the wetland and the accumulation of total suspended solids (TSS) to the wetland root zone, the only time dependent factor.

Factors like fish harvesting or the vegetation period were of minor importance for the wetland treatment efficiency.

With these main factors: inflow nutrient concentration, which is dependent on the production intensity in the farm, the hydraulic load to the wetland, responsible for the area need, and the accumulation of TSS in the root zone filter, predicting the service lifetime of the wetland; the costs needed for successful SSF wetland treatment could be estimated.

As a stand alone effluent treatment device SSF wetlands are suitable only for low production intensity. With intensive trout production, SSF wetlands should be combined with effective micro-screen pre treatment. This combination is highly suitable for dissolved and particulate nutrient polishing. Also the financial suitability is given, as the trout production costs increase by less than 10 %, an expense fisheries managers and consumers should find justifiable when nutrient emission is kept to a minimum.

General Introduction

1. Trout aquaculture

Aquaculture is the science and technology of producing aquatic plants and animals (Lawson 1995), while an aquacultural production system can be described simply as production of marketable aquatic organisms under controlled or semi controlled conditions (Wheaton & Singh 1999).

Classification of aquaculture systems are based, among other, on the construction type of the rearing units (pond, net pen, raceway or tank-based), the species reared, the intensity of production, the culture water salinity, the culture water temperature, the farming technology or the use of the water supply (Wheaton & Singh 1999, Lekang 2007). The term trout aquaculture used here refers to the land based cold-freshwater production of fishes from the family of salmonidae, especially *Oncorhynchus mykiss*, *Salmo trutta*, *Salvelinus* spp. and hybrids from these species.

Trout production takes place in intensive culture systems, as defined by Milden & Redding (1998), with a continuous water flow, to maintain an acceptable water quality, the application of high quality artificial feeds to provide the nutritional requirements of fish, and a closely system monitoring. The only remaining biological production limitations are the available water flow, and the tolerance of the cultured species to crowding (Milden & Redding 1998).

After Lekang (2007), the major components of an intensive farm are:

- Water inlet and transfer
- Water treatment facilities
- Production units
- Feeding equipment
- Equipment for internal fish transport and size grading
- Equipment for transport of fish from the farm
- Instrumentation and monitoring systems
- Equipment for waste and wastewater treatment

With these components of an intensive fish farm, a general description of typical trout production units has the following characteristics: The inlet water is taken from rivers or brooks where a dam provides the needed water gradient for supply. Spring or mine water are also frequently used. As water treatment in most instances aeration or oxygenation is used, further prior water treatment is site dependent. The inflow water is generally used once as flow-through without water reuse. The production is realized mainly in raceways or tanks, but also in raceway shaped ponds with natural embankments but relative high water exchange rates (Pillay 1993). The water residence time in the rearing unit is usually in the range of minutes (Lawson 1995). The common feeding equipment in trout farms include a feed storage and a feed application system. The feed application system can be by hand or in any degree of automation. The equipment for fish transport and grading are always farm specific. The instrumentation and monitoring system depends on the specific farm intensity. E.g. when technical oxygenation is used, also an oxygen control and alarming system is needed to prevent fish losses. And finally, the equipment needed for waste and wastewater treatment for the trout production units, which is the central theme of this thesis.

Trout farming began in the nineteenth century in Europe following the development of artificial trout fertilization (Laird & Needham 1988, Tidemand-Johannessen 1999). From these beginnings, the production of rainbow trout (*Oncorhynchus mykiss* Walbaum) has reached 215,207 mt in the European union in 2003 (European Commission 2006) and is one of the most important finfish species cultured in Europe, the USA, Canada and Chile (Fornshell 2002). The trout producing sector is dominated by small regionally rooted micro enterprises, with an annual production of 100 mt or even less (MacAlistar Elliot and Partners 1999, Varadi 2001, FAO 2003, Klinkhard 2004, Engle et al. 2005).

2. Trout aquaculture in southern Germany

Against the background of the European water framework directive, and an increasing demand on aquaculture goods (FAO 2007), the German trout producers are faced with stronger regulations concerning the effluent nutrient concentration and an increasing demand and price for their products.

The centre of trout production in Germany is in the southernmost federal states of Baden-Württemberg and Bavaria where 68 % of the total trout production is realized in 60 % of the German enterprises (Brämick 2007). In 2004, in these two states 260 commercial trout farms and more than 6,000 sideline and hobby farms produced about 15,800 mt of salmonids with a value of 69.5 million Euros (Brämick 2007). Trout production in Germany was more or less stagnant over the last four years. However, the German production can provide only 51 % of the demand on portion size rainbow trout, while the rest is imported mainly from other EU countries. The reason for the production stagnation are on the one side the limited availability of suitable sites for trout production (v. Lukowicz 1994) and the limitations in the increase of production intensity on the other side due to strict federal regulations for environmental protection referring mainly on effluent nutrient wastes (Bergheim & Brinker 2003).

Wastes from aquaculture include, per definition, all materials used in the process which are not removed from the system during harvesting. The quantity of the total waste produced, which leaves the system to load the environment, is closely correlated to the culture system used (Bergheim & Asgard 1996). In intensive production systems, as used for trout production, the principal wastes are uneaten feed, excreta, chemicals and therapeutics, but the term waste can also refer to dead and morbid fish, and even escaped fish and pathogens (Bergheim & Asgard 1996).

In Germany the local water authorities are responsible for the display of fish farms operating licences. In this authorization process they have the possibility to determine the maximal effluent nutrient concentration allowed (calculated as difference between farm inflow and outflow). Usually they follow the proposals outlined in the "Recommendations for the construction and processing of fish ponds" from Schobert et al. (2001). Here the trout farms are divided in three intensification levels in dependence on the amount of feed applied per year:

1. less than 150 kg feed per Ls^{-1} and year
2. between 150 and 500 kg feed per Ls^{-1} and year
3. more than 500 kg feed per Ls^{-1} and year.

For the farms operating in the intensification level 2 and 3 the difference between inflow and outflow should not exceed 3.0 mgL^{-1} biological oxygen demand in 5 days (BOD_5) and 15 mgL^{-1} total suspended solids (TSS) (Schobert et al. 2001). The fish farms are self responsible for the compliance with the effluent nutrient concentrations and the effluent survey, while farms in the intensification level 2 should make two effluent surveys per year and farms in level 3 at least four surveys per year (Schobert et al. 2001). The local water authorities can go beyond these recommendations and can require additional nutrient limits and conduct additional effluent surveys. Usual limits set are 1.0 mgL^{-1} total ammonia nitrogen (TAN) and 0.1 mgL^{-1} total phosphorous (TP). As the fish farmer is self responsible for the compliance of the prescriptive limits, the actual thesis aims to help in this decision process and provides an overview on suitable effluent treatment possibilities and tests new promising technologies for effluent treatment.

3. Factors influencing effluent nutrient concentrations

In a first preliminary study the interaction and the impact of fish production intensity and potential effluent treatment on the final effluent nutrient concentration was examined. The aim of this preliminary study was to highlight the strength of this interaction and identify other potentially confounding factors with influence on effluent nutrient concentration, and finaly to identify the importance of effluent treatment for south German trout aquaculture.

For this purpose the inflow and outflow nutrient concentration from 13 Bavarian trout farms was surveyed. Additionally main factors potentially influencing effluent nutrient concentration were registered.

3.1 Material and methods

3.1.1 Monitored trout farms

13 trout farms were examined for their inflow and outflow water quality. All farms were situated in southern Bavaria (Germany). Six farms take their inflow water from brooks, inflow amount 100 – 800 Ls^{-1}, while seven farms were fed by spring water, inflow amount 25 – 120 Ls^{-1}. On 163 days between end of 2005 and end of 2007 farm in- and outflow was sampled. For most farms at least 12 days, while for two farms less than 12 day samples were taken. The following factors with a potential impact on effluent nutrient concentration were recorded and scaled:

3.1.2 Rearing units:

The rearing units used for fish production has to be classified in self -cleaning units, or non self-cleaning units (Willoughby 1999). Self cleaning units are characterized by a fast export of suspended particles out of the system, like concrete raceways or circular tanks (Milden & Redding 1998, Wheaton & Singh 1999). In this study, six farms used earthen ponds only as rearing units, three farms used concrete raceways only. The other four farms used a mix of concrete and earthen ponds and raceways. The amount of concrete raceways per farm was scaled from 1.00 for earthen ponds exclusively to 2.00 for concrete raceways exclusively. For the other farms the amount of raceways compared to ponds was scaled as fraction and added to 1.00.

3.1.3 Amount of feed applied / production intensity

For each farm, the fish farmer noted the amount of feed applied per day. Additionally the amount of inflow water was measured. Flow measurement was performed with a flow meter (model HFA, Höntsch inc.), measuring the mean flow velocity. Through multiple measurements, the water amount could be calculated.

Consequently the production intensity per year (Pi) was calculates as the amount of feed applied per day (f), in dependence on the amount of inflow water (Q) on a yearly base (Pi = (f • 365) / Q). All farms applied energy rich extruded feed. The production intensity of the trout farms ranged from 200 to 3370 kg $(Ls^{-1})^{-1}year^{-1}$.

3.1.4 Effluent treatment device for the farm effluent

In the surveyed farms, only mechanical treatment devices were used as effluent treatment method. The used treatment devices were scaled after their treatment efficiency (Table 1).

Six farms used no effluent treatment scaled as treatment option 1. Four farms used sedimentation basins, with a certain fish stock, scaled as treatment option 2. One of these farms used a constructed wetland for the treatment of about 20 % of the total effluent. This treatment option was scaled with 2.20. Sedimentation basins without fish were scaled as treatment option 3. A farm used a micro-screen as effluent treatment, option 4. And a farm used two consecutive micro-screens, a coarser one in the farm (as intermediate treatment) and a fine one as ‘end of pipe’ treatment, scaled as option 5 (Table 1).

Table 1: Scales assigned to the different mechanical treatment methods used in the trout farms examined.

treatment device	**scale**
no effluent treatment	1
sedimentation basin with escaped fishes	2
sedimentation basin fish free	3
micro-screen	4
two micro-screens	5

3.1.5 Water sampling and analysis

Sampling of water probes was conducted by automated water samplers. They were positioned at the in- and outflow of the fish farm. The samplers run for 24 hours. Every 10 minutes a sub sample was collected. The sub samples were mixed to 24 hour pooled samples and transported to the lab for water analysis.

The water samples were analysed for the following water parameters measured in mgL^{-1}: total nitrogen (TN), total ammonia nitrogen (TAN), nitrite nitrogen (NO_2-N), nitrate nitrogen (NO_3-N), total phosphorous (TP), phosphate phosphorous (PO_4-P), biological oxygen demand in 5 days (BOD_5), chemical oxygen demand (COD), and total suspended solids (TSS) dry weight. The physicochemical properties of the water samples were determined following German standard methods for the examination of water, wastewater and sludge (DIN 2006). For BOD_5 the total oxygen consumption of the original probe was assessed, including nitrification, and the particulate mater in the sample was not destroyed prior to measurement.

3.1.6 Data analysis

Differences (Δp) between farm inflow and outflow nutrient concentration were calculated for each parameter as well as each pair of simultaneously taken samples. For the Δp data of each parameter a Shapiro-Wilk test for normality was performed, with a significance level of $\alpha < 0.05$. When the Δp data where normally distributed, then the one sample students t-test was performed, in order to evaluate whether Δp is significantly different from 0. When normality for the Δp data was rejected, then the Wilcoxon-Test (signed rank test) was used to test whether Δp is significantly different from 0 or not.

To identify the main effects on the effluent nutrient concentration, a multivariate regression model was calculated. The following model assumption was used: $Y_{ijkl} = \mu + \alpha_i + \beta_j + \gamma_k + \delta_l + \varepsilon_{ijkl}$, where Y_{ijkl} is the relevant effluent nutrient concentration, μ is the overall effluent nutrient concentration, α_i is the inflow nutrient concentration, β_j rearing unit, γ_k used effluent treatment device, δ_l feeding amount in $kg(Ls^{-1})^{-1}year^{-1}$ and ε_{ijkl} is the random residual error. The factor were identified as relevant at a level of $\alpha < 0.05$. The residuals were tested for homogeneity and normal distribution. All statistical calculations were performed with SAS 8e.

3.2 Results and discussion

Fish farming showed for all measured nutrients a significant increase in effluent nutrient concentration, compared to inflow concentration. Except NO_3-N, here a significant decrease in the effluent concentration was measured (Table 2).

Table 2: Mean in- and outflow nutrient concentrations and standard deviation, and difference (Δp) from all monitored trout farms, with the indication of significance of Δp.

water parameter (mgL^{-1})	**average inflow (SD)**	**average outflow (SD)**	**difference Δp (SD)**	**significance of Δp**
TN	5.35	5.79	0.44	0.0001
	(1.37)	(1.55)	(0.85)	
TAN	0.038	0.467	0.429	0.0001
	(0.030)	(0.402)	(0.408)	
NO_2-N	0.031	0.081	0.049	0.0001
	(0.048)	(0.061)	(0.049)	
NO_3-N	5.28	5.09	-0.18	0.0001
	(1.23)	(1.25)	(0.54)	
TP	0.038	0.132	0.095	0.0001
	(0.034)	(0.100)	(0.106)	
PO_4-P	0.015	0.055	0.038	0.0001
	(0.024)	(0.051)	(0.050)	
BOD_5	1.57	3.73	2.13	0.0001
	(0.07)	(1.90)	(1.74)	
COD	5.80	8.95	3.06	0.0001
	(4.42)	(3.69)	(3.35)	
TSS	6.73	6.73	0.03	0.0001
	(14.69)	(4.47)	(14.51)	

The results of the regression model are shown in Table 3.

Table 3: Estimates for the regression model on effluent nutrient concentration, in dependence on relevant trout farm factors. * indicate significant estimates (S.E. = standard error, p = significance level).

water parameter mg/L	**μ**		**inflow concentration**		**rearing unit**		**effluent treatment**		**feeding 100 kg(L/s)$^{-1}$year^{-1}**		**R^2**	**p**
	estimate	p	estimate	p	estimate	p	estimate	p	estimate	p		
TN	-0.237	0.3636	0.906 *	0.0001	0.487 *	0.0108	-0.084	0.2038	0.075 *	0.0001	**0.88**	**0.0001**
TAN	0.053	0.4051	0.370	0.4495	0.016	0.7903	-0.023	0.2480	0.049 *	0.0001	**0.82**	**0.0001**
NO_2-N	0.112 *	0.0001	0.709 *	0.0001	-0.035 *	0.0219	-0.010	0.0555	0.002 *	0.0003	**0.50**	**0.0001**
NO_3-N	-0.498 *	0.0478	0.914 *	0.0001	0.492 *	0.0036	-0.031	0.5990	0.013 *	0.0368	**0.86**	**0.0001**
TP	0.020	0.3492	0.333 *	0.0379	0.025	0.2484	-0.014 *	0.0443	0.011 *	0.0001	**0.63**	**0.0001**
PO_4-P L	-0.006	0.4938	0.658 *	0.0001	-0.007	0.4652	0.008 *	0.0122	0.005 *	0.0001	**0.74**	**0.0001**
BOD_5	0.329	0.4213	0.912 *	0.0001	0.785 *	0.0375	-0.347 *	0.0053	0.184 *	0.0001	**0.68**	**0.0001**
COD	2.461 *	0.0023	0.686 *	0.0001	1.417	0.0804	-0.636 *	0.0218	0.210 *	0.0001	**0.61**	**0.0001**
TSS	8.118 *	0.0001	0.728 *	0.0014	-4.384 *	0.0154	0.516	0.3457	0.156 *	0.0217	**0.13**	**0.0012**

Effluent nutrient concentration can be predicted by 50 – 88 % through four main factors: feed amount applied, inflow nutrient concentration, rearing unit used and effluent treatment device. Only for TSS the predictability is about 13 %, mainly due to flooding occurred during the samples, leading to increased TSS loads in the inflow and a retention and delayed export of TSS from the fish farm. Also other studies had major problems in predicting TSS outflow from trout aquaculture (Roque d'Orbcastel et al. 2008).

An increase in the amount of feed applied in the fish farm, effected for all measured nutrients a significant increase in the effluent nutrient concentration. Thus fish feeding is the most important factor influencing effluent nutrient concentration. The high impact of feeding on the effluent nutrient concentration lead to calculation formulae, predicting the effluent nutrient concentration based on the feeding amount feed wastes and digestibility data (Bergheim & Asgard 1996, Bureau et al. 2003). The calculation of effluent nutrient increase based only on feeding data might be the simplest and cheapest method (Roque d'Orbcastel et al. 2008). However, the calculation rely on feed digestibility data, hardly to achieve (Roque d'Orbcastel et al. 2008), and feeding is only one factor among other, influencing effluent nutrient concentration (Table 3).

The inflow nutrient concentration had also for all nutrients except TAN an direct effect on effluent nutrient concentration, as the initial pollution level is more or less passed unaffected through the farm. However, this reveals the crucial importance of taking inflow and outflow samples in order to correctly assess the effect of trout farming (Foy & Rossel 1991, Rennert 1994).

Self cleaning farms released more TN and NO_3-N, while less NO_2-N and TSS were released. The lower release of NO_3-N and TN from ponds is due to denitrification occurring in the pond sediments. Here oxygen free areas occur, with sufficient carbon sources from settled faeces, enhancing denitrification (Tchobanoglous et al. 2003) of the naturally nitrate rich inflow water (Table 1). Additionally less BOD_5 is exported from ponds, compared to raceways, as heterotrophic digestion is one of the first processes occurring in oxygen rich environments (Tchobanoglous et al. 2003). The reason for the higher release of TSS from ponds, compared to raceways is not clear. It might be in connection with the fact, that most trout farms with ponds as rearing units, surveyed in the study, receive their water from brooks and rivers. In case of flooding the TSS were slowly transported out of the farm, leading to a higher effluent TSS concentration when no increased TSS concentration in the inflow was found.

However, self cleaning rearing units should be characterized by a fast export of suspended particles out of the system (Milden & Redding 1998, Wheaton & Singh 1999). In salmonid farming the main part of the nutrient released are particle bound, in detail: about 80 % of organic carbon, 7 – 32 % of total nitrogen (TN) and 30 – 84 % of TP (Cripps & Bergheim 2000). Leaching of dissolved nutrient fractions from particulate waste is a time and temperature depending process (Stewart et al. 2006). Through the fast waste export out of the rearing system, high nutrient concentrations in the particulate fraction can be reached, in consequence a subsequent mechanical treatment is most effective in effluent nutrient abatement.

Non self cleaning rearing units, especially fish ponds are characterized through a high internal sedimentation (Milden & Redding 1998, Willoughby 1999, Schobert et al. 2001). The particles remain at the pond bottom during the whole rearing period until the pond is harvested, drained and cleaned. During this long period of time a main part of the particle bound nutrients were dissolved to the pond water and exported as dissolved nutrients to the effluent. In dependence of the production intensity a effective dissolved nutrient treatment is needed, as pure mechanical treatment is ineffective. Never the less a mechanical treatment for the highly particle and nutrient loaded pond cleaning water is needed (Schobert et al. 2001). Here a sedimentation basin with a subsequent media filter provided high treatment efficiencies (Sindilariu & Reiter 2007).

The mechanical effluent treatment units showed a significant treatment effect on TP, BOD_5 and COD. For PO_4-P they lead to a slight increase in the effluent concentration, due to leaching occurring especially in sedimentation ponds (Cripps & Bergheim 2000). With increased efficiency and technical improvement of the treatment unit, the effluent nutrient concentration decreased. Only for TSS where the main effect from mechanical treatment should be supposed (Cripps Bergheim 2000), no effect was found. Probably due to over and underestimation of TSS in water sampling caused by insufficient mixing of the effluent (Brinker et al. 2005).

From this preliminary study, the fish farmer has two possibilities to influence the effluent nutrient concentration, as the type of the rearing units is in most cases set:

1. through the amount of feed applied
2. through the implementation of effective effluent treatment devices.

If the farmer wants to increase the production intensity due to increased prices and higher profit margins for trout, the increased effluent nutrient concentration has to be balanced by the implementation of an effective effluent treatment. For further production expansion and effluent load reduction, as revealed by the preliminary study, an ‘end of pipe’ effluent treatment is needed.

4. Concluding research objective

For successful effluent treatment some information concerning the effluent characteristics, the effect of nutrients to the environment as well as the legislative frame are of essential importance, to choose the ‘right’ situation adapted treatment method:

1. the type of pollutants, their concentration in the effluent and the effect on adjacent ecosystems (Seymour & Bergheim 1991, Piedrahita 1994),
2. the design and performance of different treatment technologies, their possible combination and specific investment and operational costs,
3. the available geographic, legislative and financial space for the implementation of the ‘best’ methodology.

From these three crucial points, the available geographic, legislative and financial space for the implementation of the ‘best’ methodology is completely farm specific an in the decision of the particulate fish farmer.

The type of pollutants and especially the factors influencing the final nutrient concentration and distribution in the farm effluent were characterized in the preliminary study above.

The effect of the farm effluent on the adjacent ecosystems and the design and performance of different treatment technologies suitable for trout aquaculture are the core theme of the review “Reduction in effluent nutrient loads from flow-through facilities for trout production: a review” (attachment 1). This literature overview revealed, that untreated trout farm effluents can have an impact on all levels of aquatic life, starting from the aquatic microbial-, plant and invertebrate fauna, till the vertebrate fish community composition. The impact is caused either by a single factor or the combination of different factors. The most important factors are: the enrichment of the plant nutrients nitrogen and phosphorus, the discharge of biodegradable organic compounds, leading to an increased heterotrophic consumption, the elevated amounts of total suspended solids leading to interstitial clogging, the oxygen depletion in the effluent through heterotrophic consumption and ammonia nitrification, and the direct toxic effects of ammonia and nitrite.

The main part of the literature study deals with effluent treatment methods. Here suitability for trout farm applications as well as treatment efficiency and potential treatment costs were extracted from the comprehensive literature on effluent treatment methods. The information was condensed in Fig. 1 (Fig. 11 in the review) and Table 4 (Table 4 in review), proposing different treatment options in dependence of rearing system used and production intensity. Additionally the effluent treatment costs were estimated as far as possible (Table 4).

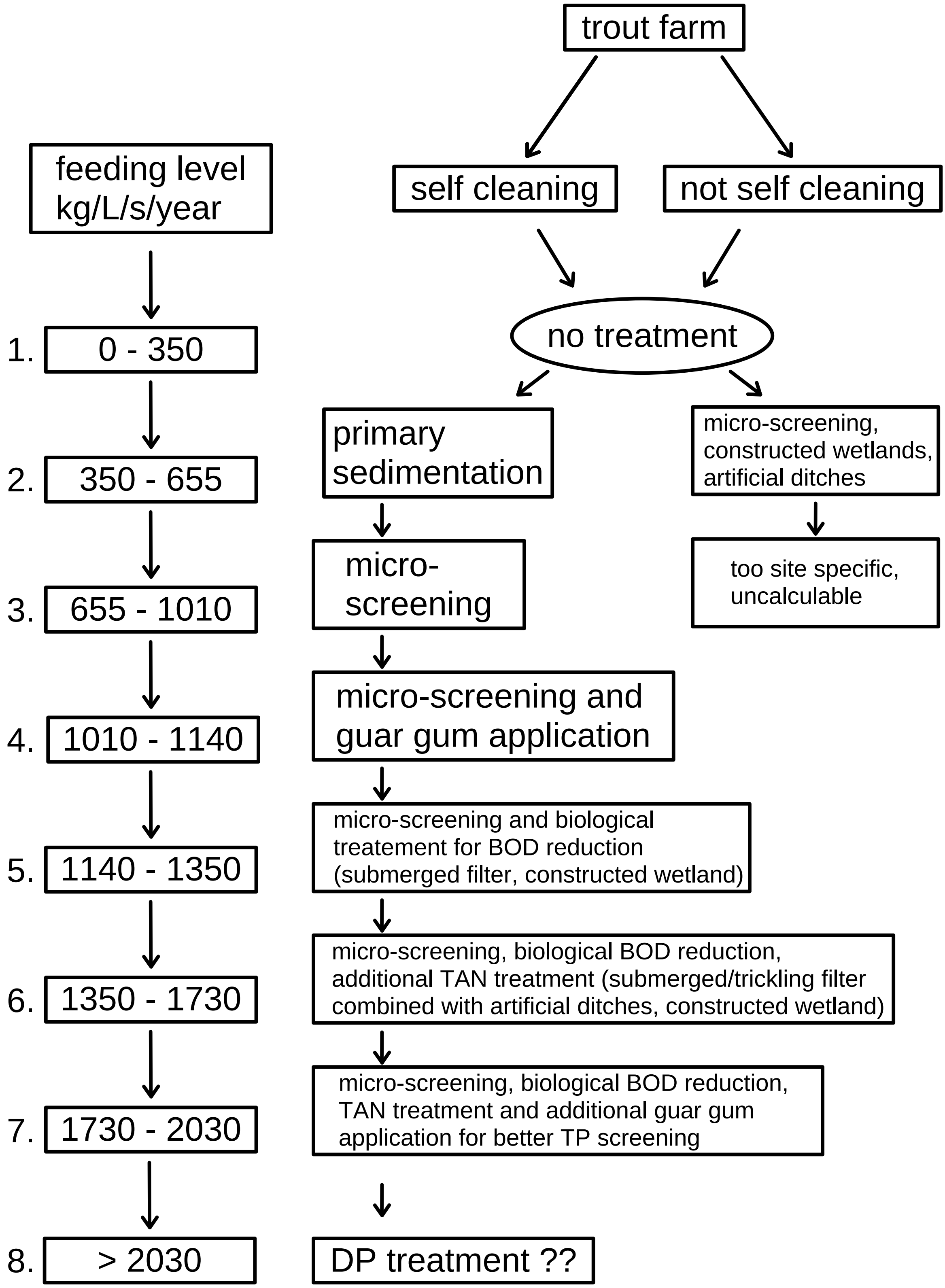


Fig. 1: Possible decision tree for the implementation of effluent nutrient reduction strategies in trout farms, in dependence on the feeding level.

Table 4: Nutrient concentrations, amount of inhabitant equivalents and profit margins in € of trout an example trout farm($Q = 542\ Ls^{-1}$, FCR = 1, FW = 4 %) at different feeding levels and resulting nutrient reduction strategies.

Feeding level	**untreated effluent nutrient concentration**	**amount of inhabitant equivalents**	**treatment options**				**profit margin €**	**profit margin €**
kg per Ls^{-1} per year	**TP; DN; BOD_5 in mgL^{-1}**	**one ieq. = 60 $gBOD_5$**	**option 1 €**	**option 2 €**	**option 3 €**	**option4 €**	**no treatment**	**with treatment**
350	0.073; 0.27; 3.1	2,419	no treatment				237,125	237,125
655	0.136; 0.50; 5.8	4,527	primary sedimentation, no data on relevant costs available				443,762	-
1010	0.210; 0.77; 8.9	6,946	micro-screening 9,600	sludge treatment 3,042			684,275	671,633
1140	0.236; 0.87; 10.0	7,805	micro-screening 9,600	sludge treatment 3,042	guar gum 28,025.		773,350	731,683
1350	0.280; 1.04; 11.9	9,288	micro-screening 9,600	sludge treatment 3,042		SF wetland 303,477	914,625	697,856
1730	0.359; 1.33; 15.3	11,941	micro-screening 9,600	sludge treatment 3,042		SSF wetland not available	1,172,075	-
2030	0.421; 1.56; 17.9	13,971	micro-screening 9,600	sludge treatment 3,042	guar gum 49,905	SSF wetland not available	1,375,325	-

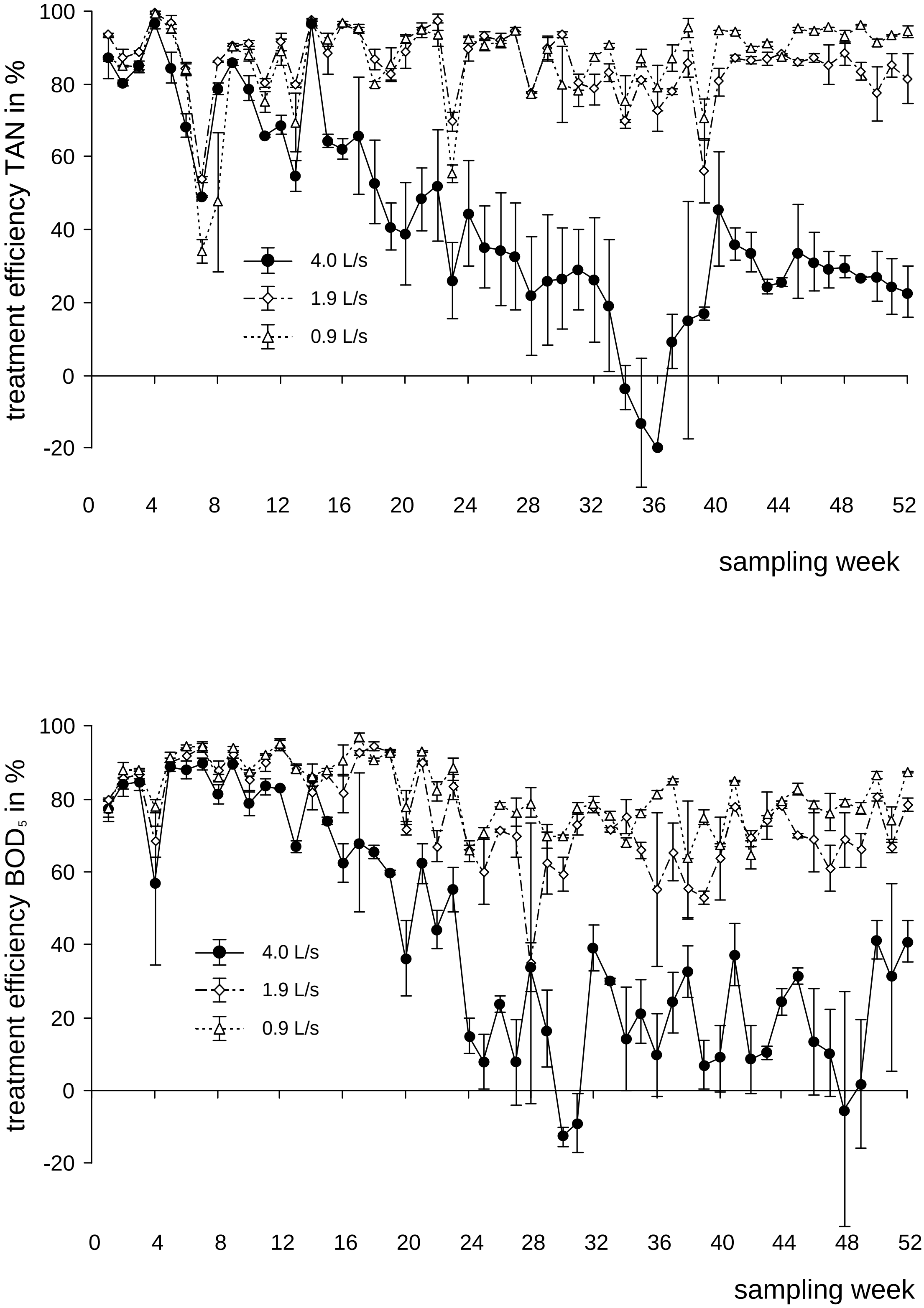
From these results two essential research needs were identified:

1. For non self-cleaning trout farms no specific treatment options are available.
2. For the application of SSF constructed wetlands under intensive production conditions, no reliable data for nutrient treatment efficiency and treatment costs are available as all reports on effluent treatment deal with experimental wetland application (Schulz et al 2003, Lin et al. 2005).

Most of the smaller south German trout farms operate with ponds as rearing units. Nevertheless even in small farms with a low amount of water, high production intensities can be reached. For pond effluents mechanical treatment is not suitable as most of the particulate matter settles in the pond and mainly dissolved nutrients were released. For low to medium intensive non self-cleaning trout farms, constructed wetlands were tested as potential effluent treatment device (attachment 2). This commercial scale test showed, that SSF wetlands with a pre-sedimentation area are highly suitable for effluent treatment from farms using earthen rearing units. Compared to the initial sedimentation basin, the sub-surface flow (SSF) wetland showed a highly improved treatment efficiency during normal operation and pond cleaning situation. Treatment efficiencies of up to 88 % for TAN were reached. These results recommend constructed wetlands as a treatment method for trout farms using earthen ponds or raceways and with a production intensity beyond 350 kg $(Ls^{-1})year^{-1}$.

To bring insight in the second research need resulted for the literature study, a commercial scale test of constructed wetlands for the treatment of intensive trout farm effluents was conducted for the first time (attachment 3, 4). The examined wetland cells were principally suitable for the effluent treatment from intensive trout farming, showing high treatment efficiencies of maximum 75 – 86 % for TAN, BOD_5, and TSS. However, the cells receiving a hydraulic load > 14 $m^3m^{-2}day^{-1}$ (about 4 Ls^{-1} per wetland cell) showed after a short service time sever clogging phenomena, were overflowed, and the treatment efficiency of these cells decreased constantly about 14 weeks after the start of the sampling period (18 weeks after the star of intensive fish farming), with no recover (Fig. 2).

Fig. 2: Temporal development of the treatment efficiencies of BOD_5 and TAN in depend ence on the hydraulic load.



Through the application of three hydraulic load treatments on six identical wetland cells, three treatments with true replicates were examined over 12 month. Thus, through linear correlative statistics, the factors influencing wetland treatment efficiency could be identified (attachment 4).

Three main factors influence wetland treatment efficiency. The first, inflow nutrient concentration, is directly dependent on the farm production intensity. The remaining two factors, the hydraulic load to the wetland and the TSS accumulation in the wetland root zone, are important considerations in wetland dimensioning, and have implications for service lifetime and financial costs at any given production intensity. Vegetation period and fish harvesting are of minor importance compared to the other factor influencing wetland treatment efficiency.

Thus, the results of the current study (attachment 4) allow for the first time a realistic estimation of costs for commercial scale wetland treatment systems for intensive trout production (Table 5). The hydraulic load influences the wetland area needed, while TSS pre-treatment directly influences the TSS accumulation in the wetland and thus the wetland service lifetime. An efficient combination between service lifetime and land requirements depends on the distribution of costs between fixed-term and service lifetime depreciations.

As a stand-alone treatment for the effluent, SFF constructed wetlands are not suitable for intensive trout farms. The annual costs of € 23,000 – 28,000 to treat 100 Ls^{-1} effluent are prohibitive (Table 5). However the costs decrease remarkably when SSF wetlands are used in conjunction with effective pre-treatment. In the present case the up-scaled cost of trout production was an additional € 0.20 per kilo, an expense fisheries managers and consumers should find justifiable when nutrient emission is kept to a minimum. Additionally this expense can be further decreased through higher production intensity in the farm.

Table 5: Calculated service lifetime (years) for TAN treatment efficiency > 50% and yearly effluent treatment costs (€) of the SSF wetland treatment for a 100 Ls^{-1} example trout farm with an annual production of about 770 $kg(Ls^{-1})^{-1}$, in dependence on hydraulic load and TSS pre-treatment.

	No pre treatment	**50% TSS treatment micro-screen**	**80% TSS treatment micro-screen**
Hydraulic load ($m^3 m^{-2} day^{-1}$)	14.5	14.5	14.5
Wetland area (m^2)	600	600	600
Service lifetime (years)	0.67	1.4	3.5
Annual costs SSF wetland (€)	27,680	14,690	7,540
Total annual costs with micro-screen (€)	**27,680**	**22,600**	**15,450**
Hydraulic load ($m^3 m^{-2} day^{-1}$)	6.9	6.9	6.9
Wetland area (m^2)	1,255	1,255	1,255
Service lifetime (years)	2.0	3.9	9.8
Annual costs SSF wetland (€)	23,410	14,850	9,430
Total annual costs with micro-screen (€)	**23,410**	**22,760**	**17,340**
Hydraulic load ($m^3 m^{-2} day^{-1}$)	3.3	3.3	3.3
Wetland area (m^2)	2,650	2,650	2,650
Service lifetime (years)	4.6	9.3	13.7
Annual costs SSF wetland (€)	28,460	20,310	17,750
Total annual costs with micro-screen (€)	**28,460**	**28,220**	**25,660**

5. Conclusions and recommendations

With increasing production intensity, the effluent nutrient concentrations in the farm effluent increase too. In order to meet stringent effluent nutrient margins, “end of pipe” effluent treatment is the only method despite internal farm management to reduce effluent nutrient concentrations.

Regarding the suitability of effluent treatment methods the trout farms have to be divided in two categories: farms using not self-cleaning rearing units like ponds with natural embankments and farms using self-cleaning rearing units, like raceways and circular tanks.

For not self-cleaning trout farms general recommendations regarding suitable effluent treatment devices can hardly be given. The main problem is the lacking predictability of the distribution between dissolved and particulate nutrient fraction in the farm effluent. Thus a treatment method with high treatment efficiency for both nutrient fractions seems most suitable. Here horizontal sub-surface flow (SSF) constructed wetlands provide an effective effluent treatment especially for low to medium intensive trout farming.

For self-cleaning trout farms, the main part of the nutrients were exported as particle bound nutrients, due to the fast transport out of the rearing system. Thus, the most suitable effluent treatment possibilities are first of all mechanical treatment devices.

At high production intensities dissolved nutrient fractions are of importance. Thus under these condition, biological treatment methods were needed to polish excess dissolved nutrient loads. SFF wetlands were tested for their suitability under high production condition, with the following results:

1. Constructed wetlands are highly effective for treating effluent from commercial scale intensive trout farms.
2. Treatment efficiency is dependent on the concentration of nutrients flowing into the wetland, the hydraulic load and accumulation of TSS in the wetland root zone.
3. Costs of SSF wetland effluent treatment are dependent on three factors: the inflow concentration (strongly influenced by the production intensity), the hydraulic load and the TSS accumulation.
4. High costs mean SSF wetlands are not suitable as a stand alone treatment method for intensive trout farm effluents, but they make an economically viable solution when used in combination with effective pre-treatment.

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Appendix

Accompanying papers
(Appendix)

Paper 1

Sindilariu, P.-D. 2007. Reduction in effluent nutrient loads from flow-through facilities for trout production: a review. Aquaculture Research 38, 1005-1036.

Paper 2

Sindilariu, P.-D., Schulz, C., Reiter, R. 2007. Treatment of flow-through trout aquaculture effluents in a constructed wetland. Aquaculture 270, 92-104.

Paper 3

Sindilariu, P.-D., Wolter, C., Reiter, R. 2008. Constructed wetlands as a treatment method for effluents from intensive trout farms. Aquaculture 277, 179-184.

Paper 4

Sindilariu, P.-D., Brinker, A., Reiter, R. 2009. Factors influencing the efficiency of constructed wetlands used for the treatment of intensive trout farm effluent. Ecological Engineering (in press).

REVIEW ARTICLE

Reduction in effluent nutrient loads from flow-through facilities for trout production: a review

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Abstract

The environmental legislation on waste loadings and the public discussion on environmental issues concerning inland aquaculture facilities have become stronger in recent years. An end of the discussion cannot be foreseen. At the same time, the pollution emitted per ton of fish produced has decreased successively over the last 20 years. In this conflict, this paper provides an overview on: (1) pollutants typical for flow-through trout aquaculture, (2) their source and potential environmental impacts and (3) strategies to reduce the effluent pollution from flow-through trout farms, a brief description of their function principles and application and, if possible, their economical feasibility. This study aims to identify the actual effluent problems of flow-through trout aquaculture and to offer possible solutions by either pollution avoidance or effluent treatment. Future trends and research needs on effluent treatment are outlined. As the amount of nutrients discharged is typically site and operation specific, farm management is most important for avoidance of effluent pollution. Nevertheless, for further production expansion, 'end of pipe' technologies are needed to reach adequate effluent qualities. Partial water reuse can improve effluent discharge. Physical, chemical and biological technologies can be used to treat trout farm effluents. Today, the commonly used physical (mechanical) treatments in trout aquaculture like screening and sedimentation only remove suspended solids, containing up to 7–32% of total nitrogen and 30–84% of total phosphorus. The remaining soluble nutrients can only be removed by either the application of chemicals or biological effluent treatments. The possible applications of biological technologies are manifold, but practical and upscale experience is lacking.

Keywords: flow-through trout aquaculture, effluent treatment

Introduction

The production of salmonids nearly reached 1.8 million metric tonnes in 2002 (FAO 2004) including about 500 000 mt rainbow trout, *Oncorhynchus mykiss* (Walbaum). Rainbow trout is an important finfish species in European aquaculture, with a production of 226 549 mt annually (Eurostat 2004), and an important culture species in the USA, Canada and Chile (Fornshell 2002). In the trout-producing sector, small, regionally rooted micro enterprises dominate, with an annual production of < 100 mt (MacAlister Elliott and Partners 1999; FAO 2003; Klinkhardt 2004; Engle, Pomerleau, Fornshell, Hinshaw, Sloan & Thompson 2005).

Trout is traditionally produced in inland flow-through systems, without water reuse. The water residence time in the rearing unit is usually in the range of minutes (Lawson 1995). The inflowing water can be used once, when the units are parallel grouped, or up to several times, when grouped in series. Sometimes, a partial water recirculation is installed (Lawson 1995; Wheaton & Singh 1999). The rearing units are either earthen or concrete raceways or tanks (Timmons, Riley, Brune & Lekang 1999).

For successful production, high quality and quantity standards are set to the inflow water. Trout culture in freshwaters is mainly determined by the amount and quality of water available, limiting the maximum production capacity at a location (von Lukowicz 1994). The water flow required to produce 1 mt of trout decreased successively with the development of oxygenation and aeration technologies and the switch to energy-rich, extruded feeds. Today, more than 1 mt of trout is produced per L s^{-1} $year^{-1}$, with a food conversion ratio (FCR) <0.8. Additionally, cost saving, ecologically driven, waste-minimizing strategies were implemented, so that pollution decreased while production increased (Milden & Redding 1998; Bergheim & Brinker 2003; McMillan, Wheaton, Hochheimer & Soares 2003; Summerfelt, Davidson, Waldrop, Tsukuda & Bebak-Williams 2004). Nevertheless, more stringent environmental legislation and increased public awareness still asks for more efficient cleaning technologies (Naylor, Goldburg, Primavera, Kautsky, Beveridge, Clay, Folke, Lubchenco, Mooney & Torell 2000; EIFAC 2001; O'Bryen & Lee 2003; Tacon & Forster 2003; Viadero, Cunningham, Semmens & Tierney 2005). The end of this development is not in sight.

In most European countries, some states of the USA and Canada, governmental policies aim to reduce the environmental impacts of aquaculture by:

- limiting the concentration and/or mass of specific dissolved/suspended inorganic/organic materials and/or nutrients contained within farm effluents,
- requiring the implementation of an environmental monitoring programme (Tacon & Forster 2003).

To meet the regulations, waste-minimizing programmes using disciplined farm-specific best management practice (BMP) plans and 'high end' effluent treatment are required (McMillan *et al.* 2003). Inclusion of partial water recirculation further improves farm management and effluent treatment (Piedrahita 2003).

Many different effluent cleaning technologies have been developed. In order to choose the 'right' situation-adapted treatment, the following information is important:

1. the type of pollutants, their concentration, source and impact on adjacent ecosystems (Seymour & Bergheim 1991; Piedrahita 1994),
2. the design and performance of the different cleaning technologies, their possible combinations and specific costs and
3. the available geographic, legislative and financial space for the implementation of the 'best' methodology.

In this article, an overview on the nutrient pollutants typical for flow-through trout aquaculture and their potential environmental impact is given. Strategies for effluent nutrient load reduction are outlined, including some tests performed in salmon smolt farms. However, the removal effects are considered to be similar in salmon smolt farms and intensively run trout farms. Finally, the review highlights future trends and developments in trout farm effluent management.

Source and degree of trout farm effluent pollution

Aquacultural waste, by definition, includes all materials that are not removed through harvesting. The principal wastes are uneaten feed, excreta, chemicals and therapeutics (Bergheim & Asgard 1996). The main waste source is formulated feed (Steffens 1985; Bergheim & Asgard 1996). Trout grower feed contains 38–50% protein, 14–35% fat, 1–4% fibre and about 8% ash. The total energy is between 21 and 25 MJ kg^{-1} (Biomar AS 2006). The feed contains 31–57% organic carbon (OC), 6–8% total nitrogen (TN) and 0.9–1.1% total phosphorus (TP).

Less than 4% of the feed remains uneaten in commercial trout farms (Rösch, Hammers & Brinker 2003). From the ingested nutrients, the undigested part is excreted as particulate faeces (Cho, Hynes, Wood & Yoshida 1994; Piedrahita 1994; Bergheim & Asgard 1996; Cho & Bureau 1997; Green, Hardy & Brannon 2002), containing mainly OC and phosphorus (Cripps 1994; Kelly, Bergheim & Stellwagen 1997; Cripps & Bergheim 2000). After Bergheim and Asgard (1996), from the ingested nutrients, about 19% of OC, 13% of TN and 55% of TP is excreted as faecal waste.

The digested nutrients are partially retained in fish body mass (Schreckenbach, Knösche & Ebert 2001). The rest is excreted as dissolved nutrients through the gills, mainly as ammonia, and via urine as phosphate and ammonium (Steffens 1985; Cho *et al.* 1994; Cho & Bureau 1997; Bureau & Cho 1999; Green *et al.* 2002; Roy & Lall 2004). The dissolved nutrient excretion from the ingested feed comprises 36% of TN and 9% of TP (Bergheim & Asgard 1996).

From these known figures, the nutrient release per metric ton of feed can be calculated for flow-through trout production, as well as the nutrient concentration increase in the effluent (Cho *et al.* 1994; Bergheim & Asgard 1996; Kelly, Stellwagen & Bergheim 1996; Cho & Bureau 1997, 1998, 2001; Bureau, Gunther & Cho 2003; Brinker, Berg & Rösch 2006). To calculate the concentration increase, the feeding rate (FR), the feed wastes (FW), the food conversion ratio (FCR), the nutrient contents in the

Table 1 Calculation formulae for the estimation of nutrient release in flow-through trout farms, depending on feeding rate [FR (kg day^{-1})], feed waste [FW (%)], feed conversion ration [FCR (kg feed kg growth^{-1})] and effluent flow rate [Q (L s^{-1})] after Brinker *et al.* (2006)

Nutrient concentration increase (mg L^{-1})	Calculation formula
PN	$n \times \frac{(FR - FR \times FW) \times (1 - d_N) + FR \times FW}{Q} \times \frac{10^4}{864}$
DN	$\frac{n \times (FR - FR \times FW) \times d_N - 0.027^{*} \times FCR \times FR}{Q} \times \frac{10^4}{864}$
PP	$p \times \frac{(FR - FR \times FW) \times (1 - d_P) + FR \times FW}{Q} \times \frac{10^4}{864}$
DP	$\frac{p \times (FR - FR \times FW) \times d_P - 0.004^{*} \times FCR \times FR}{Q} \times \frac{10^4}{864}$
BOD_5	$0.85^{*\dagger} \times 20 MJ\,(kg^{-1})^{\ddagger} \times (13.6\,MJ(kg^{-1}))^{-1\S} \times \frac{(FR - FR \times FW) \times (1 - d_{oc}) + FR \times FW}{Q} \times \frac{10^4}{864}$

PN, particulate nitrogen; DN, dissolved nitrogen; PP, particulate phosphoros; DP, dissolved phosphoros; BOD_5, biological oxygen demand in 5 days; n, the nitrogen fraction in the trout feed; p, the phosphoros fraction in the trout feed; d_i, the digestibility of the relevant nutrient fraction I, dependent on the trout feed characteristics; FR, feeding rate; FW, feeding waste; FCR, food conversion ratio.
*Percentage of nutrient in total body trout composition.
†Mean amount of organic matter in trout feed.
‡Gross energy content of the organic matter in trout feeds.
§Mean amount of energy digested per kilogram oxygen (O_2) by microorganisms using trout feeds.

applied trout feed (*n*, *p*), the apparent digestibility of the organic matter, nitrogen and phosphorus components (d_{oc}, d_N, d_P) and the effluent flow rate (Q) are needed (Cho & Bureau 2001; Rösch *et al.* 2003; Brinker *et al.* 2006). The calculation formulae are given in Table 1. For an example trout farm ($Q = 542$ L s^{-1}, FR = 800 kg day^{-1} and FW = 0.04), the calculated nutrient increase (after Table 1) is set in relation to measured effluent concentrations from the farm (Table 2).

From the feed applied in the aquaculture process, two waste streams were emitted: the particulate nutrients from uneaten feed and faecal excretion and the dissolved nutrients from dissolved excretion. The final distribution in the effluent between the two streams depends on the local physical, chemical and biological conditions. From the first contact between nutrients and water, leaching and incorporation processes influence the distribution (Vens-Cappell 1985; Garcia-Ruiz & Hall 1996; Chen, Beveridge & Telfer 1999; Temporetti & Pedrozo 2000; Tlusty, Snook, Pepper & Anderson 2000; Chen, Beveridge, Telfer & Roy 2003; Stewart, Boardman & Helfrich 2006a).

1. *Leaching*: Water turbulences and the physical, biological and chemical particle properties like viscosity and elastic resistance but also the presence of fish mucus or chemical composition influence the particle dispersion and the water contact surface area (Brinker 2005; Brinker, Koppe & Rösch 2005a; Stewart *et al.* 2006a). The amount of chemical and biological leaching is dependent on the particle-specific surface area. Generally speaking, the finer the dispersion, the higher the leaching (Vens-Cappell 1985; Phillips, Clarke & Mowat 1993; Hopkins, Sandifer & Browdy

Table 2 Comparison of calculated and measured concentration increase in an existing German trout farm

	Concentration increase (mg L^{-1})	
Nutrient	Calculated*	Measured†
TAN		0.47
NO_2-N		0.02
NO_3-N		0.06
DN	0.5382	0.55
PN	0.197	
TN	0.735	0.59
DP	0.013	0.02
PP	0.097	
TP	0.120	0.14
BOD_5	4.75	4.11

FR = 800 kg day^{-1}, FW = 4%, $Q = 542$ L s^{-1}, the applied trout feed at a FCR = 1, has the following characteristics: $n = 7\%$, $p = 1.1\%$, $d_N = 87\%$, $d_P = 45\%$, $d_{OC} = 81\%$.
TAN, total ammonia nitrogen; TN, total nitrogen; DN, dissolved nitrogen; PN, particulate nitrogen; DP, dissolved phosphorus; PP, particulate phosphorus; TP, total phosphorus; BOD_5, biological oxygen demand in 5 days.
*Calculated effluent nutrient concentration increase after Table 1. In a real trout farm feeding 800 kg extruded feed with 542 L s^{-1} inflow.
†Measured effluent nutrient concentration increase, mean values from three consecutive 24-h samples in June 2006 (P.-D. Sindilariu, unpublished).

Table 3 Nutrient concentration of flow-through trout farm effluents reported in the literature

Effluent nutrient concentration (mg L^{-1})	Location				
	Northern Europe end of 1980s	Austria end of 1980s	Northern Portugal mid of 1990s	Representative Southern Idaho 2001–2002	West Virginia 2000–2002
TN	1.4	1.62–8.46	–	–	–
NH_4-N	–	0.26	0.27–1.52	–	0.03–0.33
NO_2-N	–	0.007	<0.2	–	–
NO_3-N	–	1.42	1.0–2.1	–	–
TP	0.125	0.053	–	0.10	–
PO_4-P	–	0.038	0.06–0.591	0.06	–
BOD_5	8	2.8	0.9–14	–	0–3.3
TSS	14	3.7	1.8–17.8	4.28	1.6–9.0
Source	Cripps (1994)	Butz (1990)	Boaventura *et al.* (1997)	True *et al.* (2004a)	Viadero *et al.* (2005), Maillard *et al.* (2005)

TSS, total suspended solids.

1994; Garcia-Ruiz & Hall 1996; Milden & Redding 1998). Leaching is also time and temperature dependent (Stewart *et al.* 2006a).

2. *Incorporation*: The dissolved nutrients are incorporated and transformed into particulate biomass through biological plant and microorganisms' uptake.

The nutrient distribution in trout farm effluents is highly variable. About 80% of the OC, 7–32% of TN and 30–84% of TP in the effluent are particle bound. The remaining dissolved nutrients are found as dissolved OC (DOC), as ammonia and ammonium (NH_3-N/NH_4-N), dissolved organic nitrogen (amino acids), nitrite (NO_2-N), nitrate (NO_3-N), as phosphate (PO_4-P) and polyphosphates (Cripps & Bergheim 2000). In a raceway trout farm in Germany, Brinker, Koppe and Rösch (2005b) found a distribution of 68% particulate phosphorus from TP and 25% particulate Kjeldal nitrogen from total Kjeldal nitrogen. Nutrient effluent concentrations of trout farms from the literature are listed in Table 3. In Table 4, an overview of the amount of pollutant equivalents emitted through a trout farm in dependence of the feeding level is given.

Farm effluents can also contain pathogens, chemotherapeutants and antibiotics (Braaten 1991; Seymour & Bergheim 1991; Ackefors & Enell 1994; Johnsen & Jensen 1994; Rennert 1994; Cripps 1995; Axler, Tikkanen, Henneck, Schuldt & McDonald 1997; Cripps & Bergheim 2000; Dumas & Bergheim 2001; Chen *et al.* 2003), but these residuals are not the focus of this review.

Potential impact of aquaculture effluents on adjacent ecosystems

The effect of trout farm effluents on ecosystems is a function of the amount and type of pollutants and the assimilative capacity of the receiving system (Rosenthal 1994; O'Bryen & Lee 2003; Piedrahita 2003). Potential environmental problems that can arise from aquaculture effluents are as follows:

Reaction on nutrient enrichment

Cold water stream ecosystems can show a typical reaction or shift in the river continuum when disturbed by nutrient-rich trout farm effluents (Loch, West & Perlmutter 1996). Effluents with high organic loads [biological oxygen demand in 5 days (BOD_5) > 2.1 mg L^{-1}] show, in the vicinity of the discharge point (maximum 100 m), a dominance of heterotrophic bacteria and sewage fungi suppressing the primary production (Doughty & MCPhail 1995; Loch *et al.* 1996; Villanueva, Queimalinos, Modenutti & Ayala 2000). The heterotrophic dominance is followed by an increased primary production measured as chlorophyll *a*. The increase is related to the inorganic TN and TP enrichment (Brown & Goulder 1996; Selong & Helfrich 1998; Fries & Bowles 2002). The chlorophyll biomass follows the concentration of TN and TP conforming to Eq. (1) (Dodds, Smith & Lohman 2002).

$$\log(\text{mean chl}) = 0.155 + 0.236\log(\text{TN}) + 0.433\log(\text{TP}) \quad (1)$$

Table 4 Nutrient concentrations, amount of inhabitant equivalents and profit margins in US$ of trout farm from Table 2 at different feeding levels and the resulting nutrient reduction strategies

Feeding level	Untreated effluent nutrient concentration ($mg\ L^{-1}$)			Amount of inhabitant equivalents	Treatment options (US$)				Profit margin (US$)	Profit margin (US$)
($kg\ L^{-1}\ s^{-1}\ year^{-1}$)	TP	DN	BOD_5	one ieq. = 60 g BOD_5	option 1	option 2	option 3	option4	No treatment	With treatment
350	0.078	0.25	3.1	2419	No treatment				331 975	331 975
655	0.145	0.65	5.8	4527	Primary sedimentation, no data on relevant costs available				621 267	–
1010	0.224	1.01	8.9	6946	Micro-screening 13 440	Sludge treatment 4259			957 985	940 286
1500	0.333	1.50	13.2	10 302	Micro-screening 13 440	Sludge treatment 4259		SF wetland 303 477	1 422 750	1 101574
1680	0.373	1.68	14.8	11 551	Micro-screening 13 440	Sludge treatment 4259	Guar gum 57 821	SF wetland 303 477	1 593 480	1 214 484
1750	0.388	1.75	15.4	12 019	Micro-screening 13 440	Sludge treatment 4259		SSF wetland not available	1 659 875	–
2100	0.466	2.10	18.5	14 439	Micro-screening 13 440	Sludge treatment 4259	Guar gum 72 276	SSF wetland not available	1 991 850	–

TP, total phosphorus; TN, total nitrogen; DN, dissolved nitrogen; BOD_5, biological oxygen demand in 5 days.

where mean chl is the mean benthic stream chlorophyll in $mg\ m^{-2}$ and TN and TP in $\mu g\ L^{-1}$. The regression turning point for mean benthic stream chlorophyll is around 38 $\mu g\ L^{-1}$ for TN and 30 $\mu g\ L^{-1}$ for TP (Dodds *et al.* 2002). The recovery distance for the chlorophyll biomass depends on the nutrient load and can occur towards 175 m from the discharge point for low polluted effluents (Fries & Bowles 2002), or might need much more than 400 m (Selong & Helfrich 1998).

Farm effluent TP concentrations are between 35 and 125 $\mu g\ L^{-1}$ (Table 3). In comparison, eutrophic lakes have a TP concentration of 30–100 $\mu g\ L^{-1}$ during spring circulation (Lampert & Sommer 1993).

The heterothrophic and euthrophic change is often accompanied by a shift in the macroinvertebrate community, from intolerant, oligosaprobic species upstream the discharge point, to nutrient-tolerant species, indicating an ecosystem degradation (Camargo 1994; Doughty & McPhail 1995; Loch *et al.* 1996; Selong & Helfrich 1998). The macroinvertebrate community needs the longest distance from the point source for recovery. In Appalachian rivers, a 1.5 km flow was not enough for complete recovery (Loch *et al.* 1996), while Camargo (1994) stated 2 km for community recovery in a north Spanish river.

Effect of total suspended solids (TSS)

In case of insufficient effluent treatment, suspended solids from trout farms were deposited in the receiving effluent, leading to interstitial clogging and substrate embeddedness (Carr & Goulder 1993; Doughty & McPhail 1995; Selong & Helfrich 1998). In the deposited sediments, heterothrophic bacteria show profuse growth leading to additional interstitial clogging and deoxygenation (Carr & Goulder 1993; Selong & Helfrich 1998) as well as an increase in colony-forming units (Carr & Goulder 1990).

In general, TSS can also have a direct impact on fish and aquatic life, e.g. a reduction in fish fry survival, fin rot occurrence and gill damage (Bisson & Bilby 1982; Lloyd, Koenings & LaPerriere 1987; Newcombe & McDonald 1991; Selong & Helfrich 1998; Argent & Flebbe 1999; Summerfelt 1999). Total suspended solids concentrations should not exceed 80 $mg\ L^{-1}$ for optimal fish health in freshwater (Summerfelt 1999). Measured TSS concentrations from trout farms outlet vary between 1.6 and 16 $mg\ L^{-1}$ (Table 3). During normal farm operation, TSS do not reach critical values. However, during cleaning and harvesting TSS concentrations up to

256 mg L^{-1} (Michael 2003), 3896 mg L^{-1} or even 14 980 mg L^{-1} were measured (Sindilariu & Reiter 2006).

Oxygen depletion in the effluent

The oxygen consumption through single-pass trout farms, where natural aeration is negligible, is a multi-factorial process. The main factors are the inflowing oxygen from the water supply (1), the supply by artificial aeration (2), the consumption by fish (3), consumption by aerobic heterotrophic microorganisms (4) and for nitrification (5). The total consumption is calculated following Eqs. (2)–(7). As an example, the trout farm from Table 2 is used.

1. Oxygen from the water supply (DO_{in}): For the farm from Table 2, the inflow is oxygen saturated. Assuming a mean water temperature of 10 °C and an altitude of 610 m, the oxygen saturation is 10.46 mg L^{-1} (Lawson 1995).
2. Artificial aeration: The artificial aeration is characterized by the oxygen transfer rate (OTR) generally measured in kg h^{-1}. The OTR is depending on the aeration system used. In the example farm, no artificial aeration was applied.
3. The consumption by fish (DO_{fish}): can be calculated as the difference between in- (DO_{in}) and outflowing (DO_{out}) dissolved oxygen concentration, in a self-cleaning farm where the decomposition of organic matter and nitrification takes place outside the farm.
$$DO_{fish} = DO_{in} - DO_{out} \tag{2}$$
DO_{out} (mg L^{-1}) dependent on the influent supply, artificial aeration and feeding ratio follows Eq. (3) (Lawson 1995).
$$DO_{out} = \frac{\left[(DO_{in} \times Q) + \left(OTR \times \frac{10.000}{36}\right) - \left(FR \times K \times \frac{10}{864}\right)\right]}{Q} \tag{3}$$
K is the amount of oxygen required per kilogram of feed (salmonids 200–220 g kg^{-1}). For the example farm Q = 542 L s^{-1}, FR = 800 kg day^{-1}, K = 220 g kg^{-1} resulting in a DO_{out} of 6.73 mg L^{-1}. 3.76 g mL^{-1} oxygen was consumed by the held fish stock.
4. Consumption by aerobic heterotrophic microorganisms: The oxygen demand for organic matter metabolization is represented by the BOD_5 (mg L^1). The BOD_5 usually does not include oxygen demand for nitrification as this normally occurs after the fifth day of BOD incubation (Tchobanoglous, Burton & Stensel 2003). In the example farm the increase in BOD_5 in the effluent is 4.11 mg L^{-1} (Table 2) or can be estimated following the equation in Table 1, dependent on FR, FW and the quality of the applied feed (Table 2).
5. Consumption through nitrification: The two-step, energy-yielding ammonia oxidation to nitrate is combined with a certain oxygen consumption. Following the stochiometric formulae for the total ammonia oxidation [Eq. (4)], the theoretical oxygen demand for nitrification per milligram total ammonia-N (TAN) can be calculated.
$$NH_4^+ + 2O_2 \rightarrow NO_3^- + 2H^+ + H_2O \tag{4}$$
per milligram ammonia, expressed as TAN, the amount of 4.57 mg O_2 is consumed (Tchobanoglous *et al.* 2003). In natural systems, a part of the TAN is assimilated to cell tissue and the hydrogen ions released from Eq. (4) lead to alkalinity reduction. Through the biological ammonia assimilation, the total oxygen demand is reduced. A combined equation including the ammonia assimilation and alkalinity reduction can be estimated (Tchobanoglous *et al.* 2003).
$$\begin{aligned} NH_4^+ &+ 1.863O_2 + 0.098CO_2 \\ &\rightarrow 0.0196C_5H_7NO_2 + 0.98NO_3^- \\ &\quad + 1.98H^+ + 0.0941H_2O \end{aligned} \tag{5}$$
From the above equation, it will be noted that for each milligram TAN converted, 4.25 mg O_2 are utilized, 0.16 mg of new cells are formed, 7.07 mg of alkalinity as $CaCO_3$ are removed and 0.08 mg of inorganic carbon are utilized in the formulation of new cells (Tchobanoglous *et al.* 2003). For the example fish farm, where the effluent TAN concentration is 0.47 mg L^{-1}, an oxygen demand for nitrification (DO_{TAN}) of 2.0 mg L^{-1} can be calculated. Depending on the daily feeding ratio (FR), FW and the food quality, the DO consumption can be calculated combining the formula from Table 1 for the dissolved nitrogen release and Eqs. (5)–(6)
$$DO_{TAN} = 4.25 \times DN \tag{6}$$

Combining the five factors influencing the DO concentration in the farm effluent, the theoretical remaining DO effluent concentration, after hetrotrophic consumption and nitrification, DO_{rest} can be calculated following Eq. (7):
$$DO_{rest} = DO_{out} - BOD_5 - DO_{TAN} \tag{7}$$
DO_{rest} (mg L^{-1}) is the remaining DO effluent concentration when no aeration occurs and the heterothrophic consumption and nitrification are completed.

For the example farm from Table 2, the water will leave the farm with a DO concentration corresponding somewhat to DO_{out} as the retention time in the farm is only a few minutes and the heterothrophic consumption and nitrification would have just started. However, the effluent has a high oxygen demand of an additional 6.11 mg L^{-1} (BOD_5+DO_{TAN}). Under the theoretical presumption that no aeration will take place in the effluent, a DO_{rest} concentration of only 0.62 mg L^{-1} will remain in the effluent. In the example farm, no additional aeration is needed, but the effluent is prone to oxygen deficiencies.

Dissolved oxygen values below 5 mg L^{-1} lead to a constant distress in fish (Schäperclaus, Kulow & Schreckenbach 1990). Typical decreases in the DO content measured in trout farm effluent are between 1.26 and 3.2 mg L^{-1} (Butz 1990; Boaventura, Pedro, Coimbra & Lencastre 1997; Maillard, Boardman, Nyland & Kuhn 2005; Viadero *et al.* 2005). When the inflowing water is spring or mine water, the effluent DO content can be increased compared with the influent through aeration or oxygenation in the rearing units (Viadero *et al.* 2005).

Direct toxic effects

High ammonia or nitrite concentrations can have direct toxic effects on downstream communities (Stephens & Farris 2004). Elevated ammonia (NH_3) concentrations can lead to blood ammonia intoxication or autointoxication in fish (Schäperclaus *et al.* 1990). High ammonium (NH_4^+) concentrations lead to an ionic imbalance in the blood and acid–base disturbances (Twitchen & Eddy 1994). In Fig. 1, the effect of pH on the relation between ionized and unionized ammonia is given at a temperature of 10 °C. Figure 2 shows the relation dependent on the water temperature at a pH of 8.0. In both figures, lines A and B indicate the maximum concentrations of 6 and 10 μg L^{-1} NH_3 for rainbow trout (*O. mykiss*) successful production for fry and adults respectively (Schäperclaus *et al.* 1990). Line C indicates the unionized ammonia concentration of 25 μg L^{-1} where rainbow trout showed first stress symptoms at a short time exposure (Twitchen & Eddy 1994). Line D indicates the NH_3 concentration of 68 μg L^{-1} where rainbow trout fry has a 24-h LC 50. At line E, the 24-h LC 50 for adult rainbow trout is reached (Rice & Stokes 1975). NH_4^+ as the end product of protein metabolism is found in each aquaculture effluent (Piedrahita 1994; Twitchen & Eddy 1994). Environmental calcium increases the NH_3 concentrations tolerated (Weirich, Tomasso & Smith 1993). The mean total TAN concentrations reported from salmonid aquaculture effluents can be up to 1.6 mg L^{-1} (Dumas & Bergheim 2001).

For aquatic animals, high concentrations of nitrite (NO_2-N) result in increased concentrations of nitrous acid, leading to the oxidation of haemoglobin to methaemoglobin (Wedemeyer & Yasutake 1978; Schäperclaus *et al.* 1990; Tomasso & Carmichael 1991; Weirich *et al.* 1993; Schoore, Simco & Davis 1995; Fontenot,

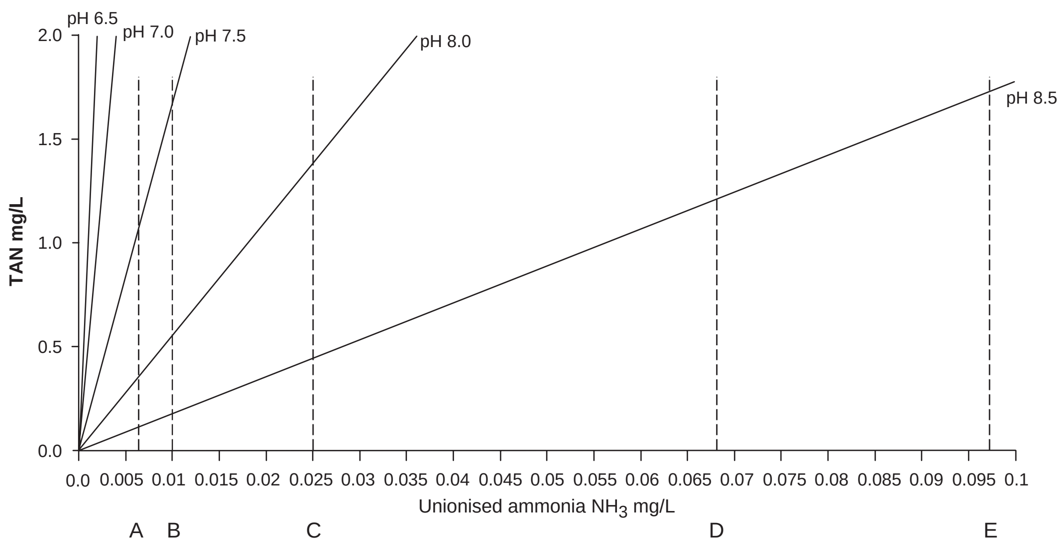

Figure 1 Relation between total ammonia-nitrogen (TAN) and unionized ammonia depending on pH at a temperature of 10 °C. Lines A, B, C, D and E indicate the critical TAN concentrations.

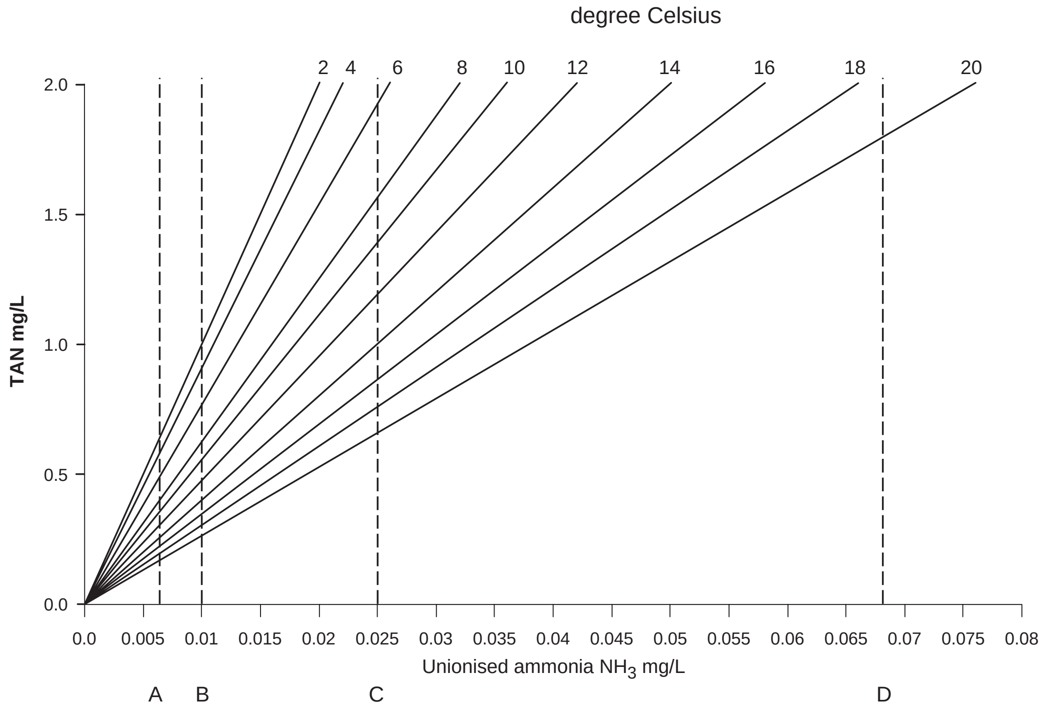

Figure 2 Relation between total ammonia-nitrogen (TAN) and unionized ammonia depending on the temperature at a pH of 8. Lines A, B, C and D indicate the critical TAN concentrations.

Isely & Tomasso 1999). Concentrations between 0.3 and 10.3 mg L^{-1} of nitrite-nitrogen (NO_2-N) had a 96-h LC 50 on juvenile rainbow trout (Wedemeyer & Yasutake 1978). The dissociation between nitrite and nitrous acid and hence the toxic effect depends on the water temperature, pH and conductivity. Increasing temperatures and basic pH reduce the concentrations of nitrous acid and the toxic effect (Schäperclaus *et al*. 1990). Increased conductivity leads to a higher nitrite tolerance (Wedemeyer & Yasutake 1978). Nitrite as the intermediate product of the bacterial nitrification of TAN is ubiquitous in fresh water (Schäperclaus *et al*. 1990). In normal flow-through aquaculture effluents, NO_2-N concentrations are very low, between 0.001 and 0.031 mg L^{-1} (Butz 1990), but in special operating situations like harvesting and cleaning, or in effluents from offline settlement ponds or micro-screen backwash sludge thickeners, NO_2-N can reach concentrations of 0.17 mg L^{-1} (Sindilariu & Reiter 2006).

Impact on wildlife

Besides the impacts on aquatic microbial-, plant- and macroinvertebrate fauna mentioned above, Oberdorff and Porcher (1994) and Prevost (1999) described changes in the stream fish community due to several trout farm effluents discharging in Brittany rivers. The authors found changes in the index of biotic integrity (IBI) based on 10 fish assemblages in correspondence to elevated dissolved nutrient concentrations discharged from trout farms. The fish assemblage changed to pollution-tolerant and exotic species (*Rutilus rutilus* L., *O. mykiss* W.) in the trout aquaculture-influenced areas, while the abundance of pollution-sensitive species (*Cottus gobio* L., *Salmo salar* L.) was reduced (Oberdorff & Porcher 1994; Prevost 1999).

In order to prevent the potentially negative effects of nutrient-rich trout farm effluents on adjacent ecosystems, effluent nutrient management is required. Removal through 'end of pipe' cleaning facilities in addition to other nutrient-reduction strategies through careful site selection, feed and feeding management and water recirculation technologies is needed (Cripps & Bergheim 2000).

Possibilities to reduce the nutrient loads from trout farms effluents

There are two major strategies to reduce the pollution from flow-through aquaculture systems: waste minimization and wastewater reconditioning (Milden & Redding 1998). Partial water recirculation minimizes the waste per kilogram fish produced and improves the efficiency of the end-of-pipe treatment through effluent pre-conditioning (Piedrahita 2003).

Waste minimization strategies

Waste minimization strategies occur in the framework of aquaculture planning, improvements in feed quality and application and on-site management measurements for waste reduction.

Waste minimization and aquaculture planning

Already in the facility-planning stage, the later effluent treatment and potential water recirculation possibilities should be included in order to reduce effluent loads and make the effluent treatment more efficient (Milden & Redding 1998). For example, eliminating onfarm waterfalls increases drum filter efficiency by 33% (Brinker & Rösch 2005), or the use of dual drain tanks to concentrate settable solids into a smaller, more effectively treated flow (Summerfelt *et al.* 2004) leads to an overall improved effluent.

Trout feed improvements

Energy supplementation. High-energy feeds reduce the FCR and the amount of feedstuff applied (Bohl, Ott & Ferling 1992; Cho *et al.* 1994; Heinen, Hankins & Adler 1996; Azevedo, Leeson, Cho & Bureau 2004). Trout feeds with an energy content of 21–27 MJ kg^{-1} are common in trout production. They reach an FCR of 0.8 at FRs of 1% fresh body weight a day. This is an improvement compared with the 1980s where commercial trout feed content was 11–12 MJ kg^{-1} at an FCR of 2–2.5 (Steffens 1986).

Energy:protein ratio. A high energy:protein ratio in the feed reduces the luxury metabolism of proteins for energy supply, leading to reduced nitrogen excretion (Rodehutscord, Mandel & Pfeffer 1994; Rodehutscord 1995; van Weerd, Verástegui & Tijssen 1995; Cai, Wermerskirchen & Adelman 1996; Forsberg 1996; Green *et al.* 2002; Azevedo *et al.* 2004). Actually, crude protein contents of 40–48% are standard in trout grow-out feeds and crude fat contents of more than 20% usual.

Phosphorus contents. The goal is to improve dietary phosphorus availability and reduce phosphorus excretion and losses (Bergheim & Sveier 1995; Gavine, Phillips & Murray 1995; Luzier, Summerfelt & Ketola 1995; Bureau & Cho 1999; Rodehutscord, Gregus & Pfeffer 2000; Sugiura, Babbitt, Dong & Hardy 2000; Vielma, Mäkinen, Ekholm & Koskela 2000; Coloso, King, Fletcher, Hendrix, Subramanyam, Weis & Ferraris 2003). Lellis, Barrows and Hardy (2004) demonstrated a potential reduction in phosphorus losses of 12.5% by P-adapted trout feeding. The P content could be reduced depending on fish size to 0.60% for 200 g fish, to 0.30% for 300 g fish and 0.15% for 400 g fish compared with normal commercial feed (1.2% P) application. A commercial-scale comparison of 1.09% low phosphorus (LP) and 1.50% regular phosphorus (RP) trout feeds resulted in a phosphorus retention of 54% and 43% in the LP and RP diets respectively. Per metric ton feed, 5.01 and 8.55 kg of phosphorus were released into the effluent with the LP and RP diet, respectively, a reduction in 41% phosphorus in the effluent. The LP feed was about 31.1% more expensive than the RP feed (Sugiura, Marchant, Kelsey, Wiggins & Ferraris 2006).

Physical properties. The stability of faecal pellets can be improved by adding binders to the feed. The aim is to increase the average size and stability of faecal pellets in the effluent, leading to a reduced water contact area of the particles and a 40% improvement in 80–100 μm mesh size sieving efficiency (Brinker *et al.* 2005a, b). The addition of a guar gum binder to a commercial extruded feed resulted in a significantly larger faecal particle size, increasing the faecal particle fraction >80 μm (micro-screen mesh size) from 83.1% of all particles (commercial feed) to 87.8%. The larger particles also had an 18% and 8% higher particulate phosphorus and nitrogen content, respectively, in comparison with the commercial feed (Brinker *et al.* 2005b). Guar gum feeds are about 3–5% more expensive than commercial trout feed.

Management actions for waste reduction

The formulation and execution of BMP plans, as outlined by the United States Environmental Protection Agency (USEPA), or codes of conduct and codes of best practice (CBP) as outlined by the Federation of European Aquaculture producers (FEAP) (Ackefors & White 2002; O'Bryen & Lee 2003) reduce the emitted wastes per farm. 'A BMP (CBP) is considered to be the best available and practical means of preventing a particular environmental impact while still allowing production to be conducted in an economically efficient manner' (Boyd 2003). The BMP planning includes plans for the whole facility, as well as plans for feed management, solids control, drugs and chemicals, escape prevention and disease prophylaxes (McMillan *et al.* 2003; Engle *et al.* 2005). For example, the BMP for feed management implies that feed storage, feeding management and feed application are optimized in order to reduce the amount of feed waste, excreted nutrients and rearing costs (Bergheim & Forsberg 1993; Nijhof 1994; Einen, Holmelfjord, Asgard & Talbot 1995; Steffens 1997; Boujard, Labbe & Auperin 2002; McMillan *et al.* 2003).

Some essential management actions to reduce waste output from intensive-run trout farms included in BMPs are outlined by Brinker *et al.* (2006):

1. The application of highly digestible feed, leading to higher fish production and reduced waste emission, see also formulae in Table 1.
2. Reduction in FW e.g. through the application of swimming or low sinking feeds in combination with an automated feed application over a large area.
3. The provision of sufficient oxygen for fish production. Dissolved oxygen concentrations should always be above $6.5\,mg\,L^{-1}$ to avoid stress, suboptimal feed utilization and elevated nutrient excretion.
4. The improvement in the rearing situation especially with respect to fish density, water current, fish size distribution and site quality can reduce stress and elevated excretion in the held stock.
5. The general health status of the stock can essentially influence the feed utilization, feed waste and nutrient excretion.
6. The application of high energy:protein ratio feeds can reduce TAN excretion in fish and prevent potential problems associated with elevated TAN concentrations in the rearing unit, leading to stress and low FCRs.

An example of the combination of different possible strategies to reduce the waste output from a trout farm is given by Brinker *et al.* (2006):

From the site-planning aspect, self-cleaning rearing units, preventing strong water currents and dead zones in the fish tank, as well as the construction of the effluent treatment unit as close as possible to the rearing tanks are recommended. Concerning the food and feeding strategy, Brinker *et al.* (2006) recommend the application of a binder-stabilized, swimming or slowly sinking feed, automatically applied on the whole water surface. As stock management, a sufficient fish density in the rearing units is recommended to maintain the self-cleaning ability of the tanks. The fish density should increase with the fish size to maintain a constant relation between fish surface area and the water volume, as the fish surface area is strongly correlated to the effluent particle size distribution (Brinker & Rösch 2005). Additionally, the farmer should prevent the holding of large fish close to the farm effluent as they negatively influence the particle size distribution (Brinker & Rösch 2005).

Beneath the farm management, water treatment is essential for effluent nutrient reduction.

Wastewater treatment

Domestic wastewater treatment has been well studied for more than 40 years compared with aquaculture effluent treatment (Cripps & Kelly 1996). Owing to low waste concentrations, high flow rates, varying nutrient distribution and effluent quality fluctuations, domestic wastewater treatment plants are not appropriate to treat trout farm effluents (Cripps 1994, 1995; Cripps & Kelly 1995, 1996; Kelly *et al.* 1997). The following treatment methods were used for trout farm effluents:

Physical methods

Physical methods include mechanical and gravitational water treatment used to separate solid particles from suspension (Cripps 1995; Cripps & Kelly 1995; Kelly *et al.* 1997). The particulate solids originate from faeces and uneaten feed (Cripps & Bergheim 2000; Wong & Piedrahita 2003a). Physical treatment methods include:

Micro screens. Micro screens accomplish a mechanical sieving of particles. The screen efficiency is determined by the effluent particle size distribution and the pore size used (Cripps 1995; Brinker *et al.* 2005a). In contrast to drinking water applications, in aquaculture clearing, the screening of filtered particles is an important issue due to the substantially higher particle load of the effluents (Cripps & Bergheim 2000). Different types of screening methods were used:

Static screens. Triangle filters operate by disturbing flow in a thin layer across a weir and onto one side of a flat sieve panel (Summerfelt, Hankins, Weber & Durant 1997). The static screen where the particles are separated and gently transported to the waste trough (see Fig. 3) can be inclined or even. The screen can additionally be washed by intermittent working nozzles (Mäkinen, Lindgren & Eskelinen 1988; Heinen *et al.* 1996; Summerfelt *et al.* 1997). The hydraulic load treated is, beneath the pore size used, dependent on the size of the filter plate installed. In the studies conducted by Mäkinen *et al.* (1988), Heinen *et al.* (1996), the pore sizes used were 65 and 80 μm. The treated hydraulic loads were 8.2 and 5.2 L s^{-1} respectively. Summerfelt *et al.* (1997) used an even filter plate and an 80 μm pore size, in a recirculating trout production unit, with a primary flow of 60 L s^{-1}. Mäkinen *et al.* (1988) measured efficiencies between 77% and 91% for TP, treating a pre-concentrated bottom outflow of circular tanks. Heinen *et al.* (1996) achieved treatment efficiencies of 54–68% for TSS for a raceway overflow, where the TSS concentrations varied between 7 and 11 mg L^{-1}.

Gap or bow-shaped filters are supposed to be self-cleaning, pure gravitational filters (Fig. 4) (Lekang, Bergheim & Dalen 2000; Heerz 2002; Klinkhardt 2005). The hydraulic load is dependent on pore size and filter plate dimensions. Available filter sizes are 62.5 × 25 × 80 cm ($L \times W \times H$) for up to 4 L s^{-1} and 62.5 × 50 × 80 cm ($L \times W \times H$) for the treatment of up to 8 L s^{-1} with a pore size of 200 μm (Heerz 2002). Lekang *et al.* (2000) used a bow-shaped filter with a pore size of 250 μm designed for maximal flow rates of 2–7.2 L s^{-1}. He used the filter for the pre-concentrated bottom drain of a self-cleaning rearing unit with flow rates of only 0.062–0.072 L s^{-1}. The filter showed maximal cleaning efficiencies of 72.1–84.2% for the particulate nutrients: TN, TP, chemical oxygen demand (COD) and TSS. High treatment efficiencies could only be achieved at flow rates lower than 0.067 L s^{-1} and high particle loads of more than 170 mg L^{-1} TSS. In the described application, the filter has to be washed and manually cleaned once a day (Lekang *et al.* 2000). Bow-shaped filters are a favourable alternative for the treatment of pre-concentrated, low-flow effluents.

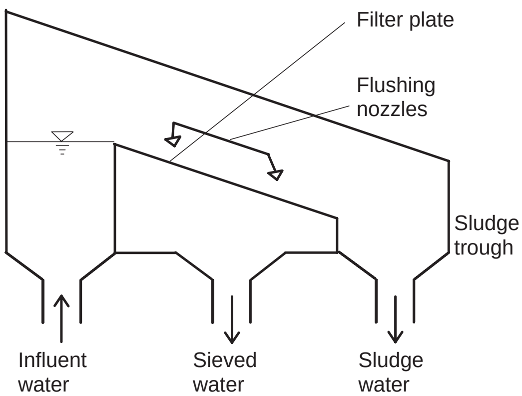

Figure 3 Schematic function principle of a triangle filter (after Mäkinen *et al.* 1988).

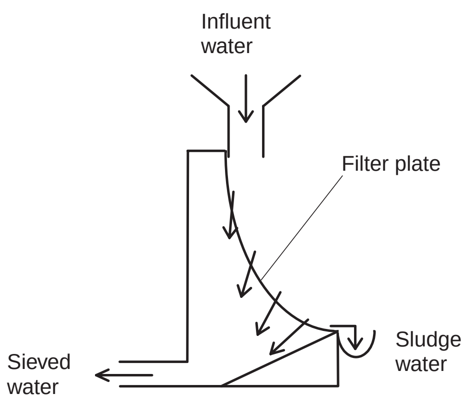

Figure 4 Schematic function principle of a gap filter (after Lekang *et al.* 2000).

Mobile screens. Belt filter are constructed like a conveyor belt made of a screen with an inclination of 10–30° to the water surface. Particulate solids are gently removed, with minimal damage, through the continuous or intermittent movement of the belt screen (see Fig. 5). They are carried by the moving belt out of the water and are scraped off and/or backwashed on the back of the belt (Fladung 1993; Cripps & Bergheim 2000; NN 2000; Ebeling & Rishel 2005; Ebeling, Welsh & Rishel 2006). The filter can be used for complete runoff treatment (Fladung 1993) or as a secondary treatment device (NN 2000; Ebeling & Rishel 2005; Ebeling *et al.* 2006). The filter investigated by Fladung (1993) with a pore size of 100 μm treated 50–125 L s^{-1} trout raceway overflow and cleaning water. The treatment efficiencies were in the ranges of 52–70%, 23–30% and 7–13% for settable solids, COD and TP respectively. Fladung (1993) found filter deficiencies at high hydraulic and particulate loads during cleaning, where the filter was too small. Ebeling *et al.* (2006) used a belt filter for thickening of backwash water from a rotating micro-screen in conjunction with the application of coagulation/flocculation aids. The filter was inclined to 10°, with a mesh size of approximately 120 μm and a belt width of 0.5 m. The configuration managed a hydraulic load of 0.67 L s^{-1} of a highly flocculate effluent. The treatment efficiency of the whole arrangement was 95% and 80% for TSS and TP respectively. The belt filter produced a thickened sludge with 12.6% solids content (Ebeling *et al.* 2006). The use of

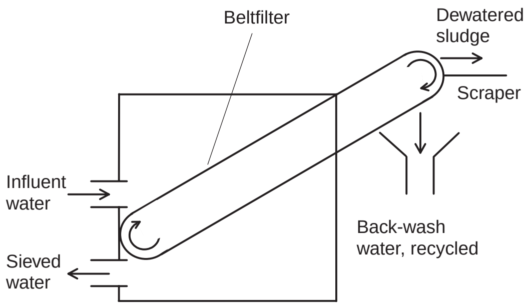

Figure 5 Schematic function principle of a belt filter (after NN 2000).

belt filters for second-stage dewatering and sludge thickening is more appropriate than the treatment of primary farm effluents.

- Drum filter are drum-shaped screens where the effluent flows axially into the drum through the open end and then passes radially to the axis of rotation out through the screen (Cripps 1994). The drum is partially submerged. Above the water level, backwash jets are located on the outside of the drum. Through the drum rotation, the filtered particles are lifted from the water and backwashed into a trough, which directs the sludge out of the drum (Cripps & Kelly 1996).
- The Disc filter consists of one to several flat circular discs of micro-screen material held approximately perpendicular to the primary wastewater flow. If more than one screen is required, the coarse screen is located upstream to remove larger particles and the finest screen is located downstream. Water jets extending across the downstream side of each screen are used for backwashing and to send the separated particles to a sludge-collection trough (Liltved & Hansen 1990; Bergheim & Forsberg 1993; Bergheim, Sanni, Indrevik & Holland 1993; Cripps & Kelly 1995, 1996; Milden & Redding 1998; Cripps & Bergheim 2000).

Rotating micro-screens like disc or drum filters are state of the art in trout farm effluent treatment (Cripps & Bergheim 1997, 2000; Bergheim & Brinker 2003; Rösch *et al.* 2003). They remove a part of the particulate matter in the effluent and the associated nutrients. The suspended solids removal efficiencies range from 10% (Wedekind 1996) to 19% (Bergheim *et al.* 1993) to 65% (Brinker & Rösch 2005) to 70% and 90% (Bergheim, Cripps & Liltved 1998) as a lower treatment efficiency and 75% (Bergheim *et al.* 1998) to 86.8% (Brinker & Rösch 2005) to 91% (Bergheim *et al.* 1993) to 99% (Bergheim *et al.* 1998) and 99.5% (Wedekind 1996) as an upper treatment efficiency. Efficiency depends on the pore size used, the effluent characteristics, especially the particle size distribution (Brinker & Rösch 2005), the concentration of solids entering the filter and the pressure of backwash nozzle flow. Wedekind (1996) and Bergheim *et al.* (1998) achieved maximum efficiencies at high particle concentrations of 305–1000 mg L^{-1}, respectively, and a pore size of 30 μm. Their results indicate that care should be exercised regarding the choice of screen pore size. To estimate the most suitable screen needed, preliminary water quality data combined with flow rate and fish stock data are required, as the investment and operational costs increase exponentially with reduced filter gauze openings (Cripps 1995; Cripps & Kelly 1995, 1996; Summerfelt 1999; Cripps & Bergheim 2000; Wedekind & Göthling 2000; Brinker *et al.* 2005a). The backwash water needed makes up between 0.03% and 1% of the primary water flow, in instances where vacuum, intermittent or continuous backwashing is applied (Bergheim & Forsberg 1993; Bergheim *et al.* 1993, 1998; Cripps & Kelly 1995; Rösch *et al.* 2003). The amount is dependent on the screen gauze opening and system configuration. The resulting backwash water–sludge mix containing about 1000 mg L^{-1} suspended solids (Bergheim *et al.* 1998) has to be further dewatered to reduce the disposal cost. Dewatering systems include sedimentation, filtration and final sedimentation, or flocculation and final belt filtration (Bergheim *et al.* 1998; Cripps & Bergheim 2000; NN 2000; Bergheim & Brinker 2003; Ebeling, Ogden, Sibrell & Rishel 2004; Ebeling & Rishel 2005; Ebeling *et al.* 2006).

The cost increase per kilogram fish produced is 0.79 US$ (0.66€) for large-scale smolt production in Norway (Bergheim *et al.* 1998), 0.64–0.37 US$ (0.53–0.31€) for middle-size US American trout farms (Engle *et al.* 2005), 0.53–0.28 US$ (0.44–0.23€) for small- and middle-size German trout farms, respectively, (Wedekind & Göthling 2000), and 0.04 US$ (0.03€) for large-size US American trout farms (Engle *et al.* 2005). However, even at the low additional micro screen costs, large US American farms are not willing to make the investment, due to the tight farm profitability margins (Engle *et al.* 2005). High fixed investment costs dominated the total costs (between 85% and 89%). The increased production costs can be partly counterbalanced by a further production expansion or better market prices (Wedekind 1996; Bergheim *et al.* 1998; Wedekind & Göthling 2000; Engle *et al.* 2005).

Media filtration. Media filters mechanically block the effluent particles in a granular filter matrix of sand, gravel or artificial material. The filter can be vacuum, pressure or gravity charged. They are widely used in municipal wastewater treatment and drinking water conditioning (Scherb, Bauer & Mayer 1984), as well as in recirculation aquaculture systems (Chen, Timmons, Aneshansley & Bisogni 1993a; Cripps & Kelly 1996; Cripps & Bergheim 2000; Palacios & Timmons 2001). Except for several public hatcheries in North America and some trout farms in Germany, this kind of filter is rarely used for trout farm effluent treatment, due to its flow rate limitations. The filter needs regular backwash leading to discontinuous operation unless several parallel filters are installed (Cripps & Kelly 1996; Kristiansen & Cripps 1996; Wedekind 1996). An adequately dimensioned media filter has two important advantages:

1. Like the micro screen pore size, the filter media used can be closely adjusted to the effluent characteristics, removing the required particle size (True, Johnson & Chen 2004c), (Oehler 1982)

$$R_m = 6.5 \times R_p \qquad (8)$$

where R_m is the radius of the used filter media and R_p the radius of the effluent particles.

A filter with a small media radius can even be used as disease prophylaxes in trout farms removing bacteria (Arndt & Wagner 2003). But also the hydraulic conductivity of the filter material is, beneath the filter shape, dependent on the particle size (Bahlo & Wach 1996), Eqs. (9), (10)

$$Q = k_f \times A_f \times I \times 1000\,(\mathrm{L\,m^{-3}}) \qquad (9)$$

where Q is the water inflow in $\mathrm{L\,s^{-1}}$, k_f is the hydraulic conductivity of the filter material ($\mathrm{ms^{-1}}$), A_f is the filter infiltration area ($\mathrm{m^2}$) and I is the hydraulic filter gradient (dimensionless). As an approximation for the hydraulic conductivity of a filter material, the following equation can be used (Bahlo & Wach 1996).

$$k_f = (d_{10})^2/100 \qquad (10)$$

where d_{10} is the effective grain size of the filter material in millimetres, defined as the lowest 10% in the grading curve of the material. Fine material exponentially decreases the hydraulic conductivity.

2. The combination of mechanical separation with biological treatment reduces the BOD and a part of the dissolved nutrients. The application is used especially in recirculating aquaculture (Paller & Lewis 1982; Nijhof & Bovendeur 1990; Cripps & Kelly 1996; Cripps & Bergheim 2000; Andreasen 2003). The dual function of biological and physical treatment processes can be problematic (Golz, Rusch & Malone 1999; Cripps & Bergheim 2000). For physical particle separation, regular backwashing is needed to dislodge the trapped particles and retain the hydraulic permeability. The biological treatment needs the buildup of a biofilm that is disturbed through regular backwashing (Golz *et al.* 1999). An additional gas exchange might be needed when high biological treatment rates need to be achieved.

Reticulated foam is used as filter media in an experimental trout raceway effluent treatment. The trials showed a head loss of 150 mm and 29% and an 11% reduction in TSS and TP respectively (True *at al.* 2004c). Sand media filtration is also used for the stabilization of micro-screen backwash water from salmon aquaculture. Backwashing and renewal intervals of 2–3 months at low hydraulic loads of 10–60 $\mathrm{L\,m^{-2}\,day^{-1}}$ were needed (Kristiansen & Cripps 1996). Granular media filters were used as a last resort in aquaculture as they require a complex backwash mechanism and produce relatively large fixed and operating costs per flow unit treated (Summerfelt 1999).

Sedimentation. Sedimentation relies upon the density differences between particulate waste and the surrounding water. Gravitational forces, in the absence of other confounding influences, lead to particulate waste with a specific gravity $> 1.0\,\mathrm{g\,cm^{-3}}$ to sink (see Fig. 6) (Cripps & Kelly 1996).

The settlement velocity of suspended solids depends on the particle surface and dimension, its specific weight and the flow velocity of the surrounding water. For detailed particle settling theory, see also Tchobanoglous *et al.* (2003). To increase the sedimentation efficiency, the flow velocity has to be lowered, the retention time in the sedimentation pond increased, the sedimentation distance shortened and, if possible, the specific size or weight of the particles has to be increased. Sedimentation velocity [V_s ($\mathrm{ms^{-1}}$)] of effluent particles can be calculated dependent on the particle density [ρ_p ($\mathrm{kg\,m^{-3}}$)] and size [s_p (m)] (Tchobanoglous *et al.* 2003). Assuming a Reynolds number RN < 0.3 Eq. (11) can be used:

$$V_s = \frac{g \times (\rho_p - \rho_w) \times s_p^2}{18\mu} \qquad (11)$$

where g is the gravity term ($\mathrm{ms^{-2}}$), ρ_w the water density ($\mathrm{kg\,m^{-3}}$) and μ the water viscosity ($\mathrm{Ns\,m^{-2}}$).

Flow velocities lower than 0.067 $\mathrm{m\,s^{-1}}$ and preferably at 0.017 $\mathrm{m\,s^{-1}}$ (Henderson & Bromage 1988), or more conservatively 0.02–0.04 $\mathrm{m\,s^{-1}}$ (Hansen 1979)

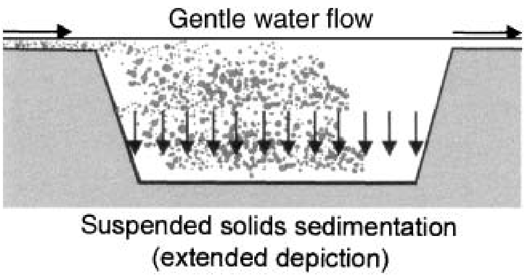

Figure 6 Schematic depiction of suspended solids sedimentation (adapted from Rösch *et al.* 2003).

and retention times of atleast 30 min (Cripps & Bergheim 2000) were recommended for trout farm effluents. Overflow rates of about 1.0–2.7 $m^3 m^{-2} h^{-1}$ produce adequate solids settlement depending on the tank design (Bergheim *et al.* 1998; Cripps & Bergheim 2000).

In settlement basins, four zones can be identified: inlet, settling, sludge and outlet (Hansen 1979). Baffles and outlet weirs are often incorporated to promote quiescent conditions (Cripps & Bergheim 2000). The fast separation and removal of settled sludge from the primary flow is the most important issue. High re-suspension and leaching rates of the settled nutrients, especially in anaerobic environments, are the main problems of sedimentation basins (Lefebvre, Bacher, Meuret & Hussenot 2001; Stewart *et al.* 2006a).

The development of sedimentation basins starts from simple ponds dug downstream to the farm, to compact second-stage cones or advanced basins with an optimized depth/length relation and an ideal flow regime, incorporating automatic sludge removal, flow manipulation, circular flow or inclined plates or tubes for accelerated settlement (Meylahn 1983; Henderson & Bromage 1988; Henderson, Bromage & Watret 1989; Cripps & Kelly 1996; Summerfelt 1999; Cripps & Bergheim 2000; Lekang, Bomo & Svendsen 2001; Baldwin 2002a; Davidson & Summerfelt 2005; Mayer 2005), or the implementation of artificial mats as particle traps for better removal and reduced re-suspension (Stewart, Boardman & Helfrich 2006b).

The simplicity of the process and the availability of suitable chambers made sedimentation one of the first wastewater treatment methods in aquaculture (Cripps & Kelly 1996). Sedimentation is used in all kinds of aquaculture production facilities from flow-through salmonid production (Hansen 1979; Gefken 1987; Henderson & Bromage 1988; Igler 1995; Wong & Piedrahita 2000, 2003b; McMillan *et al.* 2003; True, Johnson & Chen 2004a, b) to cold and warm water recirculation facilities (Lekang *et al.* 2001; Davidson & Summerfelt 2005), warm water catfish production (Engle & Valderama 2003) and shrimp ponds (Backhurst & Harker 1988; Teichert-Coddington, Rouse, Potts & Boyd 1999; Halide, Ridd, Peterson & Foster 2003; Jackson, Preston, Burford & Thompson 2003).

Using the data of Chen *et al.* (1993a) for the density of suspended particles in aquaculture facilities ($\rho_p = 1190\ kg\ m^{-3}$) in Eq. (11), only particles larger than 145 μm can be removed from the effluent in a sedimentation basin of 1 m depth, with a retention time of 20 min at 10 °C. Assuming a particle size distribution as measured by Brinker *et al.* (2006) for trout farms, this would mean a treatment efficiency of about 60% TSS. For the example trout farm from Table 2, at a flow rate of 542 $L\ s^{-1}$, the sedimentation basin should have an area of atleast 650 m^{-2} to achieve a treatment efficiency of 60%. Equation (11) is only valid for ideal sedimentation basins, where no short circuits or particle resuspension occur and the deposited particles are regularly removed. As the ideal conditions are hard to achieve in commercial farms, sedimentation is rarely suitable to treat the primary effluent from flow-through trout farms due to inadequate flow dynamics and sludge removal problems. High flow rates cause insufficient residence time, scouring of already settled particles and short circuiting. Henderson and Bromage (1988), Dumas and Bergheim (2001), Piedrahita (2003) suggest that sedimentation is only effective at solid loads $>6\ mg\ L^{-1}$ and large particle size. Findings of Brinker *et al.* (2005b), however, showed a positive treatment effect of a final 'maturation pond' where the effluent of trout raceways already mechanically sieved at 80 μm was passed through before being discharged in the receiving brook. The maturation pond had a retention time of about 3 h at a water temperature of 10–11 °C and reduced the TSS from 3.2 to 1.5 $mg\ L^{-1}$ by means of sedimentation.

The use of sedimentation is not inherently wrong; highly effective plate separators were successfully applied (Lawson 1995). The pre-concentration of suspended solids within the rearing tank through cyclic flow regimes, using the so-called 'tea cup effect', leads to higher effluent treatment efficiencies (Wang 1994; Cripps & Bergheim 1997; Baldwin 2002b; Wong & Piedrahita 2003b; Summerfelt *et al.* 2004; Davidson & Summerfelt 2004, 2005). The use of second stage–stage dewatering of separated, higher concentration sludge is also appropriate and is commonly applied

(McMillan *et al.* 2003; True *et al.* 2004a, b). But other applications are sometimes inadequate (Henderson & Bromage 1988; Cripps & Bergheim 2000).

Other mechanical treatment methods. Other mechanical treatment methods used in aquaculture are membrane filtration and flotation (foam fractioning). As they remove mainly small particles from the effluent, these treatment methods are not relevant for trout aquaculture. They were applied only in recirulating systems where the accumulation of fine particles, due to insufficient removal, can cause problems for the held stock (Chen, Timmons Bisogni Jr & Aneshansley 1993b; McMillan *et al.* 2003; Patterson & Watts 2003a, b; Patterson, Watts & Gill 2003). Mechanical effluent treatment acts exclusively on particles and particle-bound nutrients. Depending on the method used, different size fractions were removed (Fig. 7). Chemical or biological methods have to be used to treat the dissolved nutrients.

Addition of chemicals for effluent treatment

The addition of chemicals is used for the following processes in municipal wastewater treatment: (1) chemical coagulation and flocculation, (2) chemical precipitation, (3) chemical disinfection, (4) chemical oxidation, (5) advanced oxidation processes, (6) ion exchange and (7) chemical neutralization, scale control and stabilization (Tchobanoglous *et al.* 2003). For trout aquaculture effluent treatment, only coagulation, flocculation and precipitation processes (1, 2) were used. Chemical disinfection, oxidation and advanced oxidation processes (3, 4 and 5) were only used in recirculating aquaculture, or when very high effluent standards are set (Liltved & Landfald 1995; Liltved & Cripps 1999; Summerfelt 2003). Ion exchange processes (6) are not suitable either for trout aquaculture or for recirculation units. Chemicals for stabilization or neutralization (7) are applied in salmonid aquaculture for the stabilization of already dewatered sludge with lime, before land application (Bergheim *et al.* 1998), or when partial recirculation is used pH stabilization with chemicals like sodium carbonate may be necessary (Ulgenes & Ludin 2003). In the following, only the relevant processes used in trout aquaculture, coagulation, flocculation and precipitation (1, 2), were described.

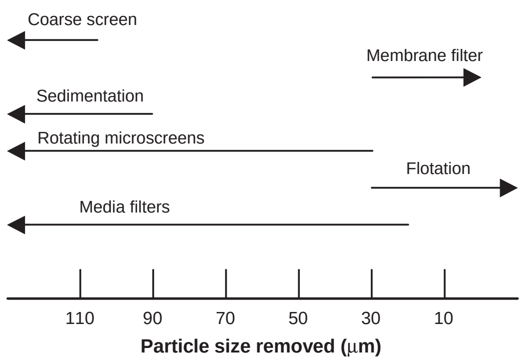

Figure 7 Particle size removed by different solids separation processes (adapted from Cripps & Bergheim 2000).

Similar electric charges in small particles in water cause the particles to repel each other and keep the small, colloidal particles apart and in suspension. There are three possible processes to solve this problem (Ebeling, Sibrell, Ogden & Summerfelt 2003; Tchobanoglous *et al.* 2003; Ebeling *et al.* 2004):

1. *Coagulation*: The coagulation process neutralizes or reduces the negative charge on the particles. This allows the van der Waals force of attraction to encourage initial aggregation of colloidal and fine-suspended materials to form microflocs.
2. *Flocculation*: Flocculation is the process of bringing together the microfloc particles to form large agglomerates by physical mixing and through the binding action of flocculants, such as long-chain polymers.
3. *Precipitation*: Precipitation is the process where two soluble reagents like phosphorus, in combination with Al^{3+} or Fe^{3+}, react to an insoluble, particulate complex that settles down.

While coagulation/flocculation are physical processes on the molecular level, precipitation is based on chemical reactions. One of the most commonly used methods to remove suspended solids and phosphorus in drinking water and municipal wastewater is the application of coagulants/flocculants, the addition of coagulation aids, which both precipitate soluble phosphorus to a solid complex, and the addition of flocculants (Overath 1978; Boller 1984; Gleisberg 1988; Huber 1993; Schwörbel 1993; Ebeling *et al.* 2003). Common coagulants/flocculants are alum ($Al_2(SO_4)_3$) and ferric chloride ($FeCl_3$). Lime ($Ca(HCO_3)_2$) is used as a coagulation aid while organic polymers are usually used as flocculants. Alum or ferric chloride should be used in quantities of 30–90 mg L^{-1}, and coagulation aids in quantities of 15–20 mg L^{-1} for effective effluent treatment (Cripps 1994; Ebeling, Rishel & Sibrell 2005). Coagulation/flocculation and precipitation are possible as trout farm effluent treatment but were not considered to be economically viable (Cripps 1994; Ebeling *et al.* 2003, 2004, 2005). The application is realistic for the treatment of relatively small and concentrated backwash flows from trout farm primary treatment (Cripps 1994; Ebeling *et al.* 2003, 2004, 2005). Assuming actual prices of about 540 US$ (448€) and 730–2050 US$ (606–1700€) per metric ton of alum or ferric chloride and coagulation aids, respectively, and flow volumes of $50\,L\,s^{-1}$, the treatment of the whole effluent entails a financial effort of atleast 47 US$ (39€) a day, only for the chemicals. At flow rates of several hundreds $L\,s^{-1}$, usual for a trout aquaculture, this amount increases with increased flow rate. The use of these chemicals for secondary treatment of e.g. micro-screen backwash sludge or sedimentation sludge for further thickening and dewatering is economically feasible (Cripps & Bergheim 2000; NN 2000; Ebeling *et al.* 2004, 2005; Ebeling & Rishel 2005) and has shown efficiencies of up to 99% reduction in suspended solids and 97% phosphate removal (Ebeling *et al.* 2003, 2004, 2005, 2006). With a micro-screen backwash volume of about 1% of the effluent also, the costs for coagulation/flocculation are only about 1% of the treatment costs of the whole effluent. If flocculation/coagulation aids are applied, the potential fish toxicity of the used chemicals has to be taken into consideration, especially of aluminium-containing materials (Schwörbel 1993).

Biological methods

Biological treatment systems are primarily used to transform dissolved and particulate biodegradable constituents into acceptable end products and to transform or remove dissolved nutrients by the metabolic process of microbial communities and plants (Landau & Scarpa 2001; Tchobanoglous *et al.* 2003). Three important processes involved in biological effluent treatment are:

- Respiration, the conversion of organic material to CO_2 under oxygen consumption.
- Nitrification and denitrification, microbial processes where TAN is converted in a first step to NO_2 then to NO_3 under oxic conditions and afterwards under anoxic conditions the NO_3 is converted to N_2 (Platzer 1998; Summerfelt 1999; Tal, Watts, Schreier, Sowers & Schreier 2003).
- Consumption and storage of soluble phosphorus. Plants and micro organisms need phosphorus for their growth, some of them even store phosphorus, transforming soluble phosphorus to particulate (Schwörbel 1993; van Rijn 1996; Milden & Redding 1998; Barak & van Rijn 2000; Barak, Cytyn, Gelfand, Krom & van Rijn 2003).

The different methods of biological effluent treatment can be classified according to their degree of mechanization.

Technical methods of biological effluent treatment. All technical methods of biological effluent treatment work with a substrate-attached microbial film (Summerfelt 1999). They are characterized by a high level of mechanization and consumption of external energy, e.g. for the cleaning process, pumping and recirculation of biologically active sludge, active aeration, use of artificial substrates for the microbial communities and active backwashing (Schwörbel 1993). The methods are highly effective and can be easily dimensioned (Eding, Kamstra, Verreth, Huisman & Klapwijk 2006; Summerfelt 2006; Timmons, Holder & Ebeling 2006; Watten & Sibrell 2006).

There are seven basic technical configurations within this category: activated sludge treatment, trickling filter, submerged filter, rotating-media filter, moving bed filter, fluidized bed filter and low-density media filter. To the author's knowledge, only submerged filters have been used until now for the high-end treatment of flow-through trout aquaculture effluents, an example reported by Bergheim and Brinker (2003), Brinker *et al.* (2006).

Submerged filters are packed with high surface, high hydraulic conductive filter media. The wastewater passes the fully submerged filter media and the nutrients are taken up by the attached microbial flora and were transformed. This two-phase system (water and medium) is prone to oxygen shortages, short circuiting, and especially in flow-through effluents, to particulate waste deposition (Paller & Lewis 1982; Rogers & Klemetson 1985; Milden & Redding 1998; Summerfelt 1999; Rida & Curz 2001; Zhu & Chen 2001a, b, 2003). Submerged filters, in combination with a sparge column for oxygen removal and methanol dosing, were used for controlled denitrification in recirculating aquaculture systems (Lee, Lea, Dohmann, Prebilsky, Turk, Ying & Whitson 2000). In flow-through facilities where the effluent standards are very strict (Bergheim & Brinker 2003) or water reconditioning for partial reuse is needed (Andreasen 2003; Mayer 2005), submerged filters can be used.

In the system described by Bergheim and Brinker (2003), and Brinker *et al.* (2006), the filter is passed horizontally by the micro-screen pre-treated total effluent of a large flow-through trout farm. The total production was 600–1000 mt year^{-1} at $Q = 500\,L\,s^{-1}$. The filter configuration is a two-chamber system with an antechamber for pre-treatment and BOD metabolization, and the main chamber for nitrification. For external cleaning, small blocs can be extracted from the filter. The filter was dominated by heterotrophic bacteria due to high BOD loads in the effluent. Heterotrophic bacteria suppressed the development of nitrifying bacteria, leading to almost no reduction in TAN loads. The filter was effective for BOD, TP and TN reduction; 48.6%, 18.5% and 31.9% treatment efficiencies were achieved (Bergheim & Brinker 2003). For effective TAN reduction, several improvements in filter cleaning and water pre-treatment have to be made.

Semi-technical methods of biological effluent treatment. The application of this kind of biological treatment in trout farm effluents ranges from polishing ponds, over surface flow (SF)-constructed wetlands and artificial ditches to sub-surface flow (SSF)-constructed wetlands.

Polishing ponds. Polishing ponds for flow-through farms combine mechanical sedimentation (Henderson & Bromage 1988) with biological treatment (Milden & Redding 1998; Schulz 2004). The amount of biological treatment is dependent on the turnover rate and the microbial active surface area in the pond. Brinker *et al.* (2005b) showed, for a polishing pond ('maturation pond'), receiving an already micro-filtered trout farm effluent, treatment efficiencies of 54% for TSS and 35% for TP in the growing season. The fact that the pond showed no leaching of soluble nutrients may be due to uptake processes of plants and microorganisms (Brinker *et al.* 2005b). The treatment efficiency depends primarily on the pond overflow rate. The transitions between polishing ponds and SF-constructed wetlands are smooth. In contrast to settling tanks, polishing ponds have natural embankments, high water retention times of atleast 2 h and are seldom cleaned, about once/twice a year (Schulz 2004).

Surface flow-constructed wetland. Surface flow-constructed wetlands are ponds where the growth of floating, submerged and especially emerged macrophytes improves the effluent treatment efficiency. The macrophytes remove nutrients, reduce currents, increase solid sedimentation and are an additional growth surface for the microbial biofilm (Ng, Sim, Ong, Kho, Ho, Tay & Goh 1990; Halide *et al.* 2003; Schulz 2004; Schulz, Gelbrecht & Rennert 2004). In marine SF wetlands, mainly seaweed is applied for nutrient removal, with the advantage of additional seaweed production and income increase (Ellner, Neori, Krom, Tsia & Easterling 1996; Neori, Krom, Ellner, Boyd, Popper, Rabinovitch, Davison, Dvir, Zuber, Ucko, Angel & Hillel 1996; Porello, Ferrari, Lenzi & Persia 2003; Porello, Lenzi, Persia, Tomassetti & Fiona 2003; Porello, Lenzi, Tomassetti, Persia, Fiona & Mercatli 2003). In freshwater aquaculture, several macrophytes are applied for effluent treatment with further use or disposal. Only *Nasturium officinale*, grown in trout effluents and harvested for human consumption, has been applied for flow-through facilities (Mayer 1995). Normally, naturally established plant communities are applied as a treatment aid, mainly reeds (Schulz 2004; Schulz *et al.* 2004). Schulz *et al.* (2004) sampled three SF wetlands with a surface area of 350 m^2 and a volume of 210 m^3 receiving 5, 10 and 15 $L\,s^{-1}$ of trout raceway runoff. He found treatment efficiencies in the range of 72–66% for TSS, 30–31% for COD, 53–41% for TP, 18–3% for soluble reactive phosphorus (SRP), 30–19% for TN and 60–41% for TAN. The treatment efficiency was dependent on the overflow rate. The examined SF wetlands all had a significant treatment effect on the effluent (Schulz *et al.* 2004).

Artificial ditches. Artificial ditches were applied to treat the aquaculture effluent, taking advantage of the natural cleaning abilities of running waters like:

- physical sedimentation and screening of particulate nutrients and
- chemical and biological flocculation, coagulation, absorption, transformation and consumption processes of suspended particulate and dissolved nutrients.

The processes are mediated through biologically and chemically active surfaces in the hyporheic interstitial or on macrophytes and benthic surfaces (Rietz 1972).

In trout aquaculture (Ganseneder 1996) and warm water pond aquaculture, (Shireman & Cichra 1994; Frimpong, Lochmann & Stone 2003) applied artificial ditches for effluent treatment, in cases where the morphological and space conditions were not favourable for the construction of polishing ponds or SF wetlands. For this kind of treatment, it is important not to overestimate the cleaning ability of running waters. The denitrification performance of running waters is about 0.2 g N m^{-2} day^{-1} (Wolf, Ostertag & Eck-Düpont 1989). For the other relevant parameters, no reliable data are available.

Subsurface-flow-constructed wetlands. Subsurface-flow-constructed wetlands consist of a media filter planted with macrophytes. The mechanical treatment through the filter media, the biological treatment through the microbial community attached on the media grains and the effect of the emergent macrophytes lead to the overall treatment effect. In municipal wastewater treatment, SSF wetlands are widely discussed, investigated and used. Factors like the most effective filter substrate, suitable macrophytes and the maximum amount of treatable pollution equivalents (60 g BOD_5 day^{-1}) were examined. An overview of the construction fundamentals is available in Kadlec and Knight (1996), Bahlo and Wach (1996), Ambros, Erhardt and Kerschbaumer (1998), Bucksteeg, Grosche Kollatsch, Lützner, Maus, Rosenwinkel, Schröder, Schweitzer, Tiedtke, Voss, Zerres, Börner and Hegemann (1998), Kadlec, Knight, Vymazal, Brix, Cooper and Haberl (2001), Wissing and Hofmann (2002). For municipal wastewater, SSF wetlands are a low-maintenance and highly effective treatment method for low to medium polluted wastewater (Platzer 1998). For trout aquaculture, SSF wetlands may be a suitable treatment method, especially for micro-screen backwash (Lekang, Skjelhaugen & Jennsen 1997). However, until now SSF wetlands have only been evaluated at an experimental level (Milden & Redding 1998). Sub-surface flow wetlands were used for: secondary treatment of micro-screen backwash sludge (Summerfelt, Adler, Glenn & Kretschmann 1999; Comeau, Brisson, Réville, Forget & Drioz 2001), primary settlement sludge (Michael 2003), experimental scale raceway runoff (Schulz and Rennert 2000; Schulz, Gelbrecht & Rennert 2003; Schulz 2004) and artificial aquaculture effluents (Naylor, Brisson, Labelle & Comeau 2003). All experiments showed high treatment efficiencies from 50% to nearly 100% removal of dissolved and particulate effluent parameters respectively. The treatment efficiency depended mainly on the kind of effluent treated (primary farm effluent or secondary effluent of a treatment unit) and on the wetland retention time. The wetlands treating low volumes of highly polluted secondary effluents reached the highest efficiencies with 75% for TAN, 89% for Kjeldal N, 93% for PO_4-P, 82–90% for TP, 81% for BOD_5 and 91–97% for TSS (Summerfelt *et al.* 1999; Comeau *et al.* 2001; Michael 2003). Wetlands treating the primary farm effluent reached much lower efficiencies of 50–87% for TAN, 4–26% for TN, -24–13% for PO_4-P, 39–68% for TP, 37–49% for BOD_5 and 35–97% for TSS (Schulz *et al.* 2003; Sindilariu & Reiter 2006). The reported operations have a high up-scaling potential, but no test was conducted for commercial-scale trout farm effluents. The potential up-scale problems should be the subject of further research.

Active bivalve filtration. Another semi technical alternative is the application of active filtration. In saltwater, nutrient removal through pacific oyster filtration, Sydney rock oyster, oyster and abalone has already been successfully tested for marine aquaculture effluents. The animals were used for secondary treatment of the effluent of oxidation ponds (Shpigel, Lee, Soohoo, Fridman & Gordin 1993a; Shpigel & Neori 1995; Hussenot, Lefebvre & Brossard 1998; Neori, Ragg & Shpigel 1998; Neori, Shpigel & Ben-Ezra 2000), for secondary treatment of sedimentation ponds overflow (Wang 1990; Shpigel, Neori, Popper & Gordin 1993b; Jones, Dennison & Preston 2001) or for primary effluent treatment (Jones & Iwama 1991; Shpigel & Blaylock 1991; Jakob, Pruder & Wang 1993; Shpigel *et al.* 1993b; Shpigel, Gasith & Kimmel 1997; Jones & Preston 1999). The animals removed suspended faecal particles and phytoplanktonic algae. Effluent turbidity reduction ranging between 57% and 97% in settlement tanks was achieved, compared with 10–16% turbidity reduction with bivalve shells only in the tank (Shpigel *et al.* 1997). *Saccostrea commercialis* (Sydney rock oyster) reduced the concentration of total Kjeldal N to 28% and total phosphorus to 14% of the initial concentration of settled shrimp aquaculture effluents (Jones *et al.* 2001). The supplemental oyster or abalone production created a secondary marketable product.

In freshwater, potentially suitable species like *Anodonta anatina*, *Unio tumids* or *Unio grassus* have not been used as a treatment method until now. It has been noted that mussel close to freshwater trout cages have a positive effect on pollution beyond the cages (Karayücel & Karayücel 1998; Soto & Mena 1999). The use of mussels as a secondary treatment of micro-screen backwash water or as a final treatment for primary sedimentation is reasonable due to filtration rates of 0.35 L h^{-1} g^{-1} dry body weight (Pusch, Siefert & Walz 2001). For freshwater molluscs, dry weights of about 2.5–20 g $individual^{-1}$ are realistic. The active filtration and coagulation capacity of the animals is enormous, also accelerating the mechanical sedimentation (Köthe 1992; Pusch *et al.* 2001; Riisgard 2001). As an additional effect, the animals remove the nutrients they need for growth. The application of molluscs for biological effluent treatment is possible, but further research on practical realization is needed. A potential problem is the parasitic life stage of freshwater molluscs in fish (Liltved & Hansen 1990).

Natural methods of biological effluent treatment. Natural methods include artificial bacteria augmentation for accelerated microbial turnover (Chiayvareesajja & Boyd 1993; Dumas, Laliberte, Lessard & de la Noüe 1998; Chen & Chen 2001) and land application of aquaculture effluents like irrigation or sprinkling (Schwörbel 1993; Pardue, DeLaune, Patrick & Nyman 1994; Gathe, Burtle, Vellidis & Newton 1997; Brown, Glenn, Fitzsimmons & Smith 1999; McIntosh & Fitzsimmons 2003). Artificial bacteria augmentation aims to reduce nutrient concentrations in the effluent through higher microbial turnover rates and microbial yields (Dumas *et al.* 1998; Chen & Chen 2001). The method has found no application in trout effluent treatment, even though Dumas *et al.* (1998) demonstrated in an experiment with the free-floating cyanobacterium *Phormidium bohneri* removal efficiencies of 82% and 85% for TAN and PO_4-P, respectively, for trout farm effluent treatment. Problems arise due to very short water-retention times in the farm, leading to bacteria washout and unfavourable temperature conditions, reducing the bacterial turnover rates drastically (Dumas *et al.* 1998).

Wastewater can be applied to crop- or grassland. From trout and salmon farming, thickened microscreen backwash sludge (Bergheim *et al.* 1998; Bergheim & Brinker 2003) and thickened sedimentation sludge (Sindilariu & Reiter 2006) are used as agricultural manures. Trout sludge application showed increased potato and grassland yields compared with no fertilizer application and similar yields compared with inorganic fertilizer application (Donaldson & Chadwick 2006). The land application of primary effluents from flow-through trout production is not suitable, as the volume is too large. The on-land application of thickened trout sludge is common in central Europe.

Partial recirculation for effluent improvement

In the last few years, a third strategy for effluent improvement, partial water recirculation in former flow-through farms, has been applied (Andreasen 2003; Summerfelt *et al.* 2004; Mayer 2005). For partial recirculation, the reused water has to be lifted and if the water quality is not optimal, atleast it has to be aerated or oxygenated (Summerfelt *et al.* 2004). Partial recirculation has several advantages for the farm operator (Piedrahita 2003):

1. a more intensive water use, with higher fish production per unit water consumed,
2. a higher nutrient concentration in the farm effluent, more suitable for high-end effluent treatment, improving the efficiency of the final treatment device and
3. a possible inclusion of biological treatment units in the recirculation system, aiming to reduce the dissolved nutrients in the effluent.

The potential disadvantages are:

1. elevated energetic costs for water pumping or lifting and
2. high water exchange rates compared with closed recirculation systems,with low control on the environmental conditions as suboptimal temperature for biofiltration and seasonal changes in water quality.

Especially in Denmark, the restricted water uptake from natural rivers and limits of the amount of nutrients discharged per farm have forced trout producers to switch to partial recirculation at former flow-through sites (Andreasen 2003).

Initially, a trickling filter was installed to treat salmonid raceway effluents, to improve the water quality before reuse (Liao & Mayo 1972, 1974). For water recirculation in trout farms, actually trickling and moving bed reactors were used.

Trickling filter

Trickling filters are columns packed with a media of high surface area and high hydraulic conductivity, for microbial settlement. The cleaning principle is microbial decomposition. Wastewater is introduced at the column top, by means of a rotating spraybar, a distribution grid or a tipping trough and trickles down through the filter bed. The filter works as a three-phase system (air/water/solid). The filter is less likely to suffer oxygen deficiency a reduction in performance and blockage. Additionally, trickling filters provide good degassing qualities. An overview of trickling filter application, planning and calculation in warm water recirculation systems has been given by Eding *et al.* (2006). Plastic rings or blocks of different forms and shapes were used as filter media. Cross-flow and vertical flow media should be preferred compared with random flow media, due to lower clogging risks (Eding *et al.* 2006). Sometimes, even clay aggregates and biological materials, e.g. straw are used as filter media (Schmitz-Schlang &

Hoogen 1992; Lekang & Kleppe 2000). In partial re-use systems for trout production, trickling filters were applied mainly for degassing, oxygenation and atleast TAN removal (Mayer 2005). A system configuration for trout farm partial recirculation is shown in Fig. 8. The filter design procedure was based on the findings of Liao and Mayo (1972, 1974) and Eding *et al.* (2006), where the TAN removal rate [N_{AR} (g TAN m^{-2} day^{-1})] is a function of the TAN loading rate [A_L (g TAN m^{-2} day^{-1})], media retention time [t_m (h)] and temperature [T (°C)]:

$$N_{AR} = A_L \times t_m \times (9.8 \times T - 21.7) \times 0.01 \quad (12)$$

when given conditions are followed (surface loading 86.4–147 m day^{-1}, pH 7.5–8, media retention time 0.26–0.46 h, TAN concentration maximum 1 mg L^{-1} and $A_L < 0.977$ g m^{-2} day^{-1}). In order to maintain high TAN removal rates in trickling filters, a highly efficient solids removal unit must be applied before the water enters the filter (Eding *et al.* 2006).

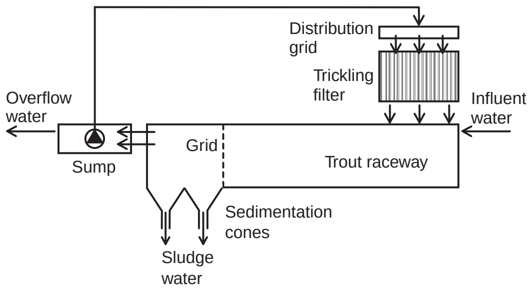

Figure 8 Possible system configuration for partial water reuse in flow-through trout production, using a trickling filter (after Andreasen 2003; Mayer 2005).

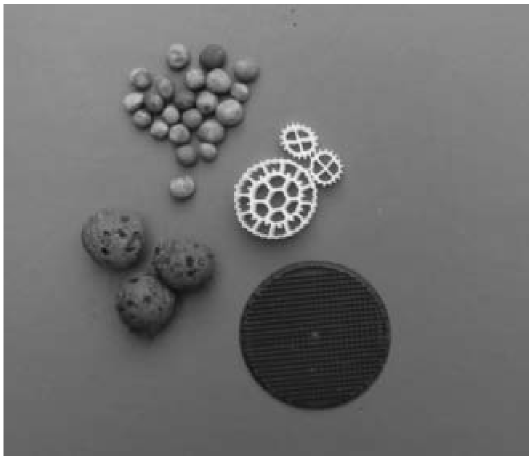

Figure 9 Possible moving filter media for moving bed reactors. On the left-hand, side clay aggregates of different size and on the right-hand side, Kaldness material and other treatment beds are shown.

Moving bed reactors

Moving bed reactors like the moving bed biofilm reactor (MBBR) described by Rusten, Eikebrokk, Ulgenes and Lygren (2006) adopt the best features of the activated sludge process as well as those of the biofilter process, without including their worst features (Rusten *et al.* 2006). The MBBR utilizes the whole tank volume for biomass growth. It has a very low head-loss. In contrarst to an activated sludge reactor, MBBR does not need any sludge recycle. This is achieved by having the biomass grow on carriers that move freely in the water volume of the reactor and are kept within the reactor volume by a sieve arrangement at the reactor outlet (Rusten *et al.* 2006). Ideally, the specific carrier density is slightly higher than water, and so they can be moved with minimal energy input. The carrier elements are mainly of polyethylene (Rusten *et al.* 2006) but clay aggregates with comparable properties can also be used. Some possible MBBR carriers are shown in Fig. 9.

In recirculation trout farms, Kaldnes-, or Kaldnes-like biofim carriers (Fig. 9 centre) made of polyethylene with a density of 0.95 g cm^{-3} a nominal diameter of 9.1 mm, a length of 7.2 mm and a specific biofilm surface area of 500 m^2 m^{-3}, as described by Rusten *et al.* (2006), were frequently used. In Denmark, the moving bed systems were combined with mechanical treatment devices for TSS and BOD reduction, mainly settlement cones and regularly backwashed submerged filters for BOD abatement (Andreasen 2003). The moving bed unit is heavily aerated for oxygen supply and to keep the beds in suspension for ideal treatment success. Water recirculation is provided mainly by huge airlift pumps, with additional water aeration (Andreasen 2003) (see Fig. 10). Unfortunately, until now, there have been no scientific reports available providing reliable data on the operation of commercial-scale partial reuse sites. However, Rusten *et al.* (2006) applied a Kaldnes moving bed reactor on the BIOFISH system described by Eikebrokk (1990). He measured TAN removal rates of 0.3 g TAN m^{-2} day^{-1} at TAN loading rates of 0.45 g m^{-2} day^{-1} at a temperature of 9 °C. The retention time in the moving bed reactor was approximately 3.5 min and the reactor was filled to 67% with Kaldnes material, providing a specific biofilm surface area of 500 m^2 m^{-3}. An automatic dosing of sodium carbonate had to be applied to maintain an average pH of 6.2 (Ulgenes & Ludin 2003; Rusten *et al.* 2006). The dosing rate unfortunately was not mentioned in

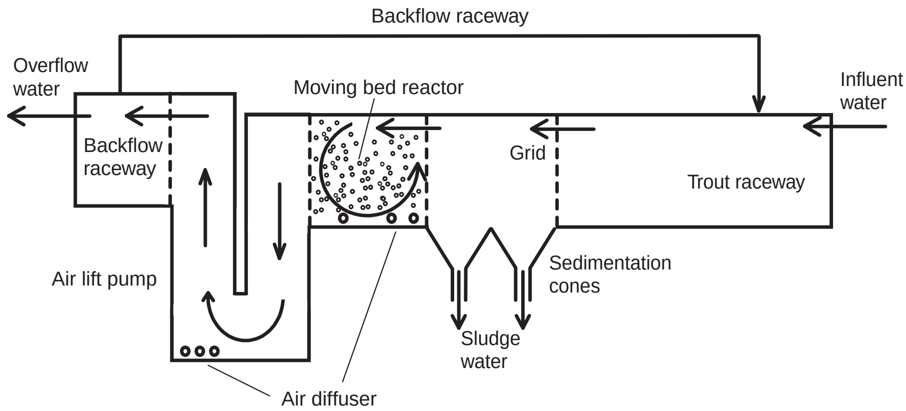

Figure 10 Possible system configuration for partial water reuse in flow-through trout production, using a moving bed filter.

both reports. Treatment efficiency of partial reuse systems in trout farms should be in the same range as the data provided by Ulgenes and Ludin (2003) and Rusten *et al.* (2006).

Strategies to reduce aquaculture effluents – synopsis

The strategies to reduce pollutions in trout farm effluents are: (1) sustainable farm management and (2) wastewater treatment. The implementation of partial water reuse in the farm represents an improvement for both strategies. The crucial decision for the trout farmer is: what kind of treatment or treatment combinations he should apply depending on his production/feeding intensity. A possible decision tree is given in Fig. 11. For the decision tree, the following assumptions were made:

- The trout feed used in Table 2 was applied ($n = 7\%$, $p = 1.1\%$, $d_N = 87\%$, $d_P = 45\%$, $d_{OC} = 81\%$), with an FW = 4%, at a FCR = 1.
- As calculation fundamentals, the formulae of Table 1 were used.
- The following limits for the concentration increases in the effluent were set: $BOD_5 = 3.0\ mg\ L^{-1}$, $TP = 0.1\ mg\ L^{-1}$ and $TAN = 1.0\ mg\ L^{-1}$, limits commonly set by local German water authorities.
- From the BOD_5 in the farm, the effluent is about 80% particle bound (Cripps & Bergheim 2000)
- For the feeding level, a constant daily feeding ratio over 365 days a year was assumed.

For the different treatment units, the following efficiencies were assumed:

- Primary sedimentation has an efficiency of about 79% of the total suspended solids (Summerfelt *et al.* 2004). As sedimentation cones were usually cleaned twice a day and the nutrient leaching from deposited sludge peaks in the first 24 h (Stewart *et al.* 2006a), a treatment efficiency of about 60% for particulate P and BOD_5 seems to be realistic.
- Micro-screening eliminates 83% of the particulate matter (Brinker *et al.* 2005b).
- With application of guar gum micro-screen efficiency increases to 88% for TSS (Brinker *et al.* 2005b).

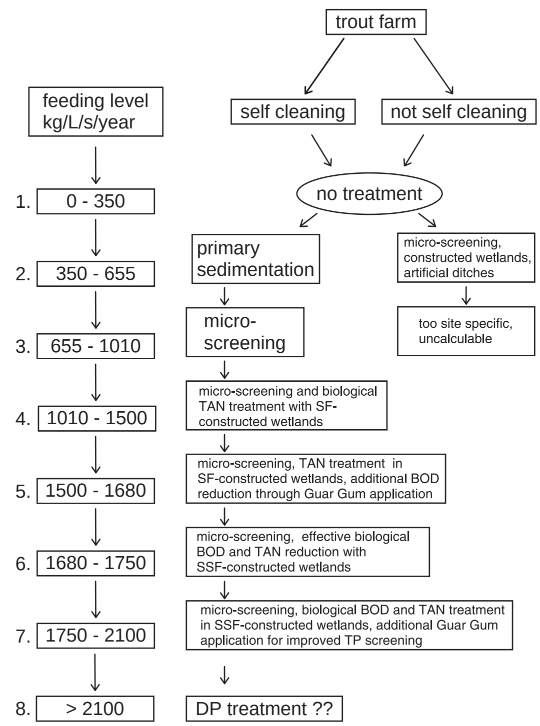

Figure 11 Possible decision tree for the implementation of effluent nutrient reduction strategies in trout farms, dependent on the feeding level.

- Submerged filters have a BOD_5 treatment efficiency of 49% with anterior micro-screening (Bergheim & Brinker 2003; Brinker *et al.* 2006).
- Constructed wetlands as SF wetland have treatment efficiencies of 41% and 31% for TAN and BOD_5 (COD), respectively, at hydraulic loading rates of 0.15 $m^3\,m^{-2}\,h^{-1}$ (Schulz *et al.* 2004). As SSF wetlands, treatment efficiencies of 87% and 86% were reached for TAN and BOD_5, respectively, at overflow rates of 0.18 $m^3\,m^{-2}\,h^{-1}$ (Sindilariu, Ettinger & Reiter 2007).

The given decision tree (Fig. 11) is just a simplistic showcase valid only for the given assumptions. The calculated feeding levels will change depending on the food quality used, the effluent nutrient increase limits and different efficiencies of the applied treatment units.

For each feeding level of Fig. 11, different nutrient reduction strategies can be recommended.

1. For a feeding level lower than 350 kg $L^{-1}\,s^{-1}$ and year no effluent treatment is needed, as the nutrient concentrations in the effluent will not increase by more than the limits set.
2. For a feeding level higher than 350 kg $L^{-1}\,s^{-1}$ and year, the BOD_5 effluent increase is more than 3.0 mg L^{-1}. Trout farms with no self-cleaning units need to install an effluent treatment like a micro-screen or constructed wetlands for particulate or dissolved nutrient treatment. As the distribution between dissolved and particulate nutrients is dependent on the leaching process within the rearing units, no further general calculation of the needed effluent treatment can be carried out. For self-cleaning rearing units with negligible nutrient leaching within the farm, the implementation of a primary sedimentation should reduce the effluent nutrients satisfactorily at this feeding level.
3. At a feeding level between 655 and 1010 kg $L^{-1}\,s^{-1}$ and year, self-cleaning farms should use micro-screens for effluent treatment as a primary sedimentation is no longer sufficient. The resulting backwash sludge, about 1% of the primary flow, has to be further treated. A chemical treatment with a flocculation aid and further sedimentation is suitable (Sindilariu & Reiter 2006).
4. At a feeding level between 1010 and 1500 kg $L^{-1}\,s^{-1}$ and year the effluent TAN concentration exceeds 1.0 mgL^{-1}. A possible solution is the application of a surface flow constructed wetland with a overflow rate of 0.15 $m^3\,m^{-2}\,h^{-1}$ (Schulz *et al.* 2004).
5. At a feeding level between 1500 and 1680 kg $L^{-1}\,s^{-1}$ and year the application of guar gum-added feed may be recommended. With normal feeding, micro-screening and SF-constructed wetland treatment, the remaining BOD_5 in the effluent exceeds 3.0 mgL^{-1}.
6. At a feeding level higher than 1680 kg $L^{-1}\,s^{-1}$ and year the effluent BOD_5 increase, exceeds 3.0 mg L^{-1} with a SF wetland applied as biological BOD_5 and TAN treatment. The additional BOD_5 treatment needed can be provided through the combination of the SF wetland with artificial ditches. Or a partial recirculation can be used for TAN and BOD_5 treatment through trickling or moving bed reactors. Alternatively the biological BOD_5 treatment is replaced through a more effective SSF wetland with overflow rates of about 0.18 $m^3\,m^{-2}\,h^{-1}$.
7. At a feeding level higher than 1750 kg $L^{-1}\,s^{-1}$ and year, the TP increase exceeds 0.1 mg L^{-1} despite the efficient micro-screening. Micro-screening can be improved through the application of guar gum-supplemented feed. Nevertheless, the biological BOD_5 and TAN treatment is further needed.
8. At a feeding level higher than 2100 kg $L^{-1}\,s^{-1}$ and year a dissolved phosphorus treatment is needed, and such a treatment method has not been commercially available until now.

To rate the costs for the effluent treatment needed, the example fish farm from Table 2 with $Q = 542\,L\,s^{-1}$ and an FCR = 1 was used. Actual prices of 3.82 US$ (3.18€) per kilogram trout (Reiter 2006), 1.27 US$ (1.06€) per kilogram trout feed (Biomar AS) and mean variable costs, excluding feed, of 0.80 US$ (0.67€) per kilogram trout produced (Engle *et al.* 2005) were used. The farm profit margins are listed in Table 4.

For the second feeding level (Table 4), unfortunately, there are no data available concerning the costs for the implementation of sedimentation cones within an existing trout farm.

For micro-screening (Table 4), a drum filter with yearly costs of 13 440 US$ (11 200€) (Engle *et al.* 2005), at a pore size 80 μm, is used. The micro-screen is well adjusted and no particles larger than 80 μm are released, as also observed by Brinker *et al.* (2005b). The remaining micro-screen backwash water has to be processed further. A flow-through of 542 L s^{-1} produces about 5 L s^{-1} backwash water. The backwash water will be settled in an idle fish pond and treated with precipitation chemicals, to avoid further release of phosphorus. The addition of precipitation chemicals may lead to a further financial setback of 2769 US$ (2307€) for the chemicals and 1077 US$ (897€) for the chemical dosage unit per year. For the sedimentation basin cleaning, 413 US$ (344€) were needed per year (Sindilariu & Reiter 2006).

The application of an SF wetland with the characteristics described by Schulz *et al*. (2004) needs a total area of 13 008 m^2 to treat the whole farm effluent. At construction costs for aquaculture ponds of 23.33 US $ m^{-2} (19.44 € m^{-2}) (Engle *et al.* 2005), also representing the costs needed for the construction of SF wetlands, this treatment method results in additional costs of 303 477 $ (252 897.50 €) at a hydraulic loading rate of 0.15 $m^3\ m^{-2}\ h^{-1}$.

For the guar gum application, additional food costs of 5% of the initial food costs were incurred.

For the biological treatment options of submerged filters, trickling filters and moving bed reactors, no information on costs for trout farm applications was available.

For the application of SSF-constructed wetlands as effluent treatment, at a hydraulic loading rate of 0.18 $m^3\ m^{-2}\ h^{-1}$, 10 840 m^2 were needed. Unfortunately, reliable data on the constructional and maintain costs for the wetlands as well as long-time data on renewal intervals are not available.

The implementation of other effluent treatment options and management actions to reduce the nutrient discharge is possible, but these options are too specific or the data are not sufficient to make general remarks on treatment efficiencies and potential costs.

Conclusions and recommendations, development trends

1. Trout aquaculture effluents contain dissolved and particulate nutrients in variable amounts, which can lead to ecological disturbances in the receiving ecosystem. There are three concepts to reduce effluent nutrient pollution: improved farm management, effluent treatment and partial water reuse. The individual combination of these concepts can effectively reduce effluent pollution.
2. Two major management strategies can improve the effluent: the provision of the best possible rearing conditions for trout to avoid stress and elevated nutrient excretion, and food improvements. Here, the application of binders, improving the treatment ability of particulate wastes is a highly promising approach.
3. Mechanical effluent treatment, especially micro-screening, is established in trout farms. Further development is needed on the processing of the micro-screen backwash sludge. Pure mechanical dewatering, the dewatering through application of coagulation/precipitation chemicals or the application of constructed wetlands are possible. Besides the mechanical methods, alternatives for the whole effluent treatment of particulate and dissolved nutrients are needed. The trend goes to cost-effective low-maintenance treatment methods like constructed wetlands.
4. For partial water reuse in trout farms, two treatment methods are established: trickling filters for low flow situations, as high pumping level are needed, and moving bed bio-filters for high flow situations.
5. For effective application of constructed wetlands further research is needed especially on:
   - the scale-up from existing experimental wetlands to the treatment of commercial farm effluents,
   - the long-time development of cleaning efficiency, maintenance work and renewal ranges, influencing the maintenance costs,
   - improvements in material and design for maximal benefit (high treatment efficiency and low total costs),
   - the calculation ranges for the construction, maintenance and operational costs for the effluent treatment in constructed wetlands and the possible and suitable combinations with other treatment methods.
6. The economical aspects of most treatment methods have not been well documented until now, an area where reliable data are imperative for the fish farmer.

Well-functioning effluent treatment in trout aquaculture is an important step towards environmentally neutral trout aquaculture as one of the four criteria for sustainable aquaculture, beneath financial self sufficiency, a stable level of returns and general social acceptability (Hishamunda & Ridler 2002).

Existing and further environmental regulations and discussion are likely to promote development and research towards achieving a sustainable freshwater flow-through trout aquaculture.

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Treatment of flow-through trout aquaculture effluents in a constructed wetland

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Abstract

A study on effluent treatment with sub-surface flow (SF) constructed wetlands was conducted in a small commercial scale Bavarian (Germany) flow-through trout farm. Under limited spatial and financial conditions a most suitable wetland was constructed. The wetland treatment efficiency at high hydraulic loading rates during raceway runoff and cleaning situation in comparison to sedimentation as initial treatment method was examined.

The constructional solution involved the alteration of six existing sedimentation basins (SB) to SF horizontal flow constructed wetlands with a pre-sedimentation area. As constructional materials only local, cheaply available materials were used in order to reduce the costs. The SF wetland had high treatment efficiencies in the two operational modes examined. During cleaning situation at a hydraulic loading rate (HLR) of 13.6 m/day treatment efficiency for total suspended solids (TSS) was highest and reached 68%. While during raceway runoff situation total ammonia nitrogen (TAN) treatment efficiency of 88% overtopped the efficiency of the other nutrients examined at a HLR of 10.6 m/day. In both treatment situations the SF wetland efficiency was significantly higher than the effect of the SB. SF constructed wetlands treating high hydraulic loading rates accompanied with short retention times were effective on dissolved nutrient treatment only for TAN and nitrite nitrogen (NO_2–N), while other dissolved nutrients like nitrate nitrogen (NO_3–N) and phosphate phosphorous (PO_4–P) showed no or even negative treatment effects through the wetland passage. To reduce these nutrients, other treatment conditions or wetland configurations are needed.

Keywords: Effluent treatment; Trout farm; Constructed wetlands

1. Introduction

Aquaculture is a worldwide constantly increasing industry (FAO, 2004). In the European Union the rainbow trout (*Oncorhynchus mykiss*) is, beneath salmon (*Salmo salar*), the most important finfish species produced (Eurostat, 2004). The European trout producing sector is mainly characterized by small, regionally rooted micro-enterprises, with an average yearly production of 100 mt or even less (MacAlister and Partners, 1999; FAO, 2003). As flow-through facilities, these enterprises operate with single- to three times water reuse. The farm effluents are characterized by low waste concentrations at high volumes. With these characteristics, they are difficult to treat (Cripps and Kelly, 1995). The environmental legislation on waste loadings and on the environmental impact from trout farms and its public discussion became stronger in recent years. Thus,

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efficient treatment methods for sustainable aquacultural practises should be developed (Naylor et al., 2000; EIFAC, 2001; Tacon and Forster, 2003; O'Bryen and Lee, 2003).

Mechanical effluent treatment on particulate wastes from trout production is actually most common (Cripps and Bergheim, 2000). The amount of particle bound nutrients in the effluent is highly variable and lies between 30 and 84% for phosphorus, 7 and 32% for nitrogen and about 80% for the organic carbon (Cripps and Bergheim, 2000). However, the soluble nutrients in the effluent remain untreated. These dissolved nutrients are ammonia/ammonium (TAN) with its degradation products nitrite nitrogen (NO_2–N) and nitrate nitrogen (NO_3–N), as well as phosphate (PO_4–P) and polyphosphates, resulting either from direct excretion of fish (Steffens, 1985; Cho and Bureau, 1997; Bureau and Cho, 1999; Green et al., 2002) or from nutrient leaching processes from the particulate fractions (Brinker et al., 2005a,b).

Although treatment efficiency could be improved by optimised diets (Brinker et al., 2005a,b), standard mechanical treatment should be combined with biological treatment methods in order to reduce both, the soluble and particulate effluent nutrient fractions. These combined treatment systems have to be cost and maintenance effective.

One possible solution of combining mechanical with biological treatment are sub-surface flow constructed (SF) wetlands. In literature it is supposed that SF wetlands might be a suitable treatment method for aquaculture effluents, but until now there exist only experimental sites (Milden and Redding, 1998). The first promising reports and data sets were reported from channel catfish (Schwartz and Boyd, 1995) and shrimp Tilley et al., 2002) effluent treatment, secondary treatment of micro-screen backwash sludge (Summerfelt et al., 1999; Comeau et al., 2001) and of settlement sludge (Michael, 2003). Furthermore, tests conducted in experimental scale constructed wetlands for the treatment of flow-through trout aquaculture (Schulz and Rennert, 2000; Schulz et al., 2003; Schulz, 2004), flow-through milkfish production (Lin et al., 2002), recirculating shrimp production (Lin et al., 2005) and experiments with artificial aquaculture effluents (Naylor et al., 2003; Lymbery et al., 2006) showed high upscaling potential. To progress on the way to commercial scale flow-through effluent treatment with constructed wetlands, a SF wetland was installed on a small commercial scale Bavarian trout farm. This study presents the treatment efficiency of the installed wetland at high hydraulic loading rates during raceway runoff and cleaning situation, in comparison to the initial treatment method (sedimentation).

2. Materials and methods

2.1. Trout farm

The aquaculture facility consisted of 6 broodstock ponds, 8 earthen raceways, 16 small concrete raceways, 12 circular tanks and an onside hatchery. In the sampling period, the farm had a standing stock of approximately 7.5 metric tons resulting in a yearly production of 10 mt of rainbow trout (*Oncorhynchus mykiss*), brown trout (*Salmo trutta*) and brook trout (*Salvelinus fontinalis*) with an average food conversion ratio of approximately 1.2. The average water supply for the whole facility was 72.3 L/s spring water, with an initial nutrient loading of 5.0 mg/L TN, 4.9 mg/L NO_3–N, 10 µg/L TP and 1.3 mg/L BOD_5 (Sindilariu and Reiter, 2005). A single water use was applied. Only the overflow of the broodstock ponds and earthen raceways was used a second time for two small raceways. The existing effluent treatment unit of the farm consisted of 8 sedimentation basins (SB). The SB were built of concrete with a pump sump and two baffles, one at the inlet, to avoid short circuit and one at the outlet to keep floating sludge within the basin (Fig. 1). The 8 SB received the overflow water from the earthen ponds and raceways circular tanks and from the hatchery. Additionally the basins received the cleaning water from all production units through a common cleaning pipe. Regular cleaning was performed once a week for all concrete raceways and circular tanks. The earthen raceways were cleaned irregularly, a situation not sampled during this study. The regular cleaning was performed within the first 6 working hours of the day, where the water level of the rearing unit was lowered until the fish remobilised the deposited sediments, then flushed to the treatment unit. A common inlet channel distributed the effluent equally to the 8 SB.

2.2. Wetland construction

For SF wetland construction the following stern premises were set:

1. The wetland cost should be as low as possible.
2. Spatial limitation of 215 m^2 representing 6 out of 8 existing SB.
3. The used SB could be drained over a common 100 mm bottom drain, resulting in a maximum head loss of 1.0 m.

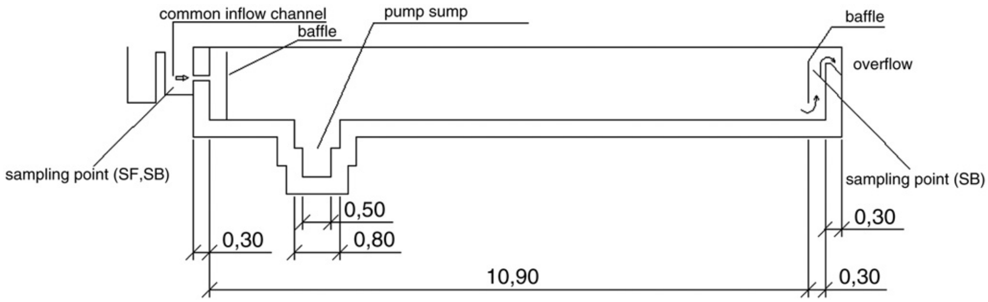

Fig. 1. Initial sedimentation basin (SB) in the examined fish farm. The sampling stations in the common inlet channel and overflow were indicated.

4. Flow rates to be treated:
   a. Constant flow of 26.3 L/s of raceway discharge.
   b. Cleaning situation with up to 72 L/s cleaning water, average flow rate of 33.9 L/s.

For the substrate size calculation the Darcy equation was used (Bahlo and Wach, 1996):

$$Q = k_f \cdot A_f \cdot I/1000 \tag{1}$$

$$k_f = 1000 \cdot Q/A_f \cdot I \tag{2}$$

where Q is the water inflow in L/s, k_f is the hydraulic conductivity of the filter material (m/s), A_f is the cross-section of the wetland root zone filter (m^2) and I the hydraulic gradient of the filter. As approximation for the hydraulic conductivity of the filter material Eq. (3) was used after Bahlo and Wach (1996).

$$k_f = (d_{10})^2/100 \tag{3}$$

$$d_{10} = (k_f \cdot 100)^{0.5} \tag{4}$$

where d_{10} is the effective grain fraction of the filter material, defined as the lowest 10% in the grading curve of the material.

Under these premises a SF wetland with a pre-sedimentation area was planed (Fig. 2). In the initial SB a wooden wall was build up, 3 m behind the inlet. For the hydraulic conditions set a conductivity (k_f) of 0.028 m/s is needed. Based on Eq. (4) an effective grain size (d_{10}) of 1.67 mm was calculated. However, a grain size of 4–8 mm was used for operational safety reasons and to extend the service life of the wetland. The area between the wooden wall and the outlet baffle was filled with local gravel of 4–8 mm grain size, with a porosity e=0.36, specific gravity of 1781 kg/m^3 and a total phosphorous (TP) content of 6 μg/g. The first 50 cm of the inlet site were packed with coarse-grained gravel of 32 mm and larger, to ensure a uniform water distribution over the front area of the root zone as recommended by Kadlec et al. (2000) and Schulz (2004). The inflow area of the wetland was cleaned with a rake, when the water started to overflow the wetland surface. The space under the outlet baffle was blocked by a layer of coarse gravel and a perforated grid, so that the outlet baffle and the perforated grid build the back wall of the SF wetland root zone. Between the back wall and the bottom drain of the basin a space of 30 cm remained (Fig. 2). SF construction finished on 01 June 2004. The total treatment area was 215 m^2 (35.8 m^2 per cell), while

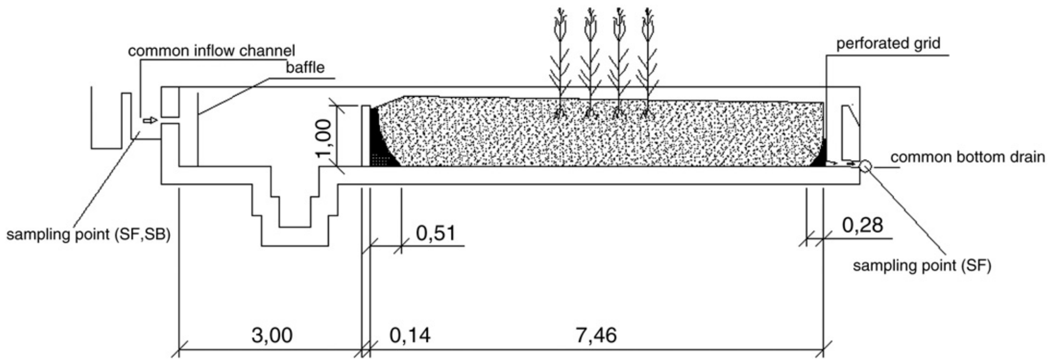

Fig. 2. Sub-surface flow (SF) wetland cell altered from the initial SB. Compared to the initial SB a wooden wall and a gravel bed was added. The sampling stations in the common inlet channel and bottom drain were indicated.

the root zone area was 143 m^2. Each cell was planted with approximately 70 root balls of reed out of a nearby fishpond. The root balls had diameters between 20 and 50 cm. After one year of effluent treatment the plant community on the constructed wetland consisted to approximately 35% of *Phragmites communis* and 35% of *Phalaris arundinacea*. The remaining 30% were inhabited by different swamp and land plants (determined after Bursche, 1963).

2.3. Sampling and analysis

After a pre-measurement phase of 7 weeks for initial establishment of reed and natural microbial consortia the collection of water samples began on 19 July 2004 and ended 06 February 2006. For sampling, portable samplers in plastic (PE) housing were used for automatic sampling according to the vacuum principle (Maxx company, Rangendingen, Germany). Water samples were taken over 24 h by an interval of 10 min for runoff situation. For cleaning situation samples were taken over 6 h by an interval of 10 min. For raceway runoff 24-hours pooled samples were analysed, while for cleaning two 3-hours pooled samples were analysed. Water samples were taken from the common inlet of the effluent treatment unit (= raceway effluent), the SB outflow and the common bottom drain of the constructed wetlands (Figs. 1 and 2). Sampling schedule was adapted to the measured water residence times, for the SB of 10 min and for the constructed wetland of approximately 20 min. In the sampling period from the common inlet and the SF wetland outflow 32 samples as 24-hours sample were taken. During cleaning also 32 samples as 3-hours sample were taken from these two sampling stations. During raceway runoff only 21 samples at SB outflow were taken simultaneously in accordance to the SF sample procedure (as 24-hours samples) and during cleaning situation 22 samples were simultaneously taken as 3-hours sample.

The samples were analysed for total nitrogen (TN mg/L), total ammonia nitrogen (TAN µg/L), nitrite nitrogen (NO_2–N µg/L), nitrate nitrogen (NO_3–N mg/L), total phosphorous (TP µg/L), orthophosphate phosphorous (PO_4–P µg/L), biological oxygen demand within five days (BOD_5 mg/L), chemical oxygen demand (COD mg/L), total organic carbon (TOC mg/L), and total suspended solids (TSS dry weight in mg/L). The physicochemical properties of the water samples were determined following German standard methods for the analyses of water, wastewater and sludge (DIN, 2006). pH and conductivity (µS/cm) were measured by electrodes (WTW Multi 197-I).

2.4. Calculations and statistics

For the statistical analyses the difference between inflow and outflow concentration was calculated with the simultaneously taken samples:

$$\Delta p = c_{\text{in}} - c_{\text{out}} \tag{5}$$

For the calculation of treatment efficiencies Eq. (6) was used:

$$\%\Delta = (\Delta p / c_{\text{in}}) \cdot 100 \tag{6}$$

where %Δ is the treatment efficiency in %; c_{in} the concentration of the water parameter in the inflow.

Hydraulic loading rates (HLR) in m/day, area loading rate (AL) and area removal (AR) in g/m^2 day^1 were calculated following Eqs. (7), (8) and (9), respectively.

$$\text{HLR} = 86.4 \cdot Q/A \quad 86.4 = (60 \cdot 60 \cdot 24)/1000 \tag{7}$$

Q is the inflow rate in L/s; A is the wetland surface area in m^2.

$$\text{AL} = c_{\text{in}} \cdot \text{HLR} \tag{8}$$

$$\text{AR} = \Delta p \cdot \text{HLR} \tag{9}$$

Statistical calculations were performed with the SAS 8e software package, according to the statements of Hatcher (2003). For the statistical efficiency analyses of the effluent treatment methods the difference, Δp (Eq. (5)) was built for each pair wise collected sample and each parameter. For each parameter a Shapiro–Wilk test for normality was performed, with a significance level of $\alpha < 0.05$. When the Δp data were normally distributed, then the one sample Student's *t*-test was performed, in order to evaluate whether Δp is significantly different from 0. When normality for the Δp data was rejected, then the Wilcoxon test (signed rank test) was used to test whether Δp is significantly different from 0 or not.

3. Results

3.1. Wetland treatment efficiency

The SF constructed wetland was operated under two different treatment situations: the treatment of raceway runoff and the treatment of cleaning water. The SF wetland treatment efficiency was compared to the efficiency of the SB as control.

3.1.1. Treatment of raceway runoff

The mean inflow and outflow concentrations of the analysed parameters are listed in Table 1. Treatment efficiency is calculated following Eq. (6). TP was the only analysed parameter that showed a non-significant concentration decrease of 38.1% in the SF wetland outflow, due to high variances in the inflow TP concentration. The other parameters had a significant concentration decrease, except TN, NO_3–N and PO_4–P, which showed non-significant increase in concentration. During runoff situation the SF wetland significant treatment efficiency ranged between 9.2% and 86.9%. The SF wetland had for most parameters a significant higher treatment efficiency than the SB, except NO_3–N, where the SB had a higher efficiency. For TN, TP and PO_4–P no significant difference between the treatment methods during raceway runoff situation could be detected (Fig. 3).

3.1.2. Treatment of cleaning water

The nutrient concentrations from the two sampling points were listed in Table 2. During cleaning situation the SF wetland had a significant effect on all analysed parameters, except pH and NO_3–N. PO_4–P is the only one showing a significant concentration increase in the SF wetland outlet. Significant treatment efficiency during cleaning situation ranged between 4.9% and 67.9% for TN and TSS respectively. Compared to the treatment efficiency of the initial SB, the SF wetland showed for nearly all analysed nutrient fractions a significant higher treatment efficiency, except TN and NO_3 (Fig. 4).

3.2. Hydraulic and nutrient loading rates, area removal rates

The two different flow situations resulted in different hydraulic and nutrient loading as well as removal rates in the wetland. In Fig. 5 the treatment efficiencies of the wetland at the two operational modes were compared. In Table 3, Δp as well as the nutrient loading and removal rates is listed. Nutrient loading rate (AL) is for all parameters, except TAN, during cleaning much higher than during raceway runoff situation. The area removal (AR) for most nutrient parameters is significantly higher during cleaning situation. Exceptions are the AR of NO_3–N and TOC, where no significant differences could be found between the two operation stages. PO_4–P showed a significant higher area release during cleaning and TAN showed a higher AR during raceway runoff situation.

4. Discussion

4.1. Nutrient loads

The nutrient concentrations measured at the outflow of the rearing units were at the lower end of the ranges reported for flow-through trout production (Butz, 1990; Cripps, 1994; Boaventura et al., 1997; True et al., 2004; Maillard et al., 2005; Viadero et al., 2005). Due to the extremely high nitrate load from the inflowing spring water NO_3–N exceeded the literature data. The extensive fish production with low phosphorous

Table 1
Inflow and outflow concentrations and difference between inflow–outflow concentrations in the SF wetland at raceway runoff situation in the sampling period (n=32), including the achieved treatment efficiency %Δ

Parameter	Inflow		Outflow		Δp	$\Delta p \neq 0$	SF
	$\bar{x}$	SE	$\bar{x}$	SE		α	%Δ
pH	7.73	0.18	7.70	0.18	0.03	0.0109	
Cond. (µs/cm)	723	2.87	724	1.52	−1	0.1108	
TN (mg/L)	5.12	0.12	5.22	0.10	−0.10	0.6852	−2.0
TAN (µg/L)	140.63	10.31	18.48	5.05	122.15	0.0001	86.9
NO_2–N (µg/L)	16.60	0.96	10.70	1.49	5.90	0.0003	35.5
NO_3–N (mg/L)	4.95	0.143	5.04	0.151	−0.08	0.3003	−1.6
TP (µg/L)	58.35	17.32	36.10	3.34	22.25	0.9156	38.1
PO_4–P (µg/L)	31.31	4.08	31.62	3.10	−0.31	0.5700	−1.0
BOD_5 (mg/L)	2.41	0.19	1.52	0.12	0.89	0.0001	36.9
COD (mg/L)	7.00	0.59	5.30	0.49	1.70	0.0001	24.3
TOC (mg/L)	2.71	0.23	2.45	0.18	0.25	0.0140	9.2
TSS (mg/L)	2.70	0.56	1.76	0.29	0.93	0.0154	34.4

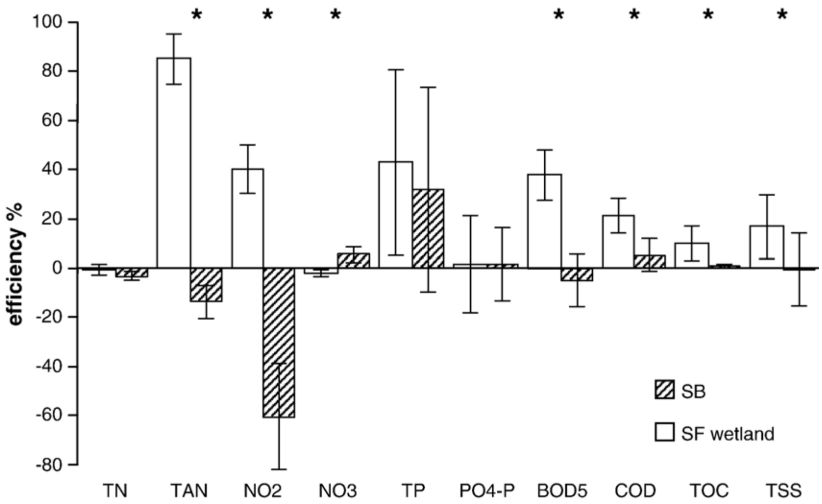

Fig. 3. Comparative treatment efficiency of the SF wetland and the initial SB basin during pond runoff situation ($n=21$). With the indication of standard error and significance, indicated with *.

containing feed leads to lower PO_4–P concentrations than reported. The farm had a yearly application of 166 kg feed per L/s inflowing water. At a yearly production intensity higher than 500 kg applied feed per L/s the installation of effluent treatment devices is recommended in Bavaria (Schobert et al., 2001).

4.2. SB effluent treatment

The treatment efficiency of the SB is very low due to low inflow TSS concentrations. The investigated SB showed only during cleaning situation a significant treatment effect of 8.1% for TN and 51.0% for TSS (Fig. 4). During raceway runoff the SB had no treatment effect (Fig. 3). In contrary the SB had a significant soluble nutrient leaching, leading to significantly increased NO_2–N and TAN concentration in the outflow during raceway runoff and cleaning situation, respectively. These findings corroborate the statements of Henderson and Bromage (1988), Cripps and Bergheim (2000), and Piedrahita (2003) that sedimentation is not a viable treatment method for the whole flow-through farm effluent due to very low TSS concentrations. Henderson and Bromage (1988) predicted a treatment

Table 2
Inflow and outflow concentrations and difference between inflow–outflow concentrations in the SF wetland at cleaning situation in the sampling period ($n=32$), including the achieved treatment efficiency %D

Parameter	Inflow		Outflow		Δp	$\Delta p \neq 0$	SF
	$\bar{x}$	SE	$\bar{x}$	SE		α	%Δ
pH	7.67	0.18	7.68	0.18	−0.01	0.8932	
Cond. (µs/cm)	732	7.05	718	1.52	14	0.0001	
TN (mg/L)	5.93	0.161	5.63	0.165	0.29	0.0147	4.9
TAN (µg/L)	107.50	8.10	54.76	9.97	52.74	0.0001	49.1
NO_2–N (µg/L)	21.58	2.12	12.61	1.24	8.98	0.0001	41.6
NO_3–N (mg/L)	5.18	0.128	4.95	0.183	0.23	0.5142	4.4
TP (µg/L)	124.32	22.62	56.35	6.09	67.97	0.0015	54.7
PO_4–P (µg/L)	45.67	9.13	56.96	5.10	−11.29	0.0021	−24.7
BOD_5 (mg/L)	3.01	0.36	1.55	0.12	1.47	0.0001	48.8
COD (mg/L)	11.95	1.14	5.71	0.448	6.24	0.0001	52.2
TOC (mg/L)	2.87	0.22	2.45	0.17	0.42	0.0039	14.6
TSS (mg/L)	7.55	1.48	2.08	0.27	5.13	0.0001	67.9

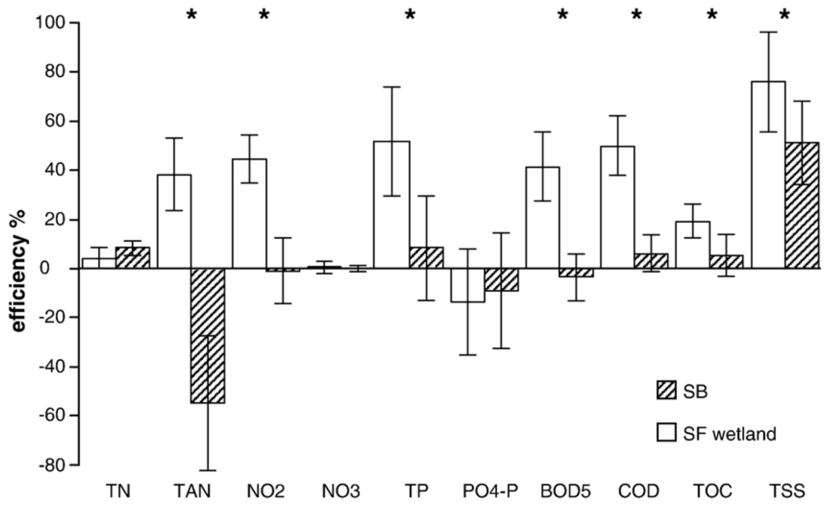

Fig. 4. Comparative treatment efficiency of the SF wetland and the initial SB basin during pond cleaning situation (n=22). With the indication of standard error and significance, indicated with *.

effect of sedimentation ponds starting at TSS concentrations >6.64 mg/L. In the present study this TSS concentration was exceeded only in 10% and 33% of the measured raceway runoff and cleaning samples, respectively.

4.3. SF effluent treatment

In both treatment situations the SF had an effect on the effluent composition. During raceway runoff especially TAN was reduced (87%), while the other nutrients were reduced by −2% to 38% (Table 1).

During cleaning situation the nutrient loading (AL) was 1.3 to 3.6 times the loading rates during raceway runoff, except for TAN, where AL was constant in both situations. With the higher nutrient loads also the treatment efficiency especially for the particulate nutrients increases (Schulz et al., 2003). Efficiencies of 49% to 68% for BOD_5, COD, TP and TSS were measured (Table 2). Compared to wetlands from other aquaculture effluent treatment studies this SF wetland had the highest HLR with 10.6–28.9 m/day (Table 4). In consequence the wetland had the lowest reported retention time of only 0.014 days and a relative coarse filter material of 4–8 mm had to be used to prevent clogging. The installed wetland had a clear treatment improvement compared to the initial SB. The AR for TSS, COD, TAN and TP were within the range reported

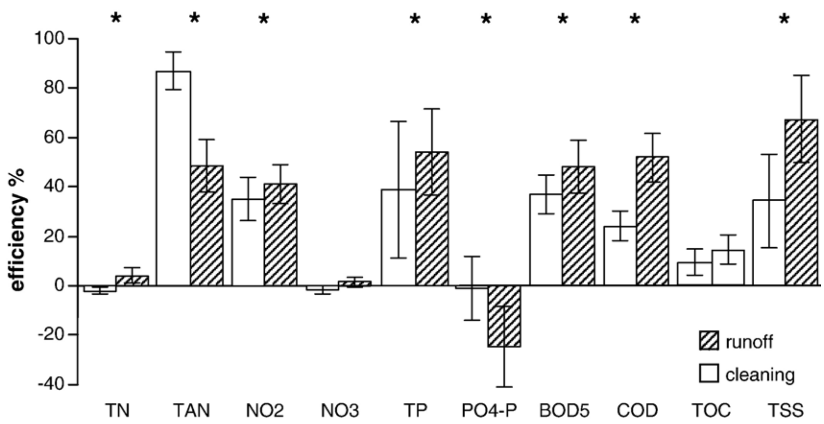

Fig. 5. Comparative treatment efficiency of the SF wetland at raceway runoff and cleaning situation. With the indication of standard error and significance, indicated with *.

Table 3
Influence of treatment situation on the absolute nutrient difference (Δp) between inflow and outflow, the area loading rate (AL, Eq. (8)) and the area removal rate (AR, Eq. (9)) of the SF wetland (n=32 for both, raceway runoff and cleaning situations)

Parameter	Δp		AL (g/m^2 day)		AR (g/m^2 day)	
	Raceway runoff	Cleaning	Raceway runoff	Cleaning	Raceway runoff	Cleaning
TN (mg/L)	-0.10^{a}	0.29^{b}	54.2	80.5	-1.02^{a}	3.49^{b}
TAN (μg/L)	122.15^{a}	52.74^{b}	1.49	1.46	1.30^{a}	0.716^{b}
NO_2–N (μg/L)	5.90^{a}	8.98^{a}	0.18	0.29	0.06^{a}	0.12^{b}
NO_3–N (mg/L)	-0.08^{a}	0.23^{a}	52.4	70.4	-0.88^{a}	3.12^{a}
TP (μg/L)	22.25^{a}	67.97^{b}	0.62	1.69	0.24^{a}	0.92^{b}
PO_4–P (μg/L)	-0.31^{a}	-11.29^{b}	0.32	0.62	-3^{a}	-153^{b}
BOD_5 (mg/L)	0.89^{a}	1.47^{b}	25.5	40.9	9.5^{a}	19.9^{b}
COD (mg/L)	1.70^{a}	6.24^{b}	74	162	18.0^{a}	84.7^{b}
TOC (mg/L)	0.25^{a}	0.42^{a}	28.6	39.0	2.68^{a}	5.67^{a}
TSS (mg/L)	0.93^{a}	5.13^{b}	29	103	9.9^{a}	69.7^{b}

[a]Raceway runoff: hydraulic loading rate (HRT) = 10.6 m/day.
[b]Raceway cleaning: hydraulic loading rate (HRT) = 13.6 m/day.
Values with same superscript letter do not differ significantly.

by Schulz et al. (2003) and Lin et al. (2005) and much higher than the AR for wetlands treating municipal wastewater (Bahlo and Wach, 1996).

4.3.1. SF particulate nutrient treatment

The wetland provides a constant mechanical screening of the suspended solids in dependence of the granular material used (Oehler, 1982). This effect is also corroborated by the higher nutrient removal rates, especially of particulate nutrients during cleaning actions (Table 3). Due to the low inflow concentrations, the average treatment efficiencies for particulate nutrients were lower than the literature reported values. Lin et al. (2005) calculated a rate constant (k) for TSS

Table 4
Characteristics of sub-surface flow constructed wetlands for aquaculture effluent treatment

Aquaculture category	Treated effluent	Used material	Production intensity (feed kg/day)	Area of application (m^2)	Total wetland volume (m^3)	Daily hydraulic load (HLR) (m/day)	Retention time (days)	Source
Channel catfish	Pond effluent	Soil	n.s.	2×1176	1435	0.077–0.0 91	1–4	Schwartz and Boyd (1995)
Trout recirculation	Clarifier backwash sludge	Soil, sand gravel combination	n.s.	6×4.44	21.3	0.304	n.s.	Summerfelt et al. (1999)
Trout production	Micro-screen backwash	Crushed lime stone 2.5–5.0 mm	n.s.	170	100	0.212	1.3	Comeau et al. (2001)
Milkfish production	Bypass	Gravel 10–20 mm	n.s.	5	3	0.270	0.6	Lin et al. (2002)
Salmon hatchery	Settling pond effluent	Soil	82	270	125	0.079	2,7	Michael (2003)
Artificial	Collected and diluted sludge	Steel slag, limestone, gravel, peat 5–10 mm	n.s.	20×1	5	0.030	4	Naylor et al. (2003)
Trout experimental	Complete effluent	Sand 1–2 mm	0.3	3×1.4	2.9	1.029–5.143	0.31–0.063	Schulz et al. (2003)
Shrimp production	Effluent recirculation	Gravel 10–20 mm	1.8	30.40	36.5	1.54–1.95	0.096–0.067	Lin et al. (2005)
Trout production	Effluent and cleaning	Gravel 4–8 mm	33	6×35.8	215	10.6–28.9	0.014	This investigation

n.s.: not specified.

treatment in aquacultural used constructed wetlands (Eq. (10)).

$$k = 6.50 \cdot \mathrm{HLR}^{1.045}(\text{for TSS removal}) \quad (10)$$

The needed wetland area for the observed treatment efficiency can be calculated (Eq. (11)).

$$A_\mathrm{w} = Q \cdot (\ln C_\mathrm{in} - \ln C_\mathrm{out}) \cdot (k \cdot \varepsilon \cdot h_\mathrm{w})^{-1} \quad (11)$$

A_w is the wetland area, ε the filter porosity (0.36) and h_w the filter height (1 m).

For the reached TSS treatment efficiency during raceway runoff situation a theoretical wetland area of 35.3 m^2 is needed. During cleaning situation the needed area would be 102.7 m^2. The used wetland (143 m^2) might be smaller for the achieved TSS removal. This is most probably due to the low inflow TSS concentration especially during runoff situation, leading to low treatment efficiencies.

The TN in the examined wetland was heavily affected by the NO_3 concentration, which did not show a significant change by the SF wetland treatment. Therefore, analysis of Kjeldal N probably would have been more appropriate to examine the nitrogen degradation processes in the SF wetlands in detail.

4.3.2. SF dissolved nutrient treatment

Dissolved nutrient treatment is one of the core advantages of constructed wetlands compared to standard mechanical effluent treatment (Schulz, 2004). The wetland treatment efficiency is strongly correlated with the nutrient dependent rate constant and the wetland retention time (Kadlec et al., 2000). For high hydraulic loading it is arguable if the wetland retention time is long enough for reduction of dissolved nutrients. In contrast, particulate nutrients are effectively treated through the mechanical filtration of the root zone.

4.3.2.1. TAN. TAN removal in biological filters used for aquaculture is a function of the TAN loading rate and filter retention time (Liao and Mayo, 1972, 1974). Using the rate constant (k) and wetland area calculation from Lin et al. (2005) (Eqs. (11) and (12)) the TAN treatment efficiency is much better during raceway runoff situation than calculated, with a theoretical needed wetland area of 393.8 m^2 for the reached treatment efficiency (Eq. (12))

$$k = 5.40 \cdot \mathrm{HLR}^{0.761}(\text{for TAN removal}) \quad (12)$$

During cleaning situation the rate constant and area calculation fit the measured situation quite well with 136 m^2 calculated. It can be assumed that in the TAN rate constant under the measured high volume, low nutrient charge situation is higher than calculated by Lin et al. (2005). During cleaning situation, the elevated BOD loads cause the lower treatment efficiencies and not a lower TAN rate constant. Changing and especially increasing BOD lead to reduced nitrification and TAN removal rates in biological filters (Eding et al., 2006). This could explain the TAN removal during cleaning situation, where the treatment efficiency dropped to only 48% and the removal rate was just half the removal rate during runoff situation. However, the calculation of Lin et al. (2005) represents much better the characteristics of this kind of wetland treatment, than the calculations used for domestic wastewater treatment in Kadlec et al. (2000).

4.3.2.2. NO_2–N. In the SF wetland NO_2–N treatment efficiency was stable for runoff and cleaning situation (35.5% to 41.6%). The treatment efficiency was lower than the values reported from literature (Table 5). However the other studies had an at least two fold higher NO_2–N inflow concentration. The wetland showed an exponential area removal rate with increased NO_2–N inflow loads. The loading rate increased 1.4 times, while the removal rate increased 2.0 times. It can be assumed, that with elevated NO_2–N loads the treatment efficiency could achieve the reported values.

4.3.2.3. NO_3–N. Nitrate was the only analysed parameter showing no significant change during wetland treatment in none of the treatment situations. This could lead to the conclusion that no dentrification occurred due to high oxygen concentrations in the rearing unit outflow. This is in line with other studies on wetlands, treating high hydraulic loads of nearly oxygen saturated aquaculture effluents, where an increase in NO_3–N concentrations in the wetland outflow was observed too (Summerfelt et al., 1999; Schulz et al., 2003; Lin et al., 2005).

4.3.2.4. PO_4–P. In this investigation the PO_4–P contents after wetland passage showed two different effects: during runoff situation the PO_4–P concentration showed no change, while during cleaning situation PO_4–P significantly increased by 25%. The PO_4–P treatment in wetlands has to be seen in the context of total phosphorous (TP) treatment. TP removal in wetlands is a manifold process. There are physical, chemical and biological forces influencing the different

Table 5
Inflow concentration and treatment efficiency of sub-surface flow wetlands from the literature in comparison to this investigation

Applied system	Analysed parameters	Treated inflow concentration (mg/L)	Treatment efficiency in %		Source
			up	to	
Two consecutive horizontal SF wetlands	TAN	0.337	71.2		Schwartz and Boyd (1995)
	NO_2–N	0.041	43.9		
	NO_3–N	0.543	52.7		
	Kjeldal N	1.61	45.3		
	TP	0.162	68.5		
	BOD_5	5.61	36.9		
	TSS	34.5	75.3		
2×3 horizontal and vertical SF wetlands	NO_3–N	0.057	−570	−80,000	Summerfelt et al. (1999)
	Kjeldal N	234	86	89	
	TP	238	82	90	
	PO_4–P	106	92	93	
	COD	6855	71.9	91.3	
	TSS	7860	95.8	97.2	
Horizontal SF wetland	TP	0.03–0.61	87		Comeau et al. (2001)
	TSS	7.8–65.5	94		
Combination of consecutive surface and SF wetland	TAN	1.406	44.6		Lin et al. (2002)
	NO_2–N	0.4	96.1		
	NO_3–N	1.372	81		
	PO_4–P	6.808	51.3		
Horizontal SF wetland divided in three cells	TAN	0.43	75.1		Michael (2003)
	TP	0.76	82.3		
	BOD_5	18.93	81.6		
	TSS	58.01	91.3		
6 SF wetlands with different filter materials	TAN	1.39	−287	81.3	Naylor et al. (2003)
	NO_3–N	0.99	44.1	69.7	
	Kjeldal N	12.41	40.1	89.6	
	TP	2.69	−14.1	91.1	
	PO_4–P	1.78	5.1	95.5	
	COD	373	52.8	91.1	
	BOD_5	104	68.3	99	
	TSS	187	96.8	100	
Three horizontal SF wetlands treating different HLR	TN	2.4	26.2		Schulz et al. (2003)
	TAN	0.61	72.5		
	NO_3–N	0.70	−46.3		
	TP	0.347	67.5		
	PO_4–P	0.124	13.4		
	COD	41.01	73.8		
	TSS	14.15	97.3		
Combination of consecutive surface and SF wetland	TAN	0.18–0.25	64	66	Lin et al. (2005)
	NO_2–N	0.13–0.35	94	83	
	NO_3–N	5.60–39.6	−5.4	−2.4	
	PO_4–P	1.05–3.59	−7.6	−4	
	BOD_5	3.0–6.2	37	54	
	TSS	11.6–20.6	66	55	
6 horizontal SF wetlands	TN	5.1–5.9	−1.8	4.3	This study
	TAN	0.14–0.11	87.8	49.1	
	NO_2–N	0.017–0.022	35.3	41.6	
	NO_3–N	5.0–5.2	−1.7	1.7	
	TP	0.058–0.12	39.6	54.6	
	PO_4–P	0.031–0.046	−1.0	−24.7	
	COD	7.0–12	24.3	52.2	
	BOD_5	2.4–3.0	37.1	48.6	
	TSS	2.7–7.6	34.6	68.0	

TP components (Kadlec and Knight, 1996). The mechanical phosphorous retention in this wetland was very effective as the outflowing TP consisted to 84% and 95% of PO_4–P, during cleaning and runoff situation, respectively. However the PO_4–P increase in the wetland effluent by 11 µg/L during cleaning situation was unexpected. Constructed wetlands are supposed to have a positive treatment effect on outflow PO_4–P concentrations (Bahlo and Wach, 1996; Kadlec and Knight, 1996, Table 5). Nevertheless in wetlands treating high hydraulic loads PO_4–P increases were observed too (Schulz et al., 2003; Lin et al., 2005). A possible explanation is the leaching of PO_4–P from the trapped particulate phosphorous. As no P removal occurred, the trapped particulate P is enclosed and accumulated in the wetland root zone. As nutrient leaching from faecal trout matter is high (Stewart et al., (2006), and during cleaning situation areas with nearly stagnant water within the wetland were streamed, dissolved P is washed out leading to the increased PO_4–P concentrations. During runoff situation the washout effect is too low to be detected.

Constructed wetlands have the potential to treat NO_3–N and PO_4–P. For this challenge other conditions than the ones described are needed. Higher retention times in combination with higher nutrient loads or organic sediments may lead to denitrification process in the wetland (Van Rijn et al., 2006). For effective denitrification, a combination of vertical and horizontal flow SF wetlands can be used (Platzer, 1998). The preliminary vertical flow wetland removes TAN, while the subsequent horizontal flow wetland provides the denitrification. In both wetlands long retention times were realized (5 and 21 days), so oxygen free areas can easily establish (Platzer, 1998). This was examined on municipal wastewater, but should also be valid for fish-farm effluents.

To treat PO_4–P effectively other substrates than gravel might be used. In test shale and bauxite or steel slag and limestone showed the highest PO_4–P retention with artificial municipal sewage and fish-farm effluents, respectively. The effect was achieved mainly through precipitation (Drizo et al., 1999; Naylor et al., 2003). However, the high pH removing of limestone inhibited plant grow on the substrate, so that two subsequent treatment basins were needed, one for effluent treatment and one for additional P binding (Naylor et al., 2003).

A potential plant harvesting will not improve the wetland treatment efficiency. After Rhodewald-Rudescu (1974) a maximum standing crop of 214.5 kg dry plant matter could be expected on the 143 m^2 wetland root zone, containing maximum 5 kg TN and 0.23 kg TP. Compared to the yearly loading of 4245 kg TN and 48.4 kg TP only during raceway runoff situation, the expected nutrient reduction through plant harvest is <0.1% and 0.5% for TN and TP, respectively.

5. Conclusions and recommendations

Commercial scale flow-through trout culture effluent treatment with constructed wetlands is possible and effective when compared to SB treatment. Particulate and dissolved nutrients in farm effluents are reduced during normal and cleaning operations, despite high hydraulic loads and low influent nutrient concentrations. Compared to standard mechanical effluent treatment the efficiency of the SF wetland for TSS polishing was in the range of micro-screening and significantly higher than the initially used treatment (sedimentation). If SB were already in use on farm level they can easily and cheaply be altered to SF wetlands resulting in higher effluent treatment efficiencies. For dissolved nutrients the examined wetland showed high treatment efficiencies only for TAN and NO_2–N. Other dissolved nutrients as NO_3–N and PO_4–P showed no or negative treatment effects.

Starting from this feasibility and preliminary study more detailed studies should be conducted, examining the treatment efficiency of the wetland at different nutrient and hydraulic loading rates. Addressing each wetland cell as independent wetland, three treatments in duplicate can be tested. Such an approach would give the possibility for extrapolation of the treatment area needed for individual, distinct farm effluents. The other important topic is the economic impact of wetland effluent treatment. Here the renewal rate of the filter is one of the main aspects. Until now no information for such a SF wetland application is available.

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Constructed wetlands as a treatment method for effluents from intensive trout farms

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ABSTRACT

This study examined the effects of different hydraulic loading rates on the treatment efficiency of subsurface flow (SSF) constructed wetlands treating effluents from trout farming over a period of 6 months. Six identical wetland cells with a pre-sedimentation zone of 9.6 m^2 and a root zone of 23.6 m^2 were loaded with effluents from intensive trout farming (>2.1 kg feeding stuff per L/s and day). The total runoff of 13.2 L/s was treated in the wetland cells, where two duplicate cells received equal hydraulic loads of 3.9, 1.8 and 0.9 L/s. All examined wetland cells had significant treatment effects on the nutrient fractions containing particulate matter [total nitrogen (TN), total phosphorous (TP), biological oxygen demand in 5 days (BOD_5), chemical oxygen demand (COD), and total suspended solids (TSS)].
Efficiency was between 5.5% for TN and 90.1% for TSS. The SSF wetland also had a high treatment effect on total ammonia nitrogen (TAN), with efficiencies of 61.2 to 87.8%. Nitrate nitrogen (NO_3–N) and phosphate phosphorous (PO_4–P) showed a significant increase in the wetland effluent by 8.4 to 209%. Nitrite nitrogen (NO_2–N), had no significant, or significant effluent increase depending on the inflow rate. Treatment efficiency for particulate nutrients and TAN increased with decreasing hydraulic load, while the differences between 1.8 and 0.9 L/s were not significant. The treatment efficiency for TP was constant for all cells, at around 40%. The wetland receiving 3.9 L/s was over-flooded after 10 to 12 weeks due to colmatation. Nevertheless, the wetland still showed high treatment efficiencies. For commercial trout farms, SSF wetlands are a highly effective method of effluent treatment. A hydraulic load of 1 L/s on 13.3 m^2 wetland area (1.8 L/s on the examined wetland) seems most suitable. Higher loads lead to accelerated wetland colmatation, while lower loads waste space.

1. Introduction

The worldwide demand for aquaculture products is constantly increasing (FAO, 2007). In the European Union, the rainbow trout (*Oncorhynchus mykiss*) is the most important cultured finfish species, with a total production of 215,207 t in 2003 followed by the Atlantic salmon with 162,585 t (European Commission, 2006). More than 90% of the European aquaculture farms are small and geographically dispersed (Varadi et al., 2001). The trout producing sector is characterized by regionally rooted enterprises with an average annual per farm production of 100 t or less (MacAlister Elliott, 1999).

Successful trout production requires substantial quantities of high quality inflow water, and thus, uses mainly brook or spring water (Schäperclaus and Lukowicz, 1998). Water quality requirements limit both the number of suitable locations and the production capacity per site (Lukowicz, 1994). Further increase of trout production can be achieved only by intensifying the production at existing sites. The water volume required for the production of one ton of trout decreased to 86,000 m^3 with the use of energy-rich extruded feeds (Schäperclaus and Lukowicz, 1998) and dropped to 26,000 m^3 with the additional use of oxygenation and aeration technologies (Brinker et al., 2006). Although waste minimising strategies have been implemented, which improved the quality of the effluent despite the increased production (Milden and Redding, 1998, Bergheim and Brinker, 2003; McMillan et al., 2003; Summerfelt et al., 2004), more stringent environmental legislation and increased public awareness still ask for more efficient cleaning technologies (Naylor et al., 2000; EIFAC, 2001; Tacon and Forster, 2003; O'Bryen and Lee, 2003; Viadero et al., 2005).

Commonly used mechanical effluent treatments (Cripps and Bergheim, 2000) often fail the stringent effluent thresholds of the water authorities' environmental constraints under intensive production conditions (Sindilariu, 2007). The dissolved total ammonia nitrogen (TAN) and the dissolved fraction of the biological oxygen demand (BOD_5) can exceed the limits at high production intensities, when only mechanical treatments like micro screening or sedimentation are used (Sindilariu, 2007). A cost-saving alternative treatment method might be provided by the use of constructed wetlands, which combine mechanical and biological effluent treatment, as described by Schulz et al. (2003, 2004), Lin et al. (2005), and Sindilariu et al. (2007). Especially the application of subsurface flow (SSF) wetlands, where the water flow is directed through a planted filter matrix and ideally no surface water flow occurred, showed promising results

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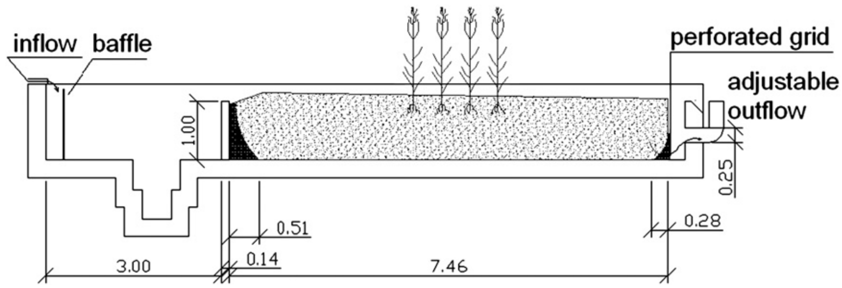

Fig. 1. Subsurface flow (SSF) wetland cell, receiving the raceways outflow from the distribution box ("inflow"). A wooden wall and a gravel bed were installed after the pre-sedimentation area. Wetland outflow sampling occurred in the adjustable outflow bend. (Measures indicated in meter).

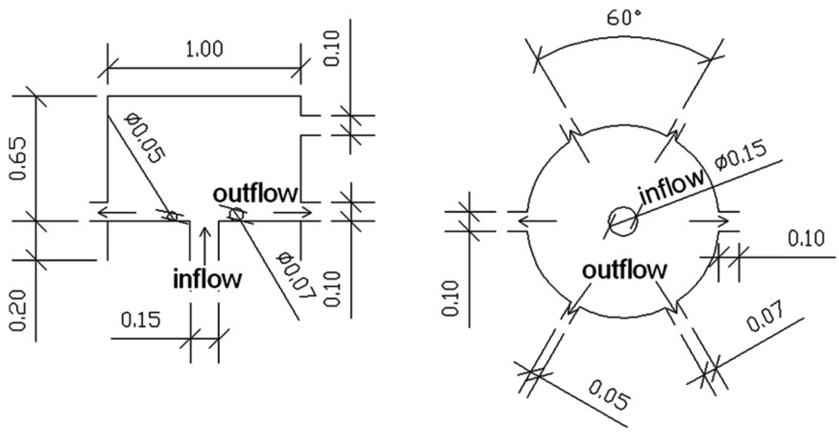

Fig. 2. Central distribution box and wetland inflow sampling station, receiving the common effluent from both raceways ("inflow") and dispensing the outflow of 3.9, 1.8 and 0.9 L/s to the six wetland cells over the six pipe fittings ("outflow"). (Measures indicated in meter, left side cross section, right side top view).

on an experimental scale. However, the efficiency, as well as the space requirements and potential hydraulic loads of constructed wetlands under intensive production, are widely unknown. Typically, constructed wetlands are considered as too space-consuming and inefficient.

This study aims to evaluate the application of constructed wetlands in intensively producing trout farms, especially the impact of hydraulic loading on the treatment efficiency. Further, it was tested if the first order TAN and TSS removal rates given by Lin et al. (2005) are also valid for wetlands treating high volume effluents from intensive trout farming. This study presents the results of a six-month effluent treatment from intensive trout production by an experimental subsurface flow (SSF) wetland.

2. Materials and methods

2.1. Experimental setup

The SSF constructed wetland used for this study consisted of six identical wetland cells. Each cell had an area of 23.9 m^2 for the rood zone and 9.6 m^2 as pre sedimentation area (Fig. 1, further details in Sindilariu et al., 2007). The wetland has been in use since June 2004. The wetland received the effluent from two raceways at the experimental trout farm of the Institute for Fishery, representing the trout production unit. Both raceways flushed into a common pipe leading to a central distribution box (Fig. 2). Valves situated on the six outflow pipes of the box enabled a controlled distribution of the total runoff to the six wetland cells. Flow distribution was controlled regularly (15 times during the sampling period) by flow rate measurement. Two cells each received the same hydraulic load of on average 3.9, 1.8 and 0.9 L/s, respectively. The total raceway runoff was 13.2 L/s from August 2006 until March 2007. Each production raceway had a water volume of 32.7 m^3. Their inflow had average nutrient concentrations of 5.12 mg/L TN, 0.14 mg/L TAN, 0.017 mg/L NO_2–N, 4.95 mg/L NO_3–N, 0.058 mg/L TP, 0.031 mg/L PO_4–P, 2.41 mg/L BOD_5, 7.00 mg/L COD, and 2.70 mg/L TSS, at an average pH of 7.73 and a conductivity of 723 µS/cm (Sindilariu et al., 2007).

Both raceways were initially stocked on August 08, 2006 with rainbow trout (*Oncorhynchus mykiss*) for a four-week pre-experimental phase. On September 07, 2006 the desired stock density was reached. During the experimental phase (07.09.2006–

Table 1
Physical water parameters in the SSF wetland cells inflow and outflow by different hydraulic loads

Parameter	Inflow		Hydraulic load					
			3.9 L/s		1.8 L/s		0.9 L/s	
	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Temp °C	9.14^a	0.04	9.04^b	0.05	9.03^b	0.04	8.89^c	0.05
Cond. µS/cm	771.4^a	0.9	773.3^b	0.2	777.0^c	0.2	777.5^c	0.2
O2 mg/L	11.04^a	0.15	2.65^b	0.08	1.19^c	0.02	0.88^d	0.01
pH	7.869^a	0.002	7.765^b	0.002	7.624^c	0.007	7.559^d	0.008

Different letters indicate significant differences ($p<0.05$) in the physical water parameters between hydraulic loads ($N=1911$).

Table 2
Mean inflow and outflow concentrations and treatment efficiencies (%Δ) of the SSF wetland cells by hydraulic load

Parameter		Hydraulic load					
		3.9 L/s		1.8 L/s		0.9 L/s	
mg/L	C_{in}	C_{out}	%Δ	C_{out}	%Δ	C_{out}	%Δ
TN	6.14^a	5.79^b	−5.5	5.72$^{b\ c}$	−6.8	5.53^c	−10.0
TAN	0.75^a	0.29^b	−61.2	0.09^c	−87.8	0.13^c	−82.9
NO_2–N	0.011^a	0.021^b	100	0.015$^{a\ b}$	43.9	0.019$^{a\ b}$	75.5
NO_3–N	4.84^a	5.24^b	8.4	5.47^b	13.0	5.31^b	9.7
TP	0.25^a	0.14^b	−43.1	0.14^b	−43.4	0.15^b	−39.9
PO_4–P	0.041^a	0.106^b	158	0.115$^{b\ c}$	180	0.127^c	209
BOD_5	6.90^a	1.96^b	−71.5	0.95^c	−86.2	0.78^c	−88.7
COD	14.20^a	6.42^b	−54.6	5.34$^{b\ c}$	−62.4	4.66^c	−67.2
TSS	7.18^a	1.10^b	−84.6	0.82^c	−88.6	0.71^c	−90.1

Different letters indicate significant differences ($p<0.05$) in the nutrient concentration between the groups ($N=26$).

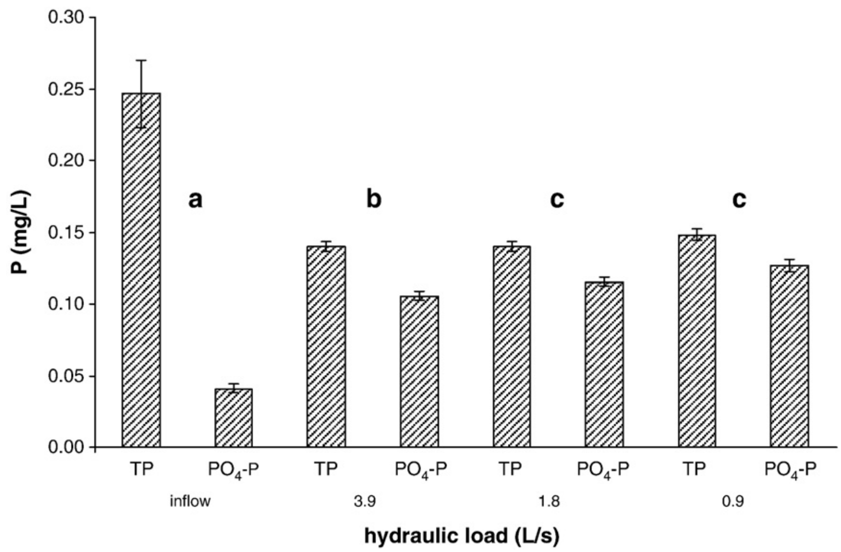

Fig. 3. Comparison between TP and PO_4–P concentration in the wetland inflow and outflow in relation to the hydraulic load. Different letters indicate significant differences in particulate P calculated as TP−PO_4–P.

08.03.2007), the stock density in the raceways was between 1200 and 2300 kg (36.7–70.3 kg/m^3) of 200 to 700 g trout. The fish increment in the raceways was harvested every fourth week in turn. The initial feeding ratio of 1% fresh weight during the first 12 weeks was reduced to 0.8% afterwards. To maintain a mean feeding rate of about 30 kg feed per day, the fish density was then increased. A total of 5372 kg of a commercial trout grower feed containing 42.0% protein, 22.0% crude fat, 3.3% fibre, 8.0% ash, and 1.1% TP, (Biomar AS) was applied during the experimental phase at a feed conversion ratio of 1.15.

2.2. Sampling and analysis

In the sampling period water samples were taken from the common outflow of the raceways as well as from each wetland cell once a week between September 07, 2006 and March 07, 2007. Sampling was performed by flexible tube pumps placed into the distribution box (Fig. 2) and into the adjustable wetland outflow bend (Fig. 1). The pumps ran for 24 h. Every 9.5 min, a magnetic valve opened a connection to the sampling canister for 30 s. From the canisters, 24-h pooled samples were analysed.

A constant sampling schedule was followed for the whole sampling period. First, the sampler for the raceway outflow was started at time $T=T_0$. Then, at time $T=T_0+$ 50 min, at time $T=T_0+70$ min, and at time $T=T_0+90$ min, the outflow samplers of the wetland cells receiving 3.9 L/s, 1.8 L/s, and 0.9 L/s respectively, were started. Sampling ended in the same order as it started. The sampling interval corresponded to the measured residence time of the water in the different wetland cells.

For six times during the sampling period, four YSI 600 XLM data loggers (Yellow Spring Instruments inc.) were used to measure inflow and outflow temperature, conductivity, dissolved oxygen, and pH of three wetland cells, receiving 3.9, 1.8, and 0.9 L/s, respectively. The physical water parameters were measured every 10 min during the 24-h sampling period.

The water samples were analysed for TN (mg/L), TAN (mg/L), NO_2–N (mg/L), NO_3–N (mg/L), TP (mg/L), PO_4–P (mg/L), BOD_5 (mg/L), COD (mg/L), and TSS dry weight in mg/L. The physicochemical properties of the water samples were determined following German standard methods for the analyses of water, wastewater and sludge (DIN, 2006).

2.3. Calculations and statistics

Differences (Δp) between outflow and inflow concentrations were calculated for each parameter as well as for each pair of simultaneously taken samples. The relative treatment efficiency (%Δ) was calculated for each parameter as $\%\Delta=(\Delta p/c_{in})\bullet 100\%$, with Δp=outflow−inflow difference in mg/L and c_{in}=inflow concentration in mg/L. Negative relative treatment efficiency indicates a reduction of the outflow nutrient concentration compared to the inflow.

Hydraulic loading rates (HLR in m^3/m^2/day=m/day) were calculated as:

$$\mathrm{HLR} = 86.4 \bullet Q/A$$

Q is inflow rate in L/s and A is the wetland surface area in m^2.

The related area loading rates and area removals (AL, AR in g/m^2 and day) were calculated as $\mathrm{AL}=c_{in}\bullet\mathrm{HLR}$ and $\mathrm{AR}=\Delta p\bullet\mathrm{HLR}$, respectively.

To test whether the removal rates constant given by Lin et al. (2005) for wetlands treating aquaculture effluents are valid also for the examined situation, the measured wetland effluent nutrient concentration were compared with calculated data. The calculation is limited to TAN and TSS, as only for these two parameters formulae regarding aquaculture effluent treatment in SSF wetlands are available. To calculate the theoretical nutrient concentration c_e, the removal rate constant formula given in Kadlec and Knight (1996) and Kadlec et al. (2000) was transformed.

$$c_e = c_{in}/\exp(k\bullet\varepsilon\bullet h_w/\mathrm{HLR})e,$$

where k=first-order removal rate constant (day^{-1}), ε=SSF wetland porosity (0.36) and h_w=wetland water depth (1 m). For the calculation of wetland effluent concentration of TAN and TSS, the k estimations of Lin et al. (2005) are used:

$$k_{TAN} = 5.40\bullet\mathrm{HLR}^{0.761} \text{ and } K_{TSS} = 6.50\bullet\mathrm{HLR}^{1.045}.$$

Each parameter was tested for normality using the Shapiro–Wilk test. In case of non-normality, the Wilcoxon-test (signed rank test) was used to test whether or not Δp is significantly different from zero. In case of normality, the one sample Student's t-test was used. Hierarchical clustering (Ward method, complete linking) of all measured effluent nutrient concentrations was used to analyse the differences between the three hydraulic load groups of SSF wetland cells (Backhaus et al., 2000). SSF wetland cell group means were compared by one-way ANOVA and post hoc Tukey tested. In case of heteroscedasticity, data were log transformed prior to variance analysis. The area retention rates in relation to the inflow concentration and the calculated and measured effluent nutrient concentrations were compared by a Welch ANOVA with a post hoc Wilcoxon-test and a final Bonferoni adjustment. Statistical calculations were performed with the SAS 8e software package, according to the statements of Hatcher (2003).

Table 3
SSF wetland nutrient area loading (AL) and area retention (AR) in relation to the hydraulic load received

	Hydraulic loading rate (HLR m^3/m^2/day=m/day)					
	14.1 m/day		6.4 m/day		3.3 m/day	
	(3.9 L/s)		(1.8 L/s)		(0.9 L/s)	
Parameter	AL g/m^2/d	AR g/m^2/d	AL g/m^2/d	AR g/m^2/d	AL g/m^2/d	AR g/m^2/d
TN	61.7	3.42^a	28.5	1.93^b	14.3	1.42^c
TAN	7.52	4.61^a	3.47	3.05^b	1.74	1.44^c
NO_2–N	0.35	−0.35^a	0.16	−0.07^b	0.08	−0.06^b
NO_3–N	48.7	−4.09^a	22.5	−2.92^b	11.2	−1.09^c
TP	2.48	1.07^a	1.15	0.50^b	0.57	0.23^c
PO_4–P	0.41	−0.65^a	0.19	−0.34^b	0.10	−0.20^c
BOD_5	69.4	49.6^a	32.0	27.6^b	16.0	14.2^c
COD	143	77.9^a	65.9	41.1^b	33.0	22.1^c
TSS	72.2	61.1^a	33.3	29.5^b	16.7	15.0^c

Different letters indicate significant differences ($p<0.05$) in the area retention amounts between loading rates (N=26).

3. Results

3.1. Wetland effect

The SSF wetland passage had a significant effect on the physical water parameters (Table 1). Through the wetland passage, water temperature, oxygen content, and pH significantly decreased, while conductivity increased. Temperature, oxygen, and pH were directly related to the hydraulic load and decreased with lower inflow rates. Dissolved oxygen (DO) decreased extremely during the wetland passage to 8–24% of the inflow value.

All six SSF wetland cells had a significant effect on the nutrient concentration. The effluent concentrations of TN, TAN, TP, BOD_5, COD and TSS were significantly decreased by 5.5 to 90.1%, during the wetland passage, compared to the wetland inflow (Table 2). The concentrations of NO_2–N, NO_3–N, and PO_4–P increased during the wetland passage. While for NO_3–N and PO_4–P, the increase of 8.4 to 209%, respectively, was significant for all cells, the NO_2–N effluent concentrations did not significantly increase in the cells receiving the 1.8 and 0.9 L/s load.

The Ward method revealed three distinct groups and always clustered cells together receiving similar inflow amounts, which indicates that the effect of the specific SSF wetland cell is of minor importance compared to the effect of the hydraulic load.

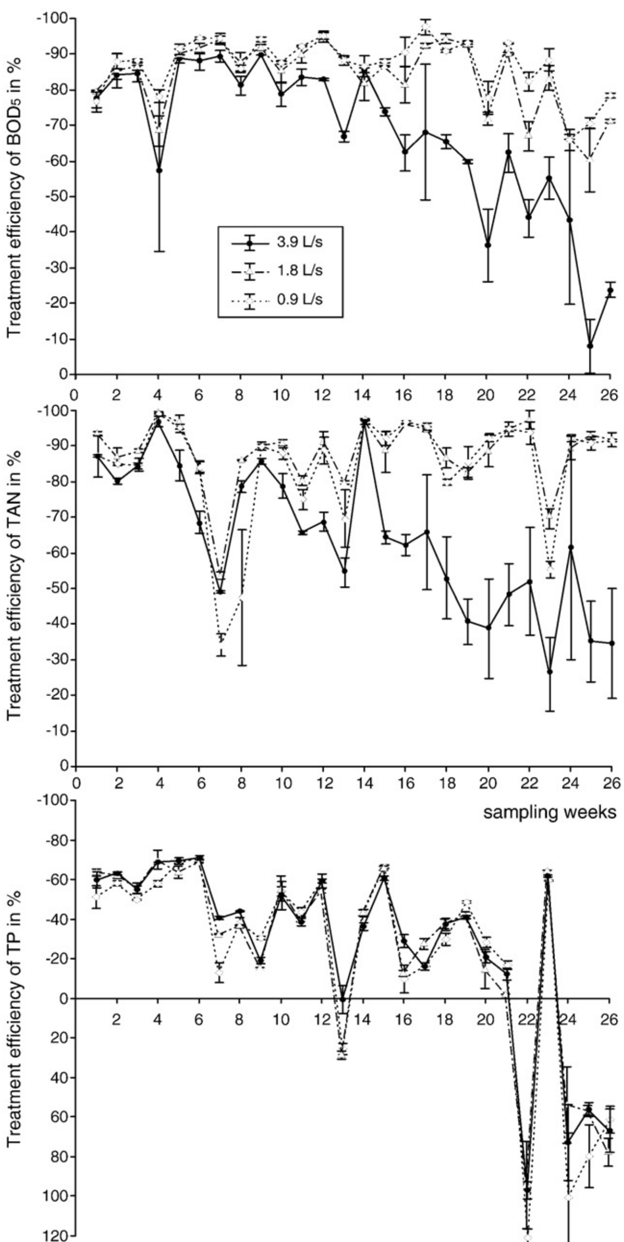

Fig. 4. Temporal development of the treatment efficiencies of BOD_5, TAN, and TP in relation to the hydraulic load.

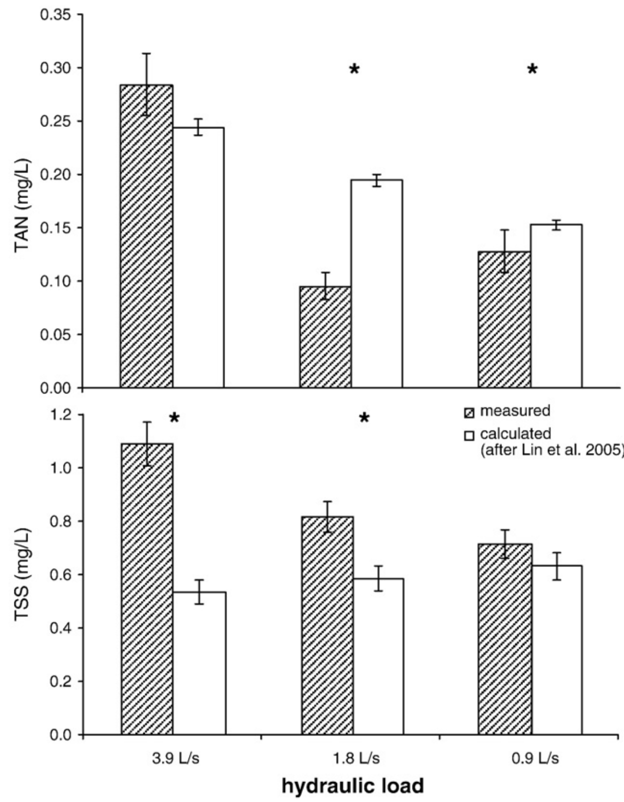

Fig. 5. Comparison of measured and calculated (after Lin et al., 2005) effluent concentrations of TAN and TSS in relation to the hydraulic load. (* indicates significant differences between measured and calculated values).

The difference between the 0.9 L/s and 1.8 L/s clusters was slightly lower than the difference between these cluster and the 3.9 L/s cluster.

At the highest hydraulic load, the treatment efficiency for TAN (61.2%), BOD_5 (71.5%), and TSS (84.6%) was lowest compared to the other two hydraulic loads (Table 2). For TN (5.5%) and COD (54.6%) a significant difference was found only in comparison to the lowest load of 0.9 L/s. The dissolved nutrient treatment efficiencies for the 3.9 L/s load did not differ significantly from the other two hydraulic load conditions, except for PO_4–P for which the effluent concentration was significantly lower compared to the 0.9 L/s load (Table 2).

The treatment efficiency at the intermediate hydraulic load did not significantly differ from the lowest load. The treatment efficiency for TAN, BOD_5 and TSS was significantly higher than those of the 3.9 L/s (Table 2).

The highest treatment efficiencies were detected at the lowest hydraulic load for TN (10.0%), BOD_5 (88.7%), COD (67.2%), and TSS (90.1%) (Table 2). However, these values did not differ significantly from those at 1.8 L/s. The treatment efficiencies for the dissolved nutrients (NO_2–N, NO_3–N, PO_4–P) showed no differences among the hydraulic loads, except for PO_4–P where the concentration increase of 209% at 0.9 L/s compared to the inflow concentration was significantly higher than the increase at 3.9 L/s. For TP, all hydraulic loads showed the same treatment efficiency of about 40% (Table 2).

With decreasing hydraulic load, the amount of PO_4–P as fraction of TP increased from initially 16.6% in the inflow to 75.5% at 3.9 L/s and 85.6% at 0.9 L/s (Fig. 3). The difference in the PO_4–P fraction between 1.8 and 0.9 L/s was not significant.

The area nutrient loading and the area retention for TN, TAN, TP, BOD_5, COD, and TSS were highest at 3.9 L/s (Table 3). Maximum retention rates of nearly 80 g COD per m^2 of wetland per day were reached. However, the dissolved nutrient release per SSF wetland area of 0.35, 4.09, and 0.65 $g/m^2/day$ for NO_2–N, NO_3–N, and PO_4–P, respectively, were also highest. The area retention for 0.9 L/s was lowest for TN, TAN, TP, BOD_5, COD, and TSS. For the dissolved nutrients (NO_3–N, and PO_4–P), the release per m^2 per day was lower for the 0.9 L/s load, compared to the other hydraulic loads. NO_2–N release at 0.9 L/s was similar to the 1.8 L/s load, but significantly lower than at the 3.9 L/s load.

3.2. Temporal development of treatment efficiency

Ten to 12 weeks after the start of intensive trout farming (08.08.2007) (6 to 8 weeks after the start of the sampling period), the water started to flow over the soil filter in the cells receiving 3.9 L/s due to colmatation. The amount of water flowing over the wetland surface increased successively during the sampling period. Consequently, the treatment efficiencies of the flooded cells decreased for most nutrient parameters from the 14th sampling week on, as shown exemplarily for BOD_5 and TAN in Fig. 4. However the treatment efficiency for TP showed no differences between the hydraulic loads in time (Fig. 4).

3.3. Theoretical and measured effluent nutrient concentration

The theoretical wetland effluent nutrient concentration was calculated in relation to the inflow concentration c_{in} and HLR. This was compared to the measured concentration (Fig. 5). For TAN, the measured concentrations at hydraulic loads of 1.8 and 0.9 L/s were significantly lower than calculated, while for 3.9 L/s, the measured effluent concentration was higher, but not significantly different from the calculated one. For TSS, the calculated SSF wetland effluent concentrations were 49 to 89% of the measured values, for 3.9 and 1.8 L/s, respectively. For TSS no significant difference between the measured and calculated values was found for the 0.9 L/s load.

4. Discussion

4.1. Effluent treatment

In contrast to other studies, using newly constructed wetlands as effluent treatment system for aquaculture (e.g., Summerfelt et al., 1999; Schulz et al., 2003; Lin et al., 2005), this study was conducted on a wetland in use for more than 2 years. Additionally, the wetland treated the highest hydraulic loading rates ever reported (Sindilariu et al., 2007). Consequently, the wetland cells, receiving 3.9 L/s, first clogged as a result of high TSS loads, 10 weeks after the installation of the new treatment regime. The other wetland cells showed no symptoms of clogging during the sampling period. As a result of wetland colmatation and increasing overflow, the treatment efficiencies for some nutrient fractions declined continuously (Fig. 4). The applied hydraulic load of 14.1 m/day, with a mean TSS area loading of 72.2 g/m^2/day is higher than any reported long-time charge of a SSF wetland. Only Schulz et al. (2003) treated comparable TSS area loadings via a wetland but at a lower HLR of 5.1 m/day. The wetland still provided acceptable treatment efficiencies, but they were much lower than the treatment efficiencies of the other wetland cells especially for the most relevant nutrients, TAN, BOD_5, and TSS. Consequently the hydraulic rate has to either be reduced or a more efficient pre-treatment removal of particles than the used pre-sedimentation area, e.g. micro-screening, has to be applied.

Given that the sampling was conducted during the winter, as indicated by the decreasing water temperatures during the wetland passage in dependence on the hydraulic load (Table 1), no nutrient extraction through plant growth was expected. The measured treatment efficiencies are only due to mechanical blockage of particles in the SSF wetland soil and biological nutrient degradation.

With increased hydraulic load, the treatment efficiency decreased for TN, BOD_5, COD, and TSS (Table 2). Compared to the previous examination of the same wetland (Sindilariu et al., 2007) the treatment efficiency increased with higher nutrient concentration in the inflow from 0 to 10% and from 17 to 90% for TN and TSS, respectively. Schulz et al. (2003) reached slightly higher treatment efficiencies at similar hydraulic loading rates. He could not find significant treatment differences between HLR of 3.1 and 5.1 m/day, as in this study between HLR of 3.3 and 6.4 m/day.

4.2. Dissolved nutrient fraction

For the dissolved nutrients, only TAN was highly effective reduced by 61 to 88% in the wetland. At a closer look at the area retention rates (Table 3), 80–96% of the inflowing TAN was transformed to NO_2–N and NO_3–N. The high TAN treatment rate, in combination with the nearly complete transformation of TAN into nitrite and nitrate, indicates a high nitrification rate in the wetland. This assumption is supported by the significant pH decrease (Table 1), typical for nitrification (Hagopian and Riley, 1998; Tchobanoglous et al., 2003; Eding et al., 2006), through the wetland passage. This pH decrease was also detected in other wetlands with high TAN reduction rates (Schulz et al., 2003; Lin et al., 2005; Sindilariu et al., 2007). The NO_3–N emission from the wetland corresponds well with the TAN treatment efficiency (Table 2), while the NO_2–N emission seems to be influenced by the short retention time which did not allow for complete nitrification (high hydraulic loading rate) in the cells receiving 3.9 L/s. TAN removal in biological filters used for aquaculture is a function of the TAN loading rate and filter retention time (Liao and Mayo 1972, 1974). Nitrification is inhibited at low DO concentrations (<0.5 mg/L) leading to incomplete nitrification with increased NO_2–N concentrations in the effluent (Tchobanoglous et al., 2003). The lowest DO concentrations occurred at 0.9 L/s, explaining the slightly elevated NO_2–N concentrations in the effluent.

4.3. Phosphorous treatment

Phosphorous treatment, in SSF wetlands under high hydraulic loading rates, has never been as effective as in wetlands treating low hydraulic loading rates or for domestic wastewater with high TP concentrations (reviewed by Sindilariu et al., 2007). TP removal in wetlands is a manifold process. There are physical, chemical, and biological forces influencing the different TP components (Kadlec and Knight, 1996). In the case of the examined wetland, the particulate phosphorous was mechanically sieved from the effluent. Seventy-eight to 84% of the wetland effluent TP consisted of PO_4–P (Fig. 3). The mechanical treatment leads to constant TP treatment efficiencies around 40%. However, dissolved PO_4–P is significantly increased by 158 to 209%. Typically, constructed wetlands have a positive treatment effect on outflow PO_4–P concentrations (Bahlo and Wach, 1996; Kadlec and Knight, 1996; Summerfelt et al., 1999; Lin et al., 2002; Naylor et al., 2003). Nevertheless, in wetlands treating high hydraulic loads, PO_4–P increases have been observed (Schulz et al., 2003; Lin et al., 2005). The only logical explanation is that PO_4–P is leaching from the trapped particulate phosphorous in the pre-sedimentation area and the wetland itself. As no P-removal from the wetland soil matrix occurred (no plant growth), the trapped particulate P is enclosed and accumulated in the wetland root zone. Nutrient leaching from faecal trout matter is high (Steward et al., 2006). Additionally, PO_4–P is solubilized under anoxic conditions (Berkheiser et al., 1980) and wetlands showing high phosphorous retentions in the start-up phase will be followed by large phosphorous exports within a few years (Richardson, 1985). The PO_4–P retention in wetlands can be improved through the addition of specialized substrates as steel slag, limestone or clay aggregates (Drizo et al., 1999; Naylor et al., 2003). However, even the PO_4–P binding capacity of these specialized substrates is limited.

The examined wetland still had a positive treatment effect on TP, however, when inflow TP concentrations was below 0.1 mg/L (sampling weeks 21, 24, 25, and 26) effluent TP exceeded inflow TP concentration.

4.4. Removal rate constant

To calculate the needed SSF constructed wetland dimensions at known effluent nutrient concentrations and set nutrient targets, the formulae of Kadlec and Knight (1996) and Kadlec et al. (2000) on the relationship between inflow and outflow nutrient concentration provide excellent tools. However, for this calculation removal rate constant estimations are needed. Lin et al. (2005) provide such an estimation for the treatment of high volumes of aquacultural effluents.

The wetland effluent nutrient concentrations calculated after Lin et al. (2005) are not in line with the measured values (Fig. 5). A new calculation formula for the removal rate constant is needed that describes the situation of highly hydraulic and nutrient-loaded wetlands under trout production conditions. The most important factor in the removal rate calculation is the probably age of the wetland. The highly loaded wetland cells examined in this study showed a clear temporal decrease in treatment efficiency mainly due to the occurrence of water flowing over the wetland. While the other two less loaded wetland cells showed nearly no differences in treatment efficiency, independent of a nearly doubled HLR. For a detailed formulation of the relationship between removal rate, inflow nutrient concentration, hydraulic load, time effect, and potentially seasonal effects, the sampling period of the wetland should be extended over a whole year.

5. Conclusions and recommendations

The SSF wetland cells in this study were highly effective for the treatment of intensive trout farm effluents. However, the hydraulic and TSS load for the wetland cells receiving 3.9 L/s was too high; in consequence, water flowed over the wetland after a few weeks of intensive effluent treatment and the wetland showed decreasing treatment efficiencies.

The most suitable area need for commercial application after this preliminary study seems to be 13.3 m^2 per 1 L/s of farm effluent. Higher loads have the risk of early colmatation, while lower loads are a waste of space and budget.

At a high production intensity of 1000 kg/year per L/s inflow amount (Brinker et al., 2006; Sindilariu, 2007), a small trout farm with, a yearly production of 100 t/year, would need a wetland of about 1330 m^2 for successful effluent treatment.

For preliminary area requirement calculations, in dependence on the effluent nutrient concentration and set nutrient targets, reliable formulae for the removal rate constant are needed. The formulae, provided by the literature, are not suitable for intensive trout farm effluent treatment. An expended sampling period would be beneficial to the estimation of removal rate constant under intensive conditions, in dependence on inflowing nutrient concentration, hydraulic load, and service time.

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Factors influencing the efficiency of constructed wetlands used for the treatment of intensive trout farm effluent

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ABSTRACT

In this paper the factors influencing treatment performance of subsurface flow constructed wetlands (SSF wetlands) treating aquaculture effluents were identified and quantified. The financial impact of advanced aquaculture effluent treatment with SSF wetlands was calculated.

It is the first long-term, commercial-scale trial of SSF wetland treatment for effluents from intensive trout farming, a highly diluted effluent at very high flow rates (mean total phosphorous concentration 0.34 mg L^{-1} at 14.3 L s^{-1}). The 12-month survey provided the opportunity to generate calculation fundamentals for the commercial application of SSF wetlands for aquaculture. Treatment efficiencies of up to 75–86% for total ammonia nitrogen (TAN), biological oxygen demand (BOD_5) and total suspended solids (TSS) were achieved. The daily area retention rate per square meter wetland area was between 2.1 and 4.5 g for TAN and between 30 and 98 g for TSS.

The performance of the six wetland cells comprising three replicated hydraulic loading groups (14.5, 6.9, 3.3 m^3 m^{-2} day^{-1}) was monitored, offering the possibility to identify factors influencing treatment efficiency through multifactor analysis. These factors turned out to be nutrient inflow concentration, hydraulic loading rate and accumulation of TSS within the wetland bed, the only time-dependent factor. Factors such as vegetation period and fish harvesting were shown to be of significant but negligible importance.

Inflow nutrient concentration is determined by production intensity, husbandry conditions, feed quality and any pre-treatment of effluent. Hydraulic load is determined by the space and budget available for SSF construction. TSS accumulation in the wetland is influenced by pre-treatment of the solid fraction prior to the wetland and determines the wetland service lifetime.

From these factors the expenses of commercial wetland application can be estimated, leading to a cost increase around €0.20 kg^{-1} fish produced (less than 10% of production costs) and therefore confirm the commercial feasibility of SSF wetland treatment.

1. Introduction

Worldwide demand for aquaculture products is increasing constantly, prompting an average increase in production of 8.8% per year since 1970, making it the fastest growing animal production sector (FAO, 2006). In the European Union, rainbow trout (*Oncorhynchus mykiss*) is the most important finfish species reared under farm conditions, with a total annual

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production of 215,207 metric tons (European Commission, 2006).

As with all other forms of livestock farming and husbandry, aquaculture produces waste, in the form of solids (uneaten feed, faeces, etc.) and dissolved material, which is transported out of the rearing system with the husbandry water. In the context of the European Water Framework Directive, aquaculture effluents are considered one point pollution sources. However, each European country is entitled to set its own rules to prevent harmful environmental effects from aquaculture effluents (Bergheim and Brinker, 2003). In Germany for instance, no common rule is applied. Here, each local authority can set own effluent standards in depending on the local needs. General guidelines recommend an increase between farm in and outflow of 3 mg L^{-1} BOD_5 and 15 mg L^{-1} total suspended solids (TSS) (Schobert et al., 2001). Also, limits for total phosphorous (TP) (0.1 mg L^{-1}) and total ammonia nitrogen (TAN) (1.0 mg L^{-1}) have in certain cases applied (Sindilariu, 2007).

One of the most promising methods for dealing with these effluents is the use of constructed wetlands, which provides biological treatment of waste, including removal of particulate nutrient material (Milden and Redding, 1998). They are used for a wide range of different effluents and are rated as low-cost systems by municipal wastewater management services (Sundaravadivel and Vingesvaran, 2001). Surface flow (SF) wetlands with open water and emergent macrophytes (Redding et al., 1997; Tilley et al., 2002; Michael, 2003; Schulz et al., 2004; Lin et al., 2005) and sub-surface flow (SSF) wetlands, where the water flows through a planted mineral filter usually consisting of a gravel matrix (Schwartz and Boyd, 1995; Summerfelt et al., 1999; Comeau et al., 2001; Naylor et al., 2003; Schulz et al., 2003; Lin et al., 2005; Sindilariu et al., 2007; Maltais-Landry et al., 2007) have usually been applied.

In aquaculture applications SSF constructed wetlands treat either the primary farm outflow, namely the entire effluent from flow-through farms (Schulz et al., 2003; Sindilariu et al., 2007), and recirculating facilities (Lin et al., 2003, 2005) or secondary effluents, as highly concentrated wastes such as micro-screen backwash sludge (Summerfelt et al., 1999; Naylor et al., 2003; Maltais-Landry et al., 2007). In both cases, treatment efficiency is high, with up to 90% removal of particulate and dissolved nutrients (reviewed in Sindilariu et al., 2007).

Inflow through salmonid production, mainly mechanical treatment methods such as micro-screening and sedimentation (Cripps and Bergheim, 2000; Brinker et al., 2006) are actually used. Both methods are limited to particle-bound nutrients comprising between 7% and 32% of total nitrogen (TN), 30–84% of TP and about 80% of organic carbon (BOD_5) (Cripps and Bergheim, 2000). Additionally only a part of the particles is retained in the effluent treatment device. For micro-screening the particle size distribution and the applied mesh size define the treatment efficiency (Brinker and Rösch, 2005), while for sedimentation the sinking speed and the basin retention time are the decisive factors (Tchobanoglous et al., 2003). This leads to maximum TSS treatment efficiencies for flow through trout farm effluents of 50–87% for micro-screening (Brinker and Rösch, 2005) and maximum 60% for sedimentation (Sindilariu, 2007). Additionally, sedimentation does not immediately remove the settled particles from the primary outflow, resulting in high dilution rates from the already settled, anaerobic material (Stewart et al., 2006). Farms with mechanical treatment only, reach the effluent limits for TAN and BOD_5 at a production intensity of about 1200–1500 kg food per year calculated per L s^{-1} inflow amount (Sindilariu, 2007).

Constructed wetlands used to treat the primary farm outflow combine mechanical treatment through fixed bed filtration with additional biological treatment of TAN and BOD_5. They have been used successfully in experimental trout rearing facilities, but are considered to be not economically feasible at a commercial scale (Schulz et al., 2003; Sindilariu et al., 2007).

The various kinds of wetland construction and the factors influencing treatment efficiency for municipal and conventional agricultural effluents have been well covered in the scientific and engineering literature (e.g. Kadlec and Knight, 1996; Brix, 1997; Tanner et al., 1998, 1999; Kadlec and Reddy, 2001; Sundaravadivel and Vingesvaran, 2001; Kadlec et al., 2005; Garcia et al., 2005; Akratos and Tsihrintzis, 2007). However, there has been relatively little published research dealing specifically with the use of SSF wetlands to treat primary aquacultural wastes (Schulz et al., 2003; Lin et al., 2005; Sindilariu et al., 2007, 2008). Such systems are required to operate under extremely high hydraulic loads, dealing with large volumes in which nutrient levels fluctuate greatly (Cripps and Bergheim, 2000). As such they are not comparable to wetland systems employed to treat municipal and agricultural wastewater.

This study provides the first long-term study (1 year) of treatment efficiency in a wetland polishing effluents from an intensive flow-through fish farm. As a mature wetland was used (it has been in use since 2004, sampling period from 2006 to 2007), the factors influencing treatment efficiency were analysed and the applicability of SSF wetlands for commercial scale aquaculture effluent treatment is examined.

2. Materials and methods

2.1. Experimental setup

Experiments were conducted in the experimental trout farm of the Bavarian State Research Center for Agriculture, Institute for Fishery. A contemporary description of the technical specifications for the rearing and treatment facilities is given also in Sindilariu et al. (2007, 2008).

2.1.1. Trout farming

The two identical production raceways each had a water volume of 32.7 m^3. Mean nutrient parameters at the raceway inflow coming from upstream ponds were as follows (±standard deviation): 5.12 (±0.61) mg L^{-1} TN, 0.14 (±0.06) mg L^{-1} TAN, 0.017 (±0.005) mg L^{-1} nitrite (NO_2-N), 4.95 (±0.76) mg L^{-1} nitrate (NO_3-N), 0.058 (±0.098) mg L^{-1} TP, 0.031 (±0.023) mg L^{-1} phosphate (PO_4-P), and a particle load of 2.70 (±3.27) mg L^{-1} TSS. Biological oxygen demand over 5 days (BOD_5) was 2.41 (±1.07) mg L^{-1}, average pH was 7.73 (±0.90) and conductivity was 723 (±14) μS cm^{-1} (Sindilariu et al., 2007; measured confirm DIN, 2006).

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During the study period (07.09.2006–06.09.2007), stock densities in the raceways ranged from 36.7 to 70.3 kg m^{-3} of trout weighing 100–700 g each. Every fourth week, density was corrected by harvesting a quantity of fish equivalent to the weight gained. Thus the feed input remained constant at about 30 kg day^{-1}. Details on the stocking and feeding procedure throughout the experiment are described in Sindilariu et al. (2008).

2.1.2. *Water distribution*

The average combined raceway runoff from August 2006 to September 2007 was 13.4 (±0.7) L s^{-1}. Both raceways flushed via a shared 150 mm diameter pipe leading to a central distribution box from which outflow was directed in a controlled fashion to each of the six wetland cells (Sindilariu et al., 2008). Of these two cells were designated high flow cells, receiving an average hydraulic load of 4.0 (±0.3) L s^{-1}, two were medium flow cells receiving on average 1.9 (±0.2) L s^{-1} and the remaining two cells received a low flow averaging 0.9 (±0.1) L s^{-1}.

2.1.3. *SSF constructed wetland*

The SSF constructed wetland under study had been in use since June 2004. The cells were modified from previous use as sedimentation basins (further details in Sindilariu et al., 2007, 2008). Each wetland cell had an open space pre-sedimentation area of 9.6 m^2 and a wetland bed of 1 m depth and 23.9 m^2, filled with 4–8 mm local available gravel (porosity ε = 0.36, specific gravity 1781 kg m^{-3}, TP content of 6 μg g^{-1}), planted with a natural plant community dominated by *Phragmites communis* and *Phalaris arundinacea* (Sindilariu et al., 2007). Hydraulic load on the wetland bed of the high, medium and low flow cells was 14.5, 6.9 and 3.3 m^3 m^{-2} day^{-1}, respectively.

2.2. *Sampling and analysis*

Water samples were taken from the outflows of the raceways and the wetland cells once a week throughout the 12-month sampling period. Sampling details are described in Sindilariu et al. (2008).

On 23 out of 52 sampling days, four YSI 600 XLM data loggers (Yellow Spring Instruments Inc.) were used to measure inflow and outflow temperature, conductivity (cond.), dissolved oxygen (DO) and pH of one high, one medium and one low flow cell. Measurements of these physical water parameters were logged every 10 min throughout the 24 h sampling period.

The 24 h pooled water samples were analysed for TN (mg L^{-1}); TAN (mg L^{-1}); NO_2-N (mg L^{-1}); NO_3-N (mg L^{-1}); total phosphorus, TP (mg L^{-1}); PO_4-P (mg L^{-1}); biological oxygen demand BOD_5 (mg L^{-1}) and dry weight of total suspended solids, TSS (mg L^{-1}). The physicochemical properties of the samples were determined following German standard methods for the analysis of water, wastewater and sludge (DIN, 2006). Biological oxygen demand was measured as the total oxygen consumption of the unmodified water sample over five days, and the particulate matter in the sample was not destroyed prior to measurement. This means of assessment for BOD_5 was used in order to maintain the comparability with the effluent margins set by local water authorities, for which BOD_5 is measured by the same method.

2.3. *Calculations and statistics*

Differences (Δp) between inflow and outflow concentrations were calculated for each parameter as well as each pair of simultaneous samples. The relative treatment efficiency (%Δ) was calculated for each parameter as:

$$\%\Delta = (\Delta p \cdot c_{\text{in}}^{-1}) \cdot 100$$

where Δp = inflow-outflow concentration in mg L^{-1} and c_{in} = inflow concentration in mg L^{-1}. Hydraulic loading rates (HLR in m day^{-1}) were calculated as:

$$\text{HLR} = 86.4 \cdot Q \cdot A^{-1}$$

where Q = inflow rate in L s^{-1}, and *A* is the wetland bed surface area in m^2. The related area loading rates (AL in g m^{-2} day^{-1}) and area removals (AR in g m^{-2} day^{-1}) of the wetland area applied, were calculated as:

$$\text{AL} = c_{\text{in}} \cdot \text{HLR}; \qquad \text{AR} = \Delta p \cdot \text{HLR}$$

For the nitrogen, BOD_5 and DO budgeting the following formulae after Tchobanoglous et al. (2003) were used:

The particulate nitrogen (PN) with in the system was calculated as:

$$\text{PN} = \text{TN} - (\text{TAN} + \text{NO}_2\text{-N} + \text{NO}_3\text{-N})$$

It is supposed that theoretically all PN is available for nitrification, as the PN source are mainly undigested protein-residuals derived from fish feed. Thus the availability of PN for nitrification is dependent on the oxygen amount within the wetland and will decrease with decreasing oxygen saturation, especially at low and very low values. Due to the high nitrogen mobility within the wetland from different storage forms (Kadlec et al., 2005) it can be assumed that at about 40% DO saturation, 100% of PN can be nitrified. At DO saturations below 40% in the wetland outflow, the amount of PN available for nitrification (n_{PN}) is calculated as:

$$n_{\text{PN}} = \left(\frac{\text{DO\%}}{40\%}\right) \cdot \Delta p(\text{PN})$$

Thus the total nitrification (*n*) within the wetland cells is calculated:

$$n = \Delta p(\text{TAN}) + n_{\text{PN}}$$

A part of the nitrified TAN and PN is released as NO_2-N and NO_3-N through the effluent out of the system, the remaining amount is denitrified. The amount of denitrification (d*n*) is calculate as:

$$\text{d}n = n - (\Delta p(\text{NO}_2\text{-N}) + \Delta p(\text{NO}_3\text{-N}))$$

Per milligram of nitrogen denitrified, about 3.5 mg of BOD_5 is consumed as electron donor (Tchobanoglous et al., 2003; van Rijn et al., 2006). Thus the theoretical DO consumption ($DO_{\text{con.}}$, mg L^{-1}) was calculated as:

$$\text{DO}_{\text{con.}} = \Delta p(\text{BOD}_5) - 3.5 \cdot \text{d}n + 4.25 \cdot n$$

The inflow and outflow nutrient concentration, removal efficiencies and treatment efficiencies at each of the three hydraulic loads were compared by one-way ANOVA. Where significant, the means were tested by post hoc Tukey–Kramer-test. In cases of heteroscedasticity, a Welch ANOVA with

Wilcoxon-test and a final Bonferoni adjustment were performed. Significance was identified at a probability level of $p < 0.05$. Statistical calculations were performed with the SAS 8e software package.

Variations in treatment efficiency dependent on hydraulic load were tested by the following multivariate regression model: $Y_{ijklm} = \mu + \alpha_i + \beta_j + \gamma_k + \delta_l + \varepsilon_m + (\alpha_i\gamma_k) + (\beta_j\gamma_k) + \zeta_{ijklm}$, where Y_{ijklm} is the treatment efficiency, $\%\Delta$ (log transformed to achieve linearity), μ is the overall log $\%\Delta$, α_i is the inflow nutrient concentration, β_j is the hydraulic load, γ_k is the accumulation of TSS in the wetland bed, the block factors δ_l and ε_m represent vegetation period (YES/NO), and fish harvesting (YES/NO) respectively, $(\alpha_i\gamma_k)$ and $(\beta_j\gamma_k)$ denote the interaction between two effects and ζ_{ijklm} is the random residual error. For nitrite-nitrogen, the inflow concentration (α_i) was represented by TAN, as the only source for nitrite build-up in the system, while log transformed values for TP were used as the relevant α_i for soluble reactive phosphates. Model calculations were carried out using Jump 7.0.1 (SAS Institute).

2.4. Financial cost calculation

Cost calculations were based on the financial budget needed for converting existing sedimentation basins into the SSF wetland cells examined in this study. The total construction costs were €13,900 for 143 m^2. Of this, €9990 went on expenses such as planning, metal work and piping, the values of which are expected to depreciate over a fixed period of 30 years. The remaining €3910 relates to expenses whose value will depreciate within the service lifetime of the wetland, i.e. gravel replacement and reed planting. This overall expenditure was transformed to a yearly cost which incorporated interest, and maintenance expenses, as is usual in the costing of agricultural buildings (Steinhauser et al., 1989). To arrive at an estimate of overall annual expense, the variable costs of wetland treatment were added.

The service lifetime was calculated based on an efficiency limit of 50% for the removal of TAN. When the treatment efficiency drops consistently below this limit the wetland bed has to be renewed. TAN was chosen as relevant nutrient parameter, in contrast to phosphate which is usually the limiting factor for municipal wetlands (Drizo et al., 1999), due to the fact that in commercial salmonid farms about 80% of TP is particle bound and can be removed by mechanical treatment (Cripps and Bergheim, 2000), thus phosphorus is the last limiting factor for in aquaculture (Sindilariu, 2007). The wetland area requirement was calculated based on three different hydraulic load regimes for wetlands processing the outflow of a theoretical 100 $L\,s^{-1}$ flow trout farm.

Annual costs for the micro-screening pre-treatment of TSS are €4720 for micro-screening itself and €3190 for the processing of the backwash sludge (unpublished data from a commercial Bavarian trout farm). For the purposes of this calculation we assumed TSS pre-treatment efficiencies of 50% and 80%, representing a typical value for current commercial micro-screening (Bergheim et al., 1998), and a maximum value achievable using recently developed farm management techniques (Brinker and Rösch, 2005).

3. Results

3.1. Wetland inflow and outflow water quality

The SSF wetland had a significant effect on all the measured water parameters except conductivity, where no influence was found (Table 1). Dissolved oxygen (DO) decreased significantly with decreasing hydraulic load, from 93 (±35)% saturation at the wetland inflow to 36 (±30)%, 19 (±26)% and 13 (±10)% at the high, medium and low flow cell outflows respectively. Regarding the measured water parameters, the wetland provided a treatment effect for both solid and dissolved fractions: total suspended solids, total phosphorus and nitrogen, BOD_5 and TAN were all significantly reduced by treatment and in most cases the effluent concentrations of nutrients decreased significantly with decreasing hydraulic load. The exception was total phosphorus, where no significant effect of hydraulic load was observed. All three wetland treatments yielded a significant increase in dissolved nitrites, nitrates and phosphates, with the exception of the medium flow cells in which no significant increase in nitrite-nitrogen concentration was observed.

At the wetland inflow, soluble reactive phosphates accounted for 10% of total phosphorus. This proportion

Table 1 – Mean in- and outflow concentrations and standard deviation (in brackets) of the SSF wetland in depending on the hydraulic load. Different upper case letters indicate significant differences ($p < 0.05$).

Water parameter (SD)	Average inflow 13.4 $L\,s^{-1}$	Average outflow I 4.0 $L\,s^{-1}$	Average outflow II 1.9 $L\,s^{-1}$	Average outflow III 0.9 $L\,s^{-1}$
Temperature (°C)	10.25^{A} (1.46)	10.32^{B} (1.63)	10.37^{BC} (1.65)	10.37^{C} (1.85)
Electric cond. ($\mu S\,cm^{-1}$)	765^{A} (22.4)	766^{A} (12.8)	766^{A} (32.9)	766^{A} (16.1)
DO ($mg\,L^{-1}$)	10.17^{A} (3.81)	3.93^{B} (3.25)	2.10^{C} (2.82)	1.41^{D} (1.06)
pH	7.59^{A} (0.29)	7.71^{B} (0.12)	7.51^{C} (0.19)	7.51^{C} (0.19)
TN ($mg\,L^{-1}$)	6.16^{A} (0.78)	5.80^{B} (0.47)	5.65^{B} (0.57)	5.43^{C} (0.46)
TAN ($mg\,L^{-1}$)	0.736^{A} (0.184)	0.423^{B} (0.225)	0.111^{C} (0.089)	0.104^{D} (0.113)
NO_2-N ($mg\,L^{-1}$)	0.097^{A} (0.046)	0.157^{B} (0.121)	0.124^{A} (0.122)	0.113^{C} (0.168)
NO_3-N ($mg\,L^{-1}$)	4.78^{A} (0.21)	5.02^{B} (0.41)	5.28^{C} (0.45)	5.21^{C} (0.51)
TP ($mg\,L^{-1}$)	0.341^{A} (0.433)	0.156^{B} (0.046)	0.144^{B} (0.033)	0.141^{B} (0.029)
PO_4-P ($mg\,L^{-1}$)	0.034^{A} (0.018)	0.090^{B} (0.026)	0.109^{C} (0.026)	0.115^{C} (0.030)
BOD_5 ($mg\,L^{-1}$)	6.79^{A} (2.13)	3.66^{B} (2.09)	1.56^{C} (0.95)	1.17^{D} (0.62)
TSS ($mg\,L^{-1}$)	10.30^{A} (11.87)	3.53^{B} (4.54)	1.78^{C} (2.77)	1.13^{D} (1.06)

Table 2 – Area loading (AL), area retention (AR) ($g\,m^{-2}\,day^{-1}$) and the percentage of mass removed of the SSF wetland cells in dependence on the hydraulic load. Different upper case letters indicate significant differences ($p < 0.05$, $N = 312$).

Water parameter	AL	AR	%Removed	AL	AR	%Removed	AL	AR	%Removed
	$4.0\,L\,s^{-1}$			$1.9\,L\,s^{-1}$			$0.9\,L\,s^{-1}$		
TN	89.1	5.11^{A}	5.7	42.3	3.48^{B}	8.2	20.0	2.38^{C}	11.9
TAN	10.6	4.52^{A}	42.6	5.06	4.29^{A}	84.8	2.39	2.06^{B}	86.2
NO_2-N	1.40	-0.88^{A}	−62.9	0.67	-0.19^{B}	−28.4	0.32	-0.06^{B}	−18.8
NO_3-N	69.1	-3.47^{A}	−5.4	32.8	-3.39^{A}	−10.3	15.6	-1.39^{B}	8.9
TP	4.93	2.67^{A}	54.2	2.34	1.35^{B}	57.7	1.11	0.65^{C}	58.6
PO_4-P	0.49	-0.83^{A}	−169.4	0.23	-0.53^{B}	−230.4	0.11	-0.27^{C}	−245.5
BOD_5	98.2	45.3^{A}	46.1	46.6	36.0^{A}	77.3	22.1	18.3^{B}	82.8
TSS	148.9	97.9^{A}	65.7	70.7	58.5^{B}	82.7	33.5	29.8^{C}	89.0

increased to 58%, 76% and 82% at the outflow from high, medium and low flow cells, respectively (Table 1).

3.2. *Area loading and area retention*

Nutrient to area loading (AL) increases with hydraulic load and thus for most of the measured nutrient parameters, area retention (AR) also increased significantly (Table 2). For TAN and BOD_5, however, there was no significant difference in AR between the high and medium flow cells. Thus there was no significant difference in the release of nitrates between the two hydraulic loads. There was no significant difference in nitrite release between the medium and low flow cells.

3.3. *Wetland treatment efficiency*

Trends in treatment efficiency varied in response to hydraulic load and corresponded largely to those observed in nutrient concentrations, with the exception that the improvement in total nitrogen levels between high and medium flow rates was also found to be significant (Fig. 1). The highest mean treatment efficiencies were reached at the lowest hydraulic load ($0.9\,L\,s^{-1}$), with 86%, 82% and 79% efficiency for TAN, BOD_5 and TSS, respectively. Wetland treatment increased PO_4-P concentration in the outflow compared to the inflow by 5–7 times to $0.09–0.12\,mg\,L^{-1}$ (Fig. 1).

The amount of PN in the wetland effluent also decreases with decreasing hydraulic load (Table 3). Between 33% and 90% of PN are reduced through nitrification, this leads to a nitrification rate (n) of 0.63, 0.82 and $0.81\,mg\,L^{-1}$ for the cells receiving 4.0, 1.9 and $0.9\,L\,s^{-1}$, respectively. Taking in account the release of nitrite and nitrate, 0.33, 0.29 and $0.35\,mg\,L^{-1}$ nitrogen were lost through denitrification in the high, medium and low flow cells respectively. The nitrogen cycling and the measured BOD_5 metabolism lead to a theoretical DO consumption (DO_{con}) of 4.65, 5.74 and $5.52\,mg\,L^{-1}$, representing 75%, 71% and 63% of the measured DO loss throughout the wetland

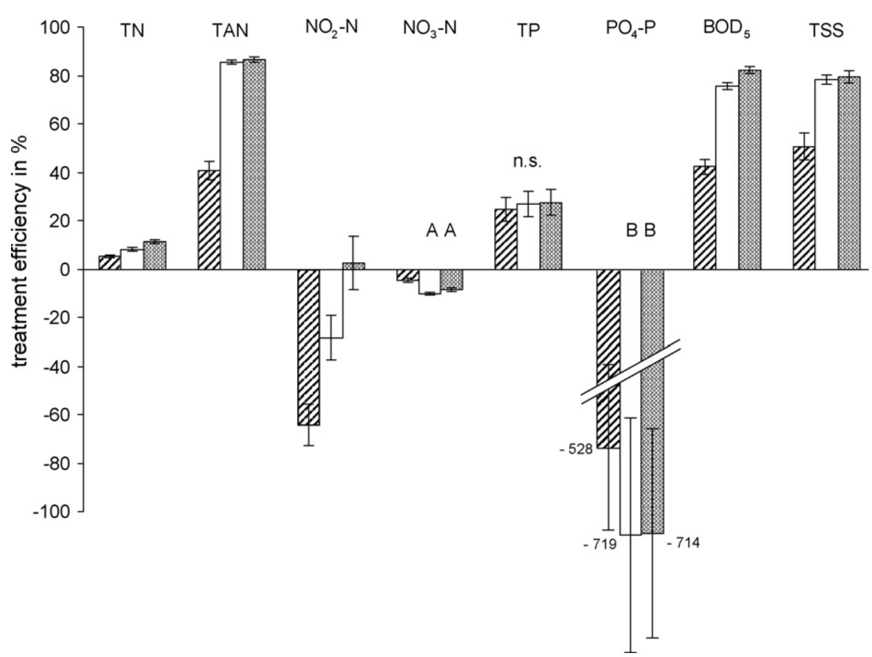

Fig. 1 – Wetland treatment efficiency in dependence on hydraulic load, as mean of each weakly measured treatment efficiency, with the indication of standard error. (Always from left to right, 4.0, 1.9, $0.9\,L\,s^{-1}$ hydraulic load; n.s. not significant, letters refer to homogenous subgroups, all other were significantly different $p < 0.05$.)

Table 3 – Relevant data for nitrification, denitrification, BOD_5, DO and nitrogen flux estimation, calculated from Table 1 following the formulae given in Section 2.

Water parameter (mg L^{-1})	Average outflow I (4.0 L s^{-1})	Average outflow II (1.9 L s^{-1})	Average outflow III (0.9 L s^{-1})
PN	0.20	0.14	0.03
Δp PN	0.35	0.41	0.54
Amount of PN nitrified (%)	90	48	33
n_{PN}	0.32	0.19	0.18
Δp TAN	0.313	0.625	0.632
n	0.63	0.82	0.81
Δp (NO_2-N) + Δp (NO_3-N)	0.30	0.53	0.45
dn	0.33	0.29	0.35
3.5·dn	1.15	0.87	1.04
Δp (BOD_5)	3.13	5.23	5.62
DO_{con}	4.65	5.74	5.52
Δp (DO)	6.24	8.07	8.76
($DO_{con}/\Delta p$ (DO)) (%)	75	71	63

PN = amount of particulate Nitrogen; Δp PN = wetland inflow–outflow concentration of PN; n_{PN} = amount of PN nitrified; dn = amount of denitrification; DO_{con} = the calculated DO consumption.

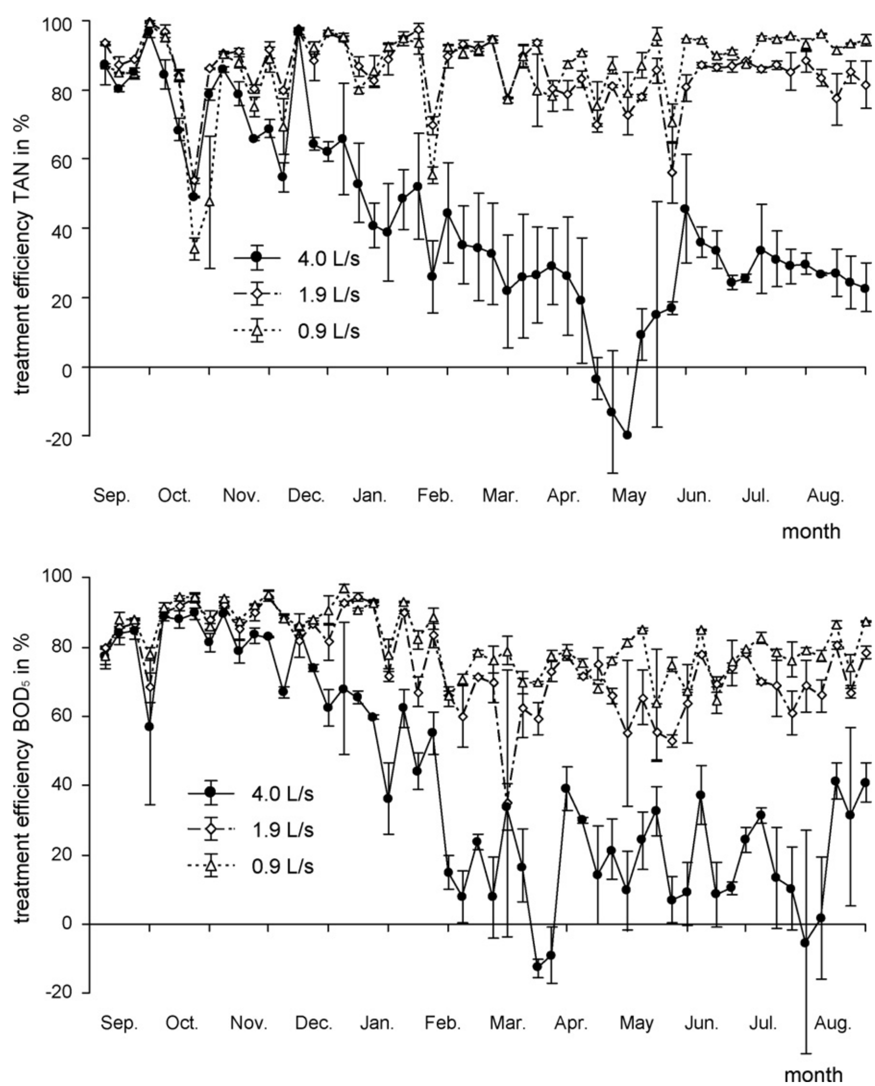

Fig. 2 – Temporal development of the treatment efficiencies of BOD_5 and TAN in dependence on the hydraulic load.

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passage, for 4.0, 1.9 and $0.9\,L\,s^{-1}$ hydraulic load, respectively (Table 3).

3.4. *Temporal development of treatment efficiency*

Six to eight weeks after the start of the sampling period, the water started to flow over the gravel filter (wetland bed) in the cells receiving $4.0\,L\,s^{-1}$ due to clogging of the filter pores caused by TSS accumulation. The amount of water flowing over the wetland surface increased successively during the sampling period. Comparing the temporal development of treatment efficiencies for TAN and BOD_5, there is no clear difference between the three treatment groups at the beginning of the sampling period (Fig. 2). From the 14th week on, however, treatment efficiency for both parameters began to decrease in the high flow wetland cells and did not recover until the end of the sampling period. The efficiencies of the cells receiving lower hydraulic loads remained relatively stable over the whole period, with the low flow cells performing slightly better overall (Fig. 2).

3.5. *Modelling of wetland treatment efficiency*

A multivariate regression model was used to assess the factors influencing wetland treatment efficiency. The results of the statistical analysis were presented in Table 4. Comparing the scaled estimates calculated in the modelling process, the wetland inflow nutrient concentration, TSS accumulation, hydraulic load and the interaction between wetland inflow concentration and TSS accumulation are the most important factors influencing treatment efficiency (significant scaled estimate with the highest absolute value). Vegetation period, fish harvesting and the interaction between inflow nutrient concentration and hydraulic load are of lesser importance (Table 4 for TAN, TP, BOD_5 and TSS, other nutrient parameters not shown).

The efficiency with which wetland treatment improved nutrient parameters associated with the particulate fraction (TN, TP, BOD_5 and TSS) increased with increasing wetland inflow concentration. Treatment efficiency for the dissolved nutrients (TAN, nitrites, nitrates and phosphates), on the other hand, decreased with increasing wetland inflow concentration, resulting in a respective increase in nutrient release (see also Table 4). Wetland inflow concentration ranges were ($mg\,L^{-1}$): TN (5.03–10.16), TAN (0.26–1.18), NO_2-N (0.033–0.305), NO_3-N (4.36–5.57), TP (0.070–3.063), PO_4-P (0.000–0.072), BOD_5 (2.90–12.53) and TSS (1.24–67.99). Treatment efficiency for most nutrients decreased as solids accumulated in the wetland bed, but there was no significant effect on total nitrogen or nitrite levels over time and the release of nitrate was actually reduced as solids accumulated.

Fig. 1 indicates a negative effect of increased hydraulic load on wetland treatment efficiency, with three exceptions: Total phosphorus and nitrates showed no significant relationship with hydraulic load, while soluble phosphate release was significantly greater with reduced hydraulic load. These findings were backed up by the scaled estimates. Vegetation period had a positive treatment effect on total phosphorus, nitrites and nitrates, but a negative effect on phosphates, BOD_5 and TSS

Table 4 – Linear model of SSF wetland treatment efficiency ($N=312$, $P<0.0001$) for TAN, TP, BOD_5 and TSS. Factors with a positive scaled estimate have a positive effect on treatment efficiency, for negative scaled estimates the factor has a negative impact on treatment efficiency.

Variables	Nutrient parameter											
	TAN			TP			BOD_5			TSS		
	Scaled estimate	S.E.	p	Scaled estimate	S.E.	p	Scaled estimate	S.E.	p	Scaled estimate	S.E.	p
Intercept	−1.2551*	0.0239	<0.0001	−1.7711*	0.0057	<0.0001	−1.4015*	0.0111	<0.0001	−1.2498*	0.0201	<0.0001
Inflow concentration	−0.1228*	0.0604	0.0428	0.7682*	0.0129	<0.0001	0.1844*	0.0236	<0.0001	0.7557*	0.0602	<0.0001
Hydraulic load	−0.2432*	0.0325	<0.0001	0.0125	0.0082	0.1287	−0.1753*	0.0159	<0.0001	−0.0878*	0.0285	0.0022
Cumulative TSS load	−0.3197*	0.0639	<0.0001	−0.0843*	0.0157	<0.0001	−0.1671*	0.0303	<0.0001	−0.2279*	0.0540	<0.0001
Vegetation period (YES)	−0.0266	0.0256	0.3004	0.0187*	0.0064	0.0035	−0.1411*	0.0125	<0.0001	−0.1793*	0.0219	<0.0001
Harvesting (YES)	−0.0068	0.0273	0.8035	−0.0106	0.0058	0.0690	−0.0155	0.0113	0.1723	−0.0269	0.0201	0.1833
Inflow concentration × hydraulic load	−0.1721*	0.0772	0.0265	−0.0026	0.0228	0.9087	0.0914*	0.0317	0.0042	0.5391*	0.1202	<0.0001
Inflow concentration × cumulative TSS load	0.5810*	0.1261	<0.0001	−0.1277*	0.0402	0.0016	−0.2946*	0.0671	<0.0001	−1.3354*	0.2199	<0.0001
R^2		**0.53**			**0.93**			**0.78**			**0.56**	

* Significant scaled estimates (S.E. = standard error, p = significance level).

Table 5 – Calculated service lifetime (years) for TAN treatment efficiency >50% and yearly effluent treatment costs (€) of the SSF wetland treatment for a 100 $L\,s^{-1}$ example trout farm with an annual production of about 770 kg $(L\,s^{-1})^{-1}$, in dependence on hydraulic load and TSS pre-treatment.

	No pre-treatment	50% TSS treatment micro-screen	80% TSS treatment micro-screen
Hydraulic load ($m^3\,m^{-2}\,day^{-1}$)	14.5	14.5	14.5
Wetland area (m^2)	600	600	600
Service lifetime (years)	0.67	1.4	3.5
Annual costs SSF wetland (€)	27,680	14,690	7,540
Total annual costs with micro-screen (€)	**27,680**	**22,600**	**15,450**
Hydraulic load ($m^3\,m^{-2}\,day^{-1}$)	6.9	6.9	6.9
Wetland area (m^2)	1,255	1,255	1,255
Service lifetime (years)	2.0	3.9	9.8
Annual costs SSF wetland (€)	23,410	14,850	9,430
Total annual costs with micro-screen (€)	**23,410**	**22,760**	**17,340**
Hydraulic load ($m^3\,m^{-2}\,day^{-1}$)	3.3	3.3	3.3
Wetland area (m^2)	2,650	2,650	2,650
Service lifetime (years)	4.6	9.3	13.7
Annual costs SSF wetland (€)	28,460	20,310	17,750
Total annual costs with micro-screen (€)	**28,460**	**28,220**	**25,660**

treatment. Fish harvesting appeared to impact only two nutrients, with a negative effect on total nitrogen treatment, and a positive effect on nitrite removal efficiency. The certainty measures (R^2) of the model for the parameters not shown in Table 4, were 0.56, 0.37, 0.40 and 0.14 for total nitrogen, nitrites, nitrates and phosphates, respectively.

3.6. *Wetland cost calculation*

Using the scaled estimate for TAN treatment efficiency from Table 4, the service lifetime of the wetland was calculated in respect of different hydraulic loads at a mean inflow concentration. The effects on overall treatment costs of TSS pre-treatments prior to the wetland processing were also estimated, based on a standard 100 $L\,s^{-1}$ trout farm (Table 5). The combination of effective pre-treatment (80% TSS removal) with small constructed wetlands processing high hydraulic loads, turned out to be economically most feasible, causing annual costs of €15,450. For a 100 $L\,s^{-1}$ trout farm with an annual production of 770 kg $(L\,s^{-1})^{-1}$ this represents a production cost increase of €0.20 kg^{-1}.

4. Discussion

The identification of three main factors influencing the efficiency of wetland treatment creates opportunities for the management and design of SSF constructed wetlands for the treatment of wastes from intensive commercial trout farms. The dimensions, efficiency and costs of comparable SSF wetland systems can be estimated based on measured values for inflow nutrient concentrations and hydraulic load. Furthermore, the results detailed above allow recommendations for efficient and cost-effective treatment regimes to be made. The price of environmental sound fish production based on these findings may be as low as €0.20 kg^{-1}.

4.1. *Effluent treatment*

The efficacy of the sub-surface flow (SSF) wetland in removing particle bound nutrients was as good, if not better than that, which might be expected of micro-screen treatment (50% to maximum 87%) (Bergheim et al., 1998; Bergheim and Brinker, 2003; Brinker and Rösch, 2005; Brinker et al., 2005). As is typical for SSF systems used in aquaculture, the wetland proved highly effective in the removal of total ammonia nitrogen (Sindilariu et al., 2007) and experimental scale aquaculture effluent treatment (Schulz et al., 2003; Lin et al., 2005).

Treatment efficiency of the high flow wetland cells (receiving 4.0 $L\,s^{-1}$) decreased over time (Fig. 2.) due to increasing overflow as the wetland bed became colmatated with trapped solids. The cells were thus gradually transformed to surface flow wetlands, which typically exhibit lower treatment efficiency for aquaculture effluents (Redding et al., 1997; Schulz et al., 2004; Lin et al., 2005).

The treatment mechanisms operating within SSF constructed wetlands have been discussed extensively. The wetland bed provides an effective filter for all suspended solids and the nutrients associated with the particulate matter (Tanner et al., 1999; Wynn and Liehr, 2001; Schulz et al., 2003; Stottmeister et al., 2003; Lin et al., 2003, 2005).

For the phosphorous fraction, leaching and microbial phosphate excretion are time dependent processes that may both contribute to an increase in dissolved phosphorus proportionate to hydraulic residence time (HRT) (Stewart et al., 2006). Chemical precipitation and absorption have previously been ascribed an important role in total phosphorus reduction in constructed wetlands (Drizo et al., 1999; Lantzke et al., 1999; Maehlum and Stalnacke, 1999; Vymazal, 2005), however the current study found no evidence for this. The precipitation potential of the wetland bed is limited and was most likely already saturated (Arias et al., 2001; Del Bubba et al., 2003; Seo

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et al., 2005), as a result of the high area retention of total phosphorus, the age of the wetland and the coarse nature of the filter material.

For the nitrogen fraction, a distinct nutrient spiralling occurs between particulate and dissolved matter, biological nutrient incorporation, decay and leaching (Kadlec et al., 2005). The nitrogen, BOD_5 and DO fraction within the wetland are highly related and depend on each other. For the dissolved TAN and parts of the PN fraction, however, in this wetland considerable nitrification is expected, as usual for aquaculture bio-filters with highest rates of $1.1\,g\,m^{-2}\,day^{-1}$ in relation to bio-filter surface (Eding et al., 2006). Only between 35% and 55% of the nitrate built in the nitrification process is again consumed through denitrification (Table 3), in contrast to municipal wetlands, where nitrification is the limiting factor and excess denitrification occurs (Platzer, 1999; Wynn and Liehr, 2001; Tanner et al., 2002; Stottmeister et al., 2003; Liu et al., 2005). Heterotrophic decomposition and nitrification are the most reliable means of reducing BOD_5 (Geller, 1997; Vymazal, 2005; Garcia et al., 2005; van Rijn et al., 2006; Akratos and Tsihrintzis, 2007). The SSF system provided ample opportunity for this to take place, as evidenced by the observed rates of DO consumption. However, the measured DO consumption was between 25%, 29% and 37% higher than theoretical calculated for the high, middle and low flow wetland cells, respectively, most probably due to additional carbon input from decaying plants, roots, root exudates and decaying microbial fauna.

4.2. *Treatment efficiency*

In agreement with previous research on municipal wastewater wetlands (Kadlec, 2000), the main factors affecting treatment efficiency are the inflow nutrient concentration and the hydraulic loading rate (HLR). However, in this case, TSS accumulation in the wetland bed was also found to be an important factor for treatment efficiency (Table 4).

4.2.1. *Vegetation period, fish harvesting*

Vegetation period and fish harvesting are of significant but minor importance compared to the other factor influencing wetland treatment efficiency. However, the role of plants in constructed wetlands has been widely discussed. In most cases planted SSF wetlands perform much better than the unplanted controls (Tanner et al., 1999; Akratos and Tsihrintzis, 2007). Nutrient removal through plant biomass is a minor contributor to overall removal efficiency in the examined wetlands. The maximum retained nutrient amount for *Phragmites communis*, the dominant plant species, is 57.6 and $6.1\,g\,m^{-2}\,year^{-1}$ for TN and TP, respectively (Tanner, 1996), representing about 6.6% and 2.6% of the lowest area retention measured in the wetland. The main advantages of plants is that they provide suitable conditions for effective nutrient treatment (Weaver et al., 2003) and they improve the aesthetic appearance of the facility (Tanner and Sukias, 1995; Brix, 1997; Stottmeister et al., 2003; Vymazal, 2005). The specific helophyte species used appears to be of minor importance (Tanner, 1996).

4.2.2. *Inflow nutrient concentration*

In trout aquaculture the quality of nutrients in the effluent is dependent on production intensity, which is reflected in the amount of feed applied per unit volume of inflow. Nutrient concentrations in effluent can be calculated by reference to feed quality and estimations of feed wastes (Bergheim and Asgard, 1996; Bureau et al., 2003).

The removal of particulate nutrients is a linear process (Mitchell and McNevin, 2001). For dissolved nutrients, treatment efficiency decreases with increased inflow concentration. Nitrification seems to be a constant process in which the concentration of total ammonia nitrogen is reduced by transformation to nitrite. This process is limited by the number of bacteria available to perform it (Mitchell and McNevin, 2001). Where environmental conditions are stable (as in the present study) the quantity of nitrifying bacteria is limited by the availability of substrates for attached growth (Eding et al., 2006). Thus, the amount of nitrogenous waste that can be processed in a certain period of time is limited. The nitrification rate remains constant regardless of increasing concentrations of TAN, and overall treatment efficiency is thereby reduced. For the other dissolved nutrients, release increases with load at the inflow.

4.2.3. *Hydraulic load*

The hydraulic load has an inverse relationship with hydraulic residence time (HRT) in the wetland. Increased HRT has a positive effect on wetland treatment efficiency for most nutrient fractions (Garcia et al., 2005; Akratos and Tsihrintzis, 2007). In this study, treatment of all measured nutrient fractions except nitrates, total phosphorus and phosphates were positively influenced by increasing HRT.

4.2.4. *Accumulation of TSS*

The accumulation of suspended solids (TSS) in the root system had a negative effect on the efficiency of treatment for most nutrients except total nitrogen and dissolved nitrites. Nitrate release was highly reduced with TSS accumulation, through reduced nitrification and also improved denitrification (Table 3) through a higher amount of biodegradable volatile suspended solids (VSS) available, as part of TSS, serving as carbon source. Wetland clogging reduces HRT, leading to short-circuiting of sub-surface flow and finally to partial or even total overflow of the wetland (Tanner and Sukias, 1995; Blazejewski and Murat-Blazejewska, 1997; Vymazal, 2005). However, TSS volume does appear to be reduced within the wetland, through the biological degradation of VSS, as more waste appeared to accumulate than theoretical limit based on the available pore volume (36%) would predict. With reduced HRT and short-circuited flow, the efficiency of time dependent microbial processes (nitrification and heterotrophic decomposition) decreases. Overflow also reduces the mechanical treatment effect of the filter matrix, leading to reduced particulate removal. With higher particle accumulation, the amount of particulate phosphorus in the filter matrix increases, leading to increased leaching of phosphates.

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4.3. Financial costs of SFF wetland treatment

In Germany, thresholds only for TSS, BOD_5, TP and TAN were applied until now. TSS, TP and BOD are mainly particle bound so they can easily be removed through mechanical effluent treatment. The first limiting effluent factor concerning production increase with the application of mechanical treatment only is dissolved BOD_5, followed by TAN (Sindilariu, 2007). BOD_5 can be easy removed through surface flow wetlands (Schulz et al., 2004) or fixed bed submerged biological filters (Brinker et al., 2006) as additional treatment. Thus, the most important nutrient fraction SSF wetlands, used for trout effluents, have to deal with is TAN (Sindilariu, 2007).

The service lifetime of the SSF wetlands in the cost calculation was set when a TAN treatment efficiency of 50% was reached. The financial cost calculation is based on the transition of already available basins to SSF wetlands. If the wetlands were newly implemented, then constructional costs, which have to be depreciated on at least 30 years, have to be added to the cost calculation. The transition costs from sedimentation basin to SSF wetlands were saved. This might slightly increase the total costs of wetland implementation indicated in Table 5.

Three main factors influence wetland treatment efficiency. The first, inflow nutrient concentration, is directly dependent on the farm production intensity (Brinker et al., 2006). The remaining two factors, hydraulic load and TSS accumulation, are important considerations in wetland dimensioning, and have implications for service lifetime and financial costs at any given production intensity. The results of the current study allow a realistic estimation of costs for commercial scale wetland treatment systems for intensive trout production, based only on these three factors (Table 5).

The hydraulic load influences the wetland area needed, while TSS pre-treatment directly influences the wetland service lifetime. An efficient combination between service lifetime and land requirements depends on the distribution of costs between fixed-term and service lifetime depreciations. Here the costs depreciated on a fixed term represented 72% of the total and the economic benefit of reduced area was therefore greater than that of prolonged service lifetime. The most cost-effective strategy is to reduce the area of wetland required by applying the highest possible hydraulic load and reducing the rate of TSS accumulation in the wetland through effective pre-treatment.

As a stand-alone treatment for effluent, SFF constructed wetlands are not suitable for intensive trout farms. The annual costs of €23,000–28,000 to treat $100\,L\,s^{-1}$ effluent are prohibitive (Table 5). However, the costs decrease remarkably when SSF wetlands are used in conjunction with effective pre-treatment. In the present case, the up-scaled cost of trout production was an additional €$0.20\,kg^{-1}$, an expense fisheries managers and consumers should find justifiable when nutrient emission is kept to a minimum.

5. Conclusions and recommendations

1. Constructed wetlands reach high treatment efficiencies for effluent from intensive commercial scale trout farms.
2. Treatment efficiency and the wetland implementation costs are dependent on the concentration of nutrients flowing into the wetland, the hydraulic load and accumulation of TSS in the wetland bed.
3. The bottleneck for wetland application is the accumulation of TSS in the wetland bed determining the service lifetime. Due to high costs, SSF wetlands are not suitable as a stand-alone treatment method for intensive trout farm effluents, but they make an economically feasible solution when used in combination with effective TSS pre-treatment.
4. The combination of mechanical effluent sieving with SSF wetlands is a suitable solution for intensive production sites, where the environmental thresholds cannot meet with mechanical treatment alone.

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