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REVIEW

Wastewater treatment for land-based aquaculture: improvements and value-adding alternatives in model systems from Australia

Sarah A. Castine[1,4,*], A. David McKinnon[2], Nicholas A. Paul[3], Lindsay A. Trott[2], Rocky de Nys[3]

[1]AIMS@JCU, Australian Institute of Marine Science and School of Marine and Tropical Biology, James Cook University, Townsville, Queensland 4811, Australia

[2]Australian Institute of Marine Science, Townsville MC, Queensland 4810, Australia

[3]School of Marine and Tropical Biology, James Cook University, Townsville, Queensland 4811, Australia

[4]*Present address:* WorldFish, Jalan Batu Maung, 11960 Bayan Lepas, Penang, Malaysia

ABSTRACT: Settlement ponds are used to remove particulate and dissolved nutrients in Australian land-based aquaculture wastewater. At best, marine and brackish water settlement ponds reduce total suspended solids by 60%, but their efficiency is inconsistent. Functional improvements to nutrient removal systems are essential to provide uniform and predictable treatment of flow-through aquaculture wastewater. Furthermore, environmental regulation of discharge from intensive systems in Australia is increasing, providing the impetus to upgrade rudimentary single-step settlement pond systems. We characterise technologies used for land-based aquaculture wastewater treatment prior to discharge from shrimp systems in Australia. We identify opportunities to integrate technologies developed for the treatment of municipal wastewaters and intensive recirculating aquaculture systems, and use these to develop a model system for intensive shrimp farm wastewater. The first stage is the reduction of solids through the use of deep anaerobic ponds, which are tailored to dilute saline wastewater. Non-settled colloidal and supracolloidal solids can subsequently be removed through trapping in a sand bed filter and biological transformation to dissolved inorganic nitrogen or N_2. The resulting dissolved nutrients can be treated in a 3-stage algal treatment system by assimilation into harvestable biomass, and finally constructed wetlands polish wastewater through further trapping of particulates, and transformation of dissolved nitrogen. Given that upgrading wastewater treatment facilities is costly, we highlight options that have the potential to offset nutrient treatment costs, such as the use of algal biomass for food or energy products, and the recycling of nitrogen and phosphorus via pyrolysis creating products such as biochar and biofuel.

KEY WORDS: Settlement ponds · Crustaceans · Nutrients · Nitrogen · Bioremediation

INTRODUCTION

Land-based aquaculture is an integral part of global aquaculture production for fishes (28.8 million t), molluscs (13.1 million t) and crustaceans (5.0 million t; FAO 2010, Hall et al. 2011). The majority of land-based systems which support the intensive culture of fresh, brackish and marine water organisms do so through the addition of high-protein feeds to sustain the rapid growth of intensively farmed animals

*Email: s.castine@cgiar.org

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(Alam et al. 2009). High-intensity shrimp ponds for water column feeding shrimps, such as *Litopenaeus vannamei*, which are cultured under biofloc regimes, are an exception to this scenario, utilising suspended flocculated material and low-protein feeds (Browdy et al. 2001, Burford et al. 2003b, 2004). In the former more prevalent scenario, the assimilation of protein by culture animals is inefficient and can result in 50 to 80% of the nutrients being lost in wastewaters (Briggs & Funge-Smith 1994, Karakassis et al. 2005, Dalsgaard & Pedersen 2011). This wastewater is typically rich in both suspended solids (particulates) and dissolved nutrients, and if untreated can cause sediment loading and eutrophication in receiving waters (Preston et al. 2001, Burford et al. 2003a, Costanzo et al. 2004, Vaiphasa et al. 2007, Tello et al. 2010, Hall et al. 2011). In some receiving waters, rapid production within the water column facilitates assimilation of waste nutrients (McKinnon et al. 2002). Tidal flushing can also mitigate the impacts of the wastewater and effects on the receiving environment, for example in healthy mangrove creeks subjected to low levels of discharge (Trott & Alongi 2001, McKinnon et al. 2002). However, the potential impacts of wastewater released from intensive land-based aquaculture systems have prompted development of Best Management Practices (BMPs) and, in some cases, strict legislation for the treatment and release of waste streams from land-based aquaculture in the USA (Tucker & Hargreaves 2008), Australia (Donovan 2001, PIMC 2005) and the Mediterranean (IUCN 2009), all of which are largely based on treatment in settlement ponds. Policies and legislation provide the imperative to improve the design and management of marine and brackish settlement pond systems, with the goal to better manage the critical water quality variables of suspended solids and dissolved nutrients (Erler et al. 2004).

Large-scale municipal wastewater (MWW) treatment is an analogous wastewater system for which similar drivers lead to ongoing improvements to treatment processes and can provide significant insights into treating the large volumes of water that are characteristic of many land-based systems (Zakkour et al. 2002). More recently, intensive recirculating aquaculture systems (IRAS) have also adapted technology from freshwater MWW treatment systems to achieve effective low-cost water treatment in saline systems (Couturier et al. 2009). Both MWW and intensive recirculating aquaculture wastewater (IRAWW) treatment technologies provide an opportunity to redefine wastewater management for land-based aquaculture systems (Deviller et al. 2004, Gray 2005, Couturier et al. 2009, Dereli et al. 2010). They offer the opportunity to treat large volumes of wastewater to an acceptable standard. Specifically, these technologies must target wastewater with comparatively low total suspended solid (TSS) and dissolved nutrient concentrations.

Our aims are to firstly identify technologies that can be transferred from MWW or IRAWW treatment, and secondly, to synthesise these into a wastewater treatment model applicable to land-based aquaculture wastewater (LBAWW) treatment which also optimises opportunities for value-adding. The focus of this study is on marine (salinity 30–50‰) and brackish (0.5–30‰) water land-based aquaculture systems (LBAS) with emphasis on the shrimp industry which operates primarily in earthen ponds. However, in instances where technologies are applicable across a broad salinity range, transfer to and from freshwater (<0.5‰) systems is also highlighted. It enforces the concept of a multi-stage treatment process to concentrate and convert suspended and dissolved wastes into manageable biomass.

LBAS and settlement systems

Flow-through LBAS generally consist of large earthen ponds or raceways ranging in size from 1 or 2 ha to hundreds of ha. Exchanges of large water volumes have typically been used to control nutrient levels in grow-out ponds (Boyd & Tucker 1998), but as resource limitations increase, the use of and energy required to exchange large volumes of water is unsustainable, providing impetus to improve water productivity through wastewater treatment and reuse (Sharma et al. 2013). Furthermore, the exchange of large water volumes dilutes wastewater and makes the removal of solids difficult, especially in saline systems where settling of solids is reduced (Mesquita et al. 2011). Aquaculture wastewater requires specific treatment technologies as it is brackish or saline, comparatively dilute in suspended solids, and rich in dissolved organic nutrients.

LBAWW characteristics

Solid constituents

Understanding the unique characteristics of LBAWW is fundamental to treatment, in particular the particle (suspended solids) density and composition, and attributes of the 'sludge', i.e. the particulate matter which settles on the pond floor. Suspended solids are categorised into 2 groups: supracolloidal (1–100 µm) and settleable (>100 µm) particulates (Levine et al. 1985). While these categories are similar to those in municipal and intensive aquaculture waste streams, the key difference is that suspended solid concentration is low in LBAWW (5–119 mg l^{-1}) compared to IRAWW (5–390 mg l^{-1}) and MWW (93–800 mg l^{-1}; Table 1). This, and the large temporal and spatial variation in suspended solid concentration, impacts on treatment technologies for LBAS primarily because the low concentration of suspended solids results in slower settling, as particles make minimal contact with each other and do not form larger, heavier aggregates that naturally settle faster than individual particles (Fornshell 2001, Jackson et al. 2004).

Particles in LBAWW are primarily comprised of unutilised or degraded formulated feed, and excrement from culture species. However, heavy inorganic particulates (>100 µm) from earthen LBAS are eroded from the floor and banks of culture ponds and can make up a large portion (60–90%) of the settleable load in wastewaters from grow-out ponds (Preston et al. 2001). This particulate fraction is best divided into 2 groups based on size and mode of remediation. Settleable solids are >100 µm (Group 1; Fig. 1) and are remediated during pre-treatment, which is largely effective and has little scope for improvement. Supracolloidal particles (1–100 µm; Group 2; Fig. 1) are remediated during primary treatment which is more complex. Notably, particulates in LBAWW are typically 1.5 to 210 µm (Maillard et al. 2005), with the majority being <30 µm (Cripps 1995, Jones et al. 2002). Broadly speaking, the larger fraction of this size range (i.e. >100 µm) will settle in grow-out ponds (pre-treatment), while approximately 50% of the supracolloidal particles (1–100 µm) are settled in settlement ponds (primary treatment; Fig. 1).

Table 1. Examples of wastewater characteristics in municipal wastewater (MWW), intensive recirculating aquaculture waste-water (IRAWW) and land-based aquaculture wastewater (LBAWW). Values are in mg l^{-1}; means are given ±SD. TSS: total suspended solids

Characteristic	MWW	IRAWW	LBAWW
TSS	800[d]	5–50[b]	5–30[e]
	100–200[h]	390[i]	76.8 ± 7.8[j]
	93[m]		1.3[i]
	340[c]		0.4–2.5[l]
			40–119[g]
Total ammonia nitrogen	39[m]	6.8–25.6[a]	0.41[j]
	36[c]		
	100–800[a]		
Nitrate	<0.01[m]	11.7 ± 1.5[k]	0.19[j]
	2[c]		0.091[f]
Nitrite	<0.01[m]	0.06 ± 0.025[k]	0.23[j]
			0.004[f]
Total nitrogen	52[c]	18.00[i]	1.5–3[f]
			2.1–3.1[g]
Total phosphorus	10[c]	2.1[i]	0.02–0.09[l]
			0.22–0.28[g]

[a]Chen et al. (1993), [b]Cripps & Bergheim (2000), [c]Dereli et al. (2010), [d]Hammer & Hammer (2008), [e]Henderson & Bromage (1988), [f]Jackson et al. (2003), [g]Jackson et al. (2004), [h]Morari & Giardini (2009), [i]Piedrahita (2003), [j]S. Castine (unpubl.), [k]Summerfelt et al. (2009), [l]True et al. (2004), [m]Woertz et al. (2009)

In either case, the settled particulates form a semi-solid sludge layer that is relatively low in organic carbon (mean ± SD 0.85 ± 0.52% during early production cycle and 1.88% ± 0.46% during late production cycle) and total nitrogen (0.16 ± 0.09%; Burford et al. 1998). Additional constituents of sludge include phosphorus, potassium, calcium and magnesium, which respectively make up 4.3, 0.04, 11.3 and 0.16% of the dry mass (DM) of the sludge (Bergheim et al. 1998). Sludge also contains trace metals including copper, zinc, lead, cadmium, chrome, nickel and cobalt, of which the most quantitatively important are zinc (562–608 mg kg^{-1} DM), copper (24–29 mg kg^{-1} DM) and nickel (10–19 mg kg^{-1} DM; Bergheim et al. 1998). Sludge is viewed as a waste product that is removed from culture ponds at the end of each production cycle (Chen et al. 1993). However, sludge could become the basis of a product comprising highly volatile dry matter (19.8–29.7% of DM) and beneficial micro- and macro-nutrients (i.e. calcium 11.3% of DM; and zinc 562–608 mg kg^{-1} DM; Bergheim et al. 1998). In addition to the suspended solids, the wastewater is also rich in dissolved constituents that require further remediation prior to release (Jackson et al. 2003).

Dissolved constituents

As culture species respire and excrete waste, colloidal particles (0.45–1 µm) and dissolved nutrients (<0.45 µm), in particular ammonium (NH_4^+), are expelled into the water column (collectively Group 3; Fig. 1). There are a range of biological pathways by which these colloidal and dissolved waste compounds can be transformed to different compounds, for example, from an organic compound to an inorganic compound, or vice versa. These metabolic transformation pathways occur in bacteria, archaea,

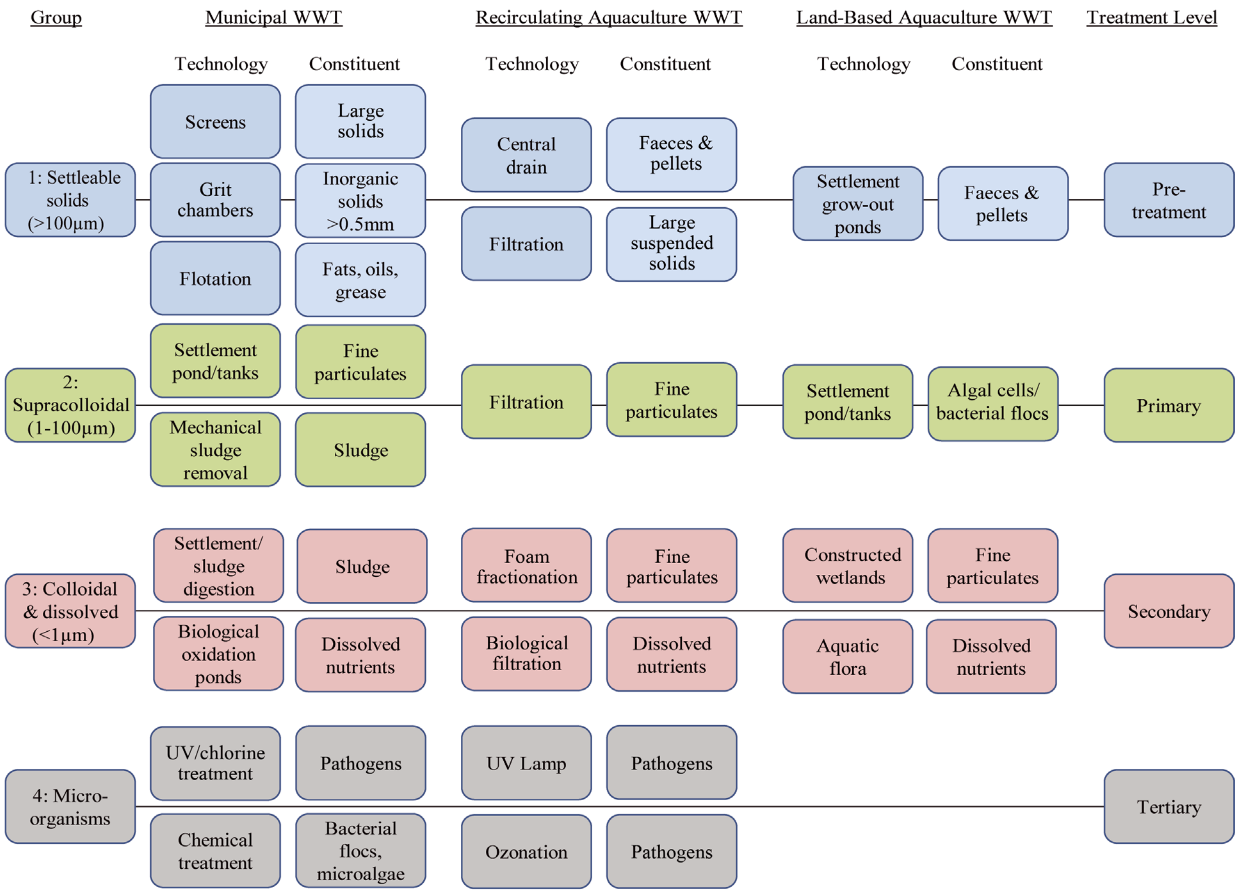

Fig. 1. Principal treatment steps in municipal, intensive recirculating and land-based aquaculture wastewater treatment (WWT) systems. Blue boxes represent pre-treatment options to treat Group 1 constituents; green boxes represent primary treatment options to treat Group 2 constituents; red boxes represent secondary treatment options to treat Group 3 constituents; and grey boxes represent secondary treatment options to treat Group 4 constituents. Based on Gray (2005) and Hammer & Hammer (2008)

microscopic fungi, algae, eukaryotes and viruses and are either dissimilatory (energy creating) or assimilatory (biomass creating) transformations (Burgin et al. 2011). There are both beneficial and detrimental dissimilatory pathways. The beneficial dissimilatory pathways which permanently remove nitrogen from aquaculture systems are nitrification, denitrification and anammox (Fig. 2). Nitrification transforms NH_4^+ to nitrate (NO_3^-) and is often coupled to denitrification in which denitrifiers reduce NO_3^- to nitrogen gas (N_2) (Knowles 1982) that is subsequently lost to the atmosphere (Fig. 2). Similarly, anammox results in the production of N_2, but it proceeds through the oxidation of NH_4^+ with NO_3^- (van de Graaf et al. 1995).

Conversely, the detrimental dissimilatory pathways of mineralisation, remineralisation and dissimilatory nitrate reduction to ammonium (DNRA) retain nitrogen within the system (Fig. 2). Mineralisation (the release of dissolved organic nitrogen, DON) and remineralisation (the release of NH_4^+) occur due to oxidation (degradation) of organic matter. These pathways typically occur in the sludge, fuelled by organic matter in the settled faeces, uneaten feed pellets, dead microalgae and microbial biomass (Burford & Williams 2001, Burford & Lorenzen 2004). Consequently, DON constitutes a significant proportion of the dissolved fraction (>60%) of the wastewater. However, many of the DON components, with the exception of urea, are not readily bio-available (Burford & Williams 2001; Fig. 2). NH_4^+ concentration is also typically high (means ± SD: 52.7 ± 7.4 to 61.2 ± 6.5 µM) in aquaculture wastewater (Bartoli et al. 2005), but nitrate (NO_3^-) concentration is typically much lower (3.6 ± 1.9 to 5.4 ± 1.8 µM; Bartoli et al. 2005).

Dissolved nitrogen compounds (NH_4^+ or NO_3^-) can also be transformed through assimilation and incorporation into microbial or algal biomass that can be removed from the system by harvest (de Paula Silva et al. 2012). Remediation of dissolved nitrogen is more effective if dissolved nitrogen exists in the inorganic form (NO_3^-, NO_2^- and NH_4^+) because these compounds are essential for the nitrification, denitri-

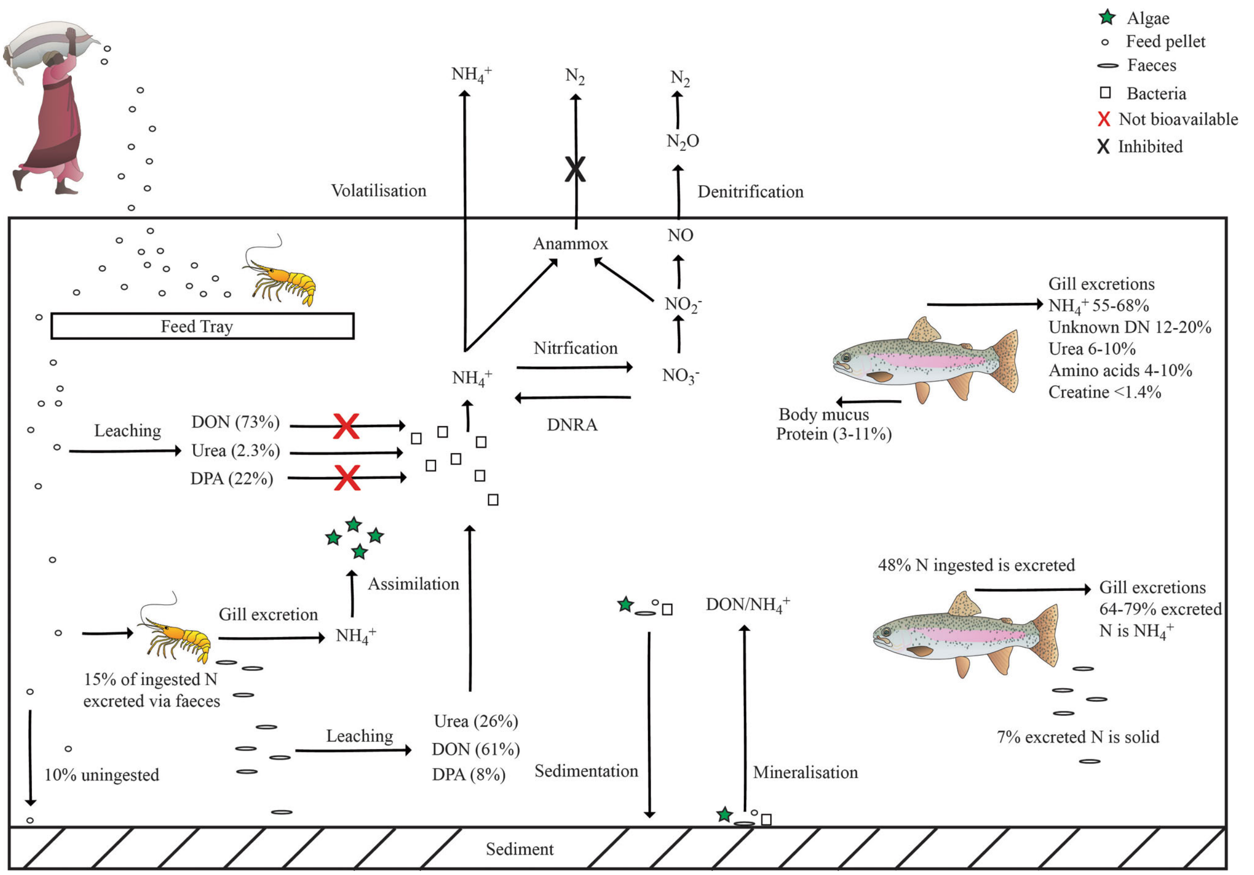

Fig. 2. Sources and fate of dissolved nitrogen in experimental shrimp and trout systems. Data are from Burford & Williams (2001), Kajimura et al. (2004) and Dalsgaard & Pedersen (2011). Graphics are from the Integration & Application Network (ian.umces.edu/symbols/). DON: dissolved organic nitrogen, DPA: dissolved primary amines, DNRA: dissimilatory nitrate reduction to ammonium

fication and anammox pathways occurring in the sludge and NO_3^- and NH_4^+ are preferentially assimilated by algae (Erler et al. 2010, de Paula Silva et al. 2012). Conversion from DON to dissolved inorganic nitrogen (DIN) is therefore beneficial to reducing total nitrogen (Erler et al. 2010).

Existing treatment technologies

The following description refers to flow-through systems used to culture benthic feeding shrimp and exclude low water-exchange biofloc systems suitable for shrimps which feed in the water column (Burford et al. 2003b).

Pre-treatment

The removal of settleable solids (>100 µm) is the first step in managing wastewater. Initially, particles settle efficiently because these particles (usually >100 µm; e.g. heavy faeces, particulates eroded from the pond floor and waste feed pellets) are accumulated in the centre of the grow-out pond using strategically placed aerators (Boyd 1998). This built-up sludge, estimated at 35 to 60 metric tonnes (t) ha^{-1} $harvest^{-1}$ within a single 1 ha shrimp grow-out pond (Preston et al. 2001), is removed at the end of each production cycle and is currently not further utilised.

Primary treatment

Settlement ponds (also known as settlement basins, bioremediation ponds, wastewater treatment ponds or waste stabilisation ponds) are large basins in which wastewater is retained on the principle that particulate wastes <100 µm will settle under gravitational forces (von Sperling & de Lemos Chernicharo 2005). Decreased flow and a long residence time facilitate gravitational settlement, and dissolved nutrients are concurrently transformed through the unmanaged processes of nitrification, denitrification, anammox or assimilation into biomass (Fig. 2). This simple approach is used by more than 70% of Australian land-based aquaculture farms (Preston et al. 2001). Newly constructed settlement ponds used by marine and brackish LBAS reduce TSS by 60%, total phosphorus load by 30% and total nitrogen load by 20%, despite being rudimentary single-step ponds of relatively small area (~0.1–0.8 ha) and shallow depth (0.25–2 m deep; Preston et al. 2001). Importantly, their efficacy decreases markedly over time if not maintained by removing settled solids which build up within the pond, forming a thick nutrient-rich sludge layer. These settled particulates decompose, releasing dissolved nutrients, hydrogen sulphide and methane into the water column (Burford et al. 1998, Burford & Lorenzen 2004). Removal of this sludge reduces mineralisation and remineralisation of nutrients and ensures that ponds remain sufficiently deep for particulates <100 µm to settle. LBAS currently operate on unsynchronized production cycles with heavy demand for multiple production cycles within the year, and therefore settlement ponds cannot be dried and sludge removed systematically. Consequently, pond design and management strategies need to be modified to include deep anaerobic pond systems for enhanced efficiency of solid transformation, and biological filtrations for the management of the resulting nutrients (DON and NH_4^+) as the critical steps in secondary treatment.

Secondary treatment

All settlement pond systems facilitate the biological transformation of nutrients, or secondary treatment, through microbial transformation and assimilation into biomass. However, because there is little control over the prevailing abiotic variables in these environments, it is difficult to enhance the beneficial microbial pathways of nitrification, denitrification and anammox. These pathways compete with detrimental pathways that recycle nitrogen in the system, such as DNRA and mineralisation, and the latter often prevails (Fig. 2), especially in tropical and subtropical sulphidic environments (Castine et al. 2012). However, biological transformation of nutrients by bacteria is an extremely powerful mechanism which has not yet been optimised for LBAS and represents significant opportunities for marine and brackish water LBAWW treatment.

Secondary treatment is also facilitated through the growth of micro- and macro-algae, as phototropic plant growth (assimilation) can rapidly remove nitrogen and phosphorus from wastewater (Neori et al. 2004). However, the efficacy of this approach relies on the optimisation of biomass production as nutrient remediation is proportional to algal growth. Consequently, it is necessary to actively manage algal standing stocks by removing biomass in proportion to growth to extract nutrients from the settlement pond system (de Paula Silva et al. 2008). As with the management and removal of suspended solids (sludge), the management and removal of algal biomass provides a resource of carbon, nitrogen, phosphorus, trace elements and minerals to deliver novel value-adding products.

MWW and IRAWW

There is an opportunity to improve the fundamental designs of LBAWW treatment by utilising technologies and strategies from pre-existing industries that have evolved beyond basic remediation principles. MWW and IRAWW treatment technologies are highly advanced due to the drivers of population pressure, water scarcity, environmental concerns and environmental regulation (for extensive reviews see Steicke et al. 2009, Terry & Krause 2009). However, the wastewater characteristics within each industry vary, with higher suspended solid and dissolved nutrient loads in MWW compared to both IRAWW and LBAWW, and higher suspended solid and dissolved nutrient loads in IRAWW compared to LBAWW (Table 1). Therefore, technology transfer needs to be optimised to target the unique characteristics of LBAWW. MWW treatment plants are the most evolved technology and involve 3 or 4 stages of treatment (pre-, primary, secondary and tertiary treatment stages; Fig. 1), and treat highly concentrated wastewater (93–800 mg l^{-1} TSS; Table 1) to a level that is safe for human consumption. Similarly, multi-stage treatment systems are used in IRAWW treatment plants (Fig. 1) with many technologies optimised to saline systems. Our objective is to consider technologies suitable for transfer to LBAWW treatment, given the characteristics of LBAWW and the capacity to engineer technologies at a scale relevant to LBAS (100s of ha).

LESSONS AND TECHNOLOGIES FROM MWW AND IRAWW TREATMENT

There are both obvious and subtle improvements for LBWW systems using treatment technologies from MWW and IRAWW. These technology transfers primarily need to address the concentration of supracolloidal and colloidal (<100 µm) particulates, the removal and treatment of concentrated particulates and the removal of residual dissolved nutrients in 'treated' wastewater. These technologies are synthesised in a proposed model treatment system for LBAWW.

Concentrating, settling and removing particulates

The relatively low concentration of particulate constituents in aquaculture wastes (0.4–119 mg l^{-1} in LBAWW compared to 93–800 mg l^{-1} in MWW; Table 1) necessitates the concentration of particulates to enhance removal or settlement. Removal and settlement can be enhanced using physical processes, a combination of physical and biological processes, or multi-stage pond systems.

Physical processes utilise screens and barriers to capture or settle particulates. Tube settlers enhance settling by forcing water to flow up through a settling plate (angled at 45 to 60° above horizontal to facilitate self-cleaning), capturing solids on the underside of the plate (Timmons & Ebeling 2007). They are used to settle supercolloidal (1–100 µm; 80% removal efficiency) and colloidal (0.45–1 µm; 55% removal efficiency) particulates in IRAWW (Easter 1992). Tube settlers may be a better option than enhancing settlement through natural or chemical flocculation techniques, although the economic viability of tube settlers in LBAS should be tested. During natural flocculation, filamentous bacteria enhance the structure of the floc and protozoa grazing on non-settleable bacteria enhance the size of the floc, making it heavier and more likely to settle. However, mortality of filamentous bacteria and protozoa may increase in the presence of salt, decreasing the natural settling capability of particles in marine and brackish water systems (Mesquita et al. 2011). There is little evidence that chemical flocculants and coagulants (lime, iron sulphate, iron chloride, aluminium sulphate and aluminium chloride) are efficient in saline water and these have proven prohibitively expensive as treatment in freshwater systems (Cripps & Bergheim 2000, Parsons & Smith 2008). Instead, rotating micro-screens, drum screens, drum filters or swirl concentrators with screen sizes ranging from 60 to 200 µm have been successfully implemented in land-based intensive fish farms (Cripps & Bergheim 2000, Sindilariu et al. 2009), and 60 µm mesh has the potential to capture >80% of solids in freshwater fish farms (Kelly et al. 1997). The selection of physical filters is dependent on individual system requirements. For example, in a comparison of drum filters and swirl separators at a recirculating salmon-smolt farm, swirl separators removed 63% of TSS compared to 22% by drum filtration (Couturier et al. 2009). Granular and porous media filters provide an alternative to screen filters in IRAWW treatment and have the advantage of acting as both physical and biological filters, by trapping and transforming particles, respectively (Chen et al. 1993, Cripps & Bergheim 2000).

Examples of settlement techniques can also be drawn from the agricultural industry where multi-stage baffled settlement ponds are used to control flow rates and depth regimes to enhance settlement and biological treatment for settleable (>100 µm), supracolloidal (1–100 µm) and colloidal (<1 µm) particles in MWW treatment and for intensive agribusiness (dairy, piggeries, feed-lot cattle). Initially, physical processes such as grit screens (>30 mm) are also employed to remove wood, rags, grit and coarse solids. After the coarse solids are removed, a deep anaerobic pond facilitates settlement of particles <100 µm and sludge removal (Archer & Mara 2003, Craggs et al. 2004a, 2008). Anaerobic ponds have almost been universally adopted across industries (municipal, agriculture and aquaculture), and their design is recommended to be 4 to 5 m deep and twice as long as they are wide (DEC 1996). Modifications to this design include a set of 2 deep anaerobic ponds, providing the flexibility to dry and remove sludge from one pond while continuing treatment in the adjacent pond (Fig. 3). However, LBAWW treatment systems do not use initial physical processes (grit screening) or designated anaerobic ponds, and the settlement ponds rarely contain baffles to direct water flow. This means that water in land-based aquaculture settlement ponds commonly short circuits, where water takes the most direct path to the outlet, resulting in low residence times irrespective of their expansive layout. This affects both particulate and dissolved nutrient remediation by reducing the residence time of wastewater available for transformation and assimilation. Retrofitting baffles using earth or high density polyethylene, as is done in the dairy industry (Craggs et al. 2004a), may provide a simple means of increasing residence time and improving the settlement and consistency of LBAWW treatment (Fig. 3).

In MWW treatment, settled particles form a semi-solid sludge, which necessitates removal prior to the dissimilation of captured nutrients into DON (mineralisation) and NH_4^+ (remineralisation). The opportunity to concentrate and settle particles before they contribute to nutrient release through mineralisation is potentially more cost-effective than large-scale dissolved nutrient remediation, though this has not been specifically quantified. Improved solids capture will certainly reduce downstream particulate nitrogen and phosphorus concentrations, but dissolved nutrients will remain a significant waste stream in LBAWW treatment.

Removing dissolved nutrients

Reducing dissolved nutrient concentration in any type of wastewater using rudimentary single-step settlement pond technology presents a significant challenge because ponds are not optimised for the assimilation of nutrients through algal growth and harvest, or for transformation by beneficial microbial pathways (Craggs et al. 1996, Bolan et al. 2009). However, 3 types of processes can be transferred from MWW and IRAWW treatment to improve the removal of dissolved nutrients from LBAWW. The first is to use microbial processes that are enhanced through biological filters, reactors or digesters with examples from temperate IRAS and MWW treatment systems (Summerfelt 2006, Sharrer et al. 2007, Chen et al. 2010, Roy et al. 2010). The second is to use phototrophs (algae, cyanobacteria) to enhance assimilation, as has been demonstrated in commercial land-based fish farms and in laboratory studies with shrimps (Bartoli et al. 2005, Henry-Silva & Camargo 2006, Mai et al. 2010). Finally, the third is to combine these processes into a functional bioremediation mesocosm, comprised of multi-stage systems (Craggs et al. 2004a) that may include a constructed mangrove wetland as an end point (Lin et al. 2003).

Enhanced microbial processes in fluidised sand-bed filters can promote nitrification, removing up to 90% of the NH_4^+ in a rainbow trout IRAS (Heinen et al. 1996). As NH_4^+ is also a problematic nutrient in LBAS culturing shrimps, the use of such filters would be beneficial to promote nitrification and could be readily adapted to the large volumes of LBAS. Membrane biological reactors are also efficient in promoting microbial processes, as demonstrated in a temperate seabream system (Tal et al. 2009), and NH_4^+ removal is efficient over a range of salinities, from 0 to 32‰ in temperate IRAS (Sharrer et al. 2007). However, at higher salinity the startup time for a nitrifying reactor is extended because the nitrifying microbial community takes longer (~118 d) to acclimate and become effective (Sharrer et al. 2007).

Enhanced phototrophic processes have been a focus for the improvement of remediation in the dairy industry. Craggs et al. (2004a) overcame the fundamental issue of high dissolved nutrient content in dairy farm wastewater by upgrading a traditional 2-stage treatment pond system to a 4-stage pond system. Two-pond freshwater systems (primary and secondary treatment) consisting of a deep anaerobic settling basin and a shallower facultative pond, were traditionally recommended as they were effective in remediating biochemical oxygen demand, carbon and TSS in dairy farm wastewater (Bolan et al. 2009). However, they were not optimised for dissolved nutrient removal, and the upgrade to a multi-stage pond system increased NH_4^+ removal by 37% and TSS removal by 44% (see Fig. 2 in Craggs et al. 2004a). Treatment of agricultural wastewater and MWW now includes a series of ponds with different physical characteristics, each performing a separate function in the wastewater remediation process (Craggs et al. 2004a), and widening the opportunity for energy production and capture (Craggs et al. 2011).

Agricultural wastewater treatment comprises 4 stages, the first of which is a single anaerobic pond to enhance settlement of particulates. The second is a high rate microalgal pond with a large surface area and shallow profile (0.2–1 m deep) to increase exposure of algal cells to light for enhanced biomass production (Park et al. 2011). The third is a pair of algal settling ponds that are deeper near the inflow and become shallower, terminating in a surface outflow pipe at the discharge to ensure that solids are separated and recovered (Craggs et al. 2004a). The final stage is a maturation pond, used for disinfection through UV radiation and removal of remaining microalgal cells through protozoan grazing (Craggs et al. 2004b). Such complexity of pond systems has not yet been adopted by marine and brackish water LBAS, due in part to the less concentrated nature of aquaculture effluents compared to that of agricultural and sewage effluent. However, it provides a working model on which to build an upgraded system for LBAWW treatment that is re-designed to meet the composition of saline dilute wastewater.

Primary producers play a major role in a multi-stage pond system (e.g. the high rate microalgal pond in the second stage) and are employed in marine land-based and open water aquaculture systems (Chopin et al. 2001, Troell 2009), and to a lesser extent in freshwater aquaculture systems (Hasan & Charkrabarti 2009). Integration of commercially valuable microalgae into the second stage of a multi-

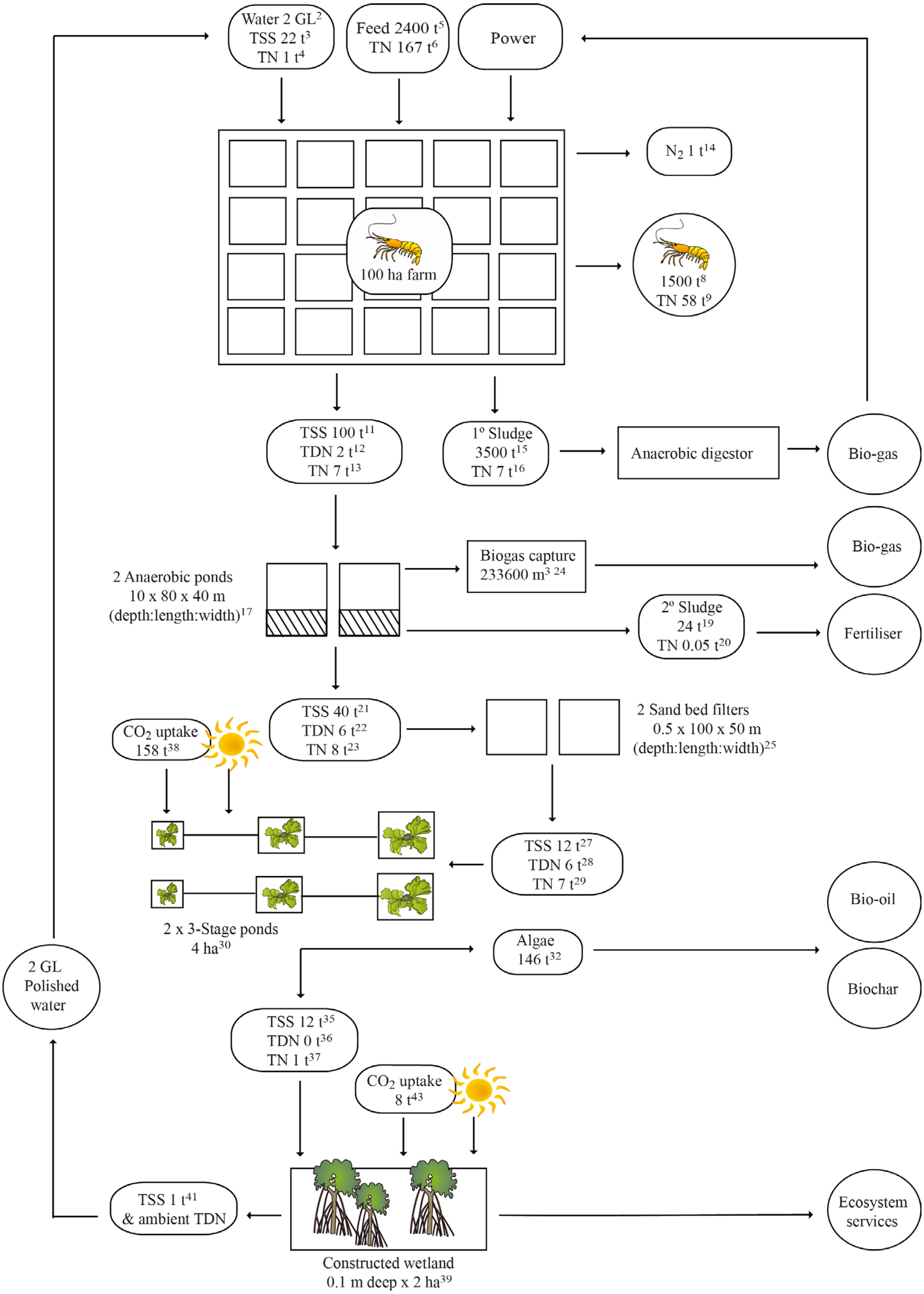

Fig. 3. Conceptual model of recommended treatment for land-based aquaculture wastewater. Loads are based on a 100 ha shrimp farm over 1 yr and the assumptions and calculations for this model are given in Table S1 in the Supplement at www.int-res.com/articles/suppl/q004p285_supp.pdf. Superscripts denote the ID number for cross reference to Table S1. Graphics are from the Integration and Application Network (ian.umces.edu/symbols/). TSS: total suspended solids, TN: total nitrogen, TDN: total dissolved nitrogen, GL: gigalitre

stage pond system has many economic advantages over stand-alone algal ventures, including lower capital, water, harvesting, operational and maintenance costs (see Table 1 in Park et al. 2011). Similarly, macro-flora can be integrated to manage dissolved nutrients. Mixed macro-algal stands grown in freshwater dairy wastewater treatment ponds or steam mesocosms assimilate large quantities of dissolved nitrogen (70 mg g^{-1} DM m^{-2} d^{-1}) and phosphorus (13 mg g^{-1} DM m^{-2} d^{-1}) under experimental conditions (Craggs et al. 1996, Kebede-Westhead et al. 2003). Other models demonstrate nitrogen removal efficiencies by filamentous green tide algae of 3.3 kg N ha^{-1} d^{-1} in a commercial flow-through land-based barramundi *Lates calcarifer* wastewater treatment pond (de Paula Silva et al. 2008). These algae have high tolerance for environmental variation and correspondingly high annual growth rates and nitrogen assimilation ability, although the commercial potential of these algae has not been demonstrated (de Paula Silva et al. 2008). Commercial seaweeds, such as *Gracilaria* spp., have also been used for small-scale mariculture as both a waste mitigation tool and for value-adding in China, Vietnam, Indonesia, India and the Philippines (for a comprehensive review see Troell 2009). For example, *G. birdiae* was used as a biofilter for shrimp pond effluent and reduced nutrients by 34, 93.5 and 99.3% for NH_4^+, phosphate (PO_4^{3-}) and NO_3^-, respectively (Marinho-Soriano et al. 2009). Similarly, water hyacinth promoted 50% reductions in nitrogen and phosphorus from freshwater MWW (Abbasi & Abbasi 2010). However, the selection of high value species for tropical marine and brackish water LBAS requires careful consideration of the fluctuating environmental conditions which have potential to impact algal growth and survival and limit reliable treatment (Paul & de Nys 2008).

As a final polishing step for residual nutrients, the use of mangrove constructed wetlands combines physical, microbial and phototrophic processes to treat municipal, aquaculture and agriculture wastewater over a broad range of salinities (Erler et al. 2008, Dong & Reddy 2010, Kadlec et al. 2010, Yeh et al. 2010). They achieve nutrient abatement by physically trapping and burying particulates, microbial assimilation and transformation of nutrients, phototrophic assimilation into plant biomass as well as volatilization and sorption (Erler et al. 2008, 2010, Qiu et al. 2011). Removal efficiencies of up to 98% for DIN occur in the treatment of shrimp farm wastewater using constructed wetlands (Lin et al. 2002). A subsequent study demonstrated reductions of biochemical oxygen demand (24%), TSS (71%), chlorophyll *a* (88%), NH_4^+ (57%), nitrite (NO_2^-; 90%) and NO_3^- (68%) of shrimp farm wastewater treatment in constructed wetlands (Lin et al. 2003). Furthermore, denitrification and anammox rates of (means ± SE) 199.4 ± 18.7 and 965.3 ± 122.8 µmol N m^{-2} h^{-1}, respectively, have been recorded in freshwater wetlands treating MWW (Erler et al. 2008). The physical design of wetlands and application of wastes should be optimised to enhance oxidation of DON to DIN and improve nitrogen removal (Erler et al. 2010)

LBAWW treatment — a model

We propose a multi-stage treatment system utilising the 'best of' technologies to increase the removal of settled sludge and dissolved nutrients from land-based aquaculture (Fig. 3). The proposed system is modelled on a 100 ha farm which discharges 2000 Ml of wastewater per annum containing a mean TSS load of 50 mg l^{-1}. The assumption of this model is that there is little fluctuation in the mean concentration of TSS. In reality, TSS load varies both spatially and temporarily (Henderson & Bromage 1988, Jackson et al. 2003, 2004), therefore scaling of the model requires testing of the likely failure rates of each step of the system under varying effluent composition. The model would begin with a set of deep anaerobic ponds with a required treatment area of 10 × 80 × 40 m (depth × length × width; see Table S1 in the Supplement at www.int-res.com/articles/suppl/q004p285_supp.pdf). Notably, the length to width ratio should be between 2:1 and 3:1 (Mara & Pearson 1998, Alexiou & Mara 2003), with the longest side perpendicular to prevailing wind to reduce wind-driven turbulence (Craggs et al. 2004a). The hydraulic retention time would be 10 d and would reduce TSS load by up to 60% (Table S1). Water would flow into the pond at depth, via an inlet pipe at the bottom of the pond, and subsequently filter up through coarse media (e.g. gravel), which would aid in the capture of particulates of all size fractions. This step would reduce the reliance on settlement of particulates in anaerobic ponds where the inlet was above the height of the bottom of the pond. A pair of anaerobic ponds would allow both to be used simultaneously or individually, so that one pond could be dried and de-sludged while continuing treatment in the other. A de-sludging interval of approximately 3 to 4 yr is recommended (Craggs et al. 2004a). Wastewater would subsequently pass through a pair of large-scale (0.5 × 100 × 50 m; depth × length × width) sand bed filters

(Table S1). Flow in the sand filters would be optimised so that wastewater and its associated nutrients are in contact with biofilm for a sufficient length of time to ensure that particulate organic matter and DON are oxidised to inorganic nitrogen species. Resulting DIN would then be transformed in the next stage of treatment through assimilation into algal biomass (Fig. 3). Two parallel, aerated algal treatment areas, each of 2 ha (Table S1), would be tailored in depth relative to the light conditions of their location, but typically less than 0.5 m. Each 2 ha treatment area would be divided into 3 smaller units; the first pond of 10 000 m^2, the second pond of 6500 m^2 and the third pond of 3500 m^2. A flow rate of 4 m^3 h^{-1} into the first treatment stage is recommended and flow rates through the smaller units would be faster, therefore increasing DIN flux despite the fact that DIN would be less concentrated by the time it reaches the smaller units (due to assimilation by algae in the first treatment unit; Neori et al. 2003). This improves the biofilter performance of the algae (Neori et al. 2003) and would result in an annual biomass yield of 146 t (DM) based on a conservative productivity of 37 t ha^{-1} yr^{-1} (or 10 g DM m^{-2} d^{-1}; Table S1). The final stage of treatment would comprise a mangrove wetland (0.1 m deep and 2 ha) which would facilitate capture (through trapping) of persistent colloidal particles and would enhance beneficial microbial nutrient transformation (through sediment aeration) (Fig. 3). Although sediment accumulation and associated carbon and nitrogen trapping are effective almost immediately after wetland construction, heterotrophic activity and primary production take between 5 and 15 yr to reach rates equivalent to natural marshes (Craft et al. 2003). Once fully functional wetlands have the potential to produce polished wastewater containing only ambient total dissolved nitrogen and TSS and could be recycled back to the culture ponds as required. This model predicts overall removal of nitrogen by 99 to 100% but requires simulation with farm-specific and temporal parameters. In addition, a cost–benefit model is required to ensure that construction, operation and maintenance costs are within the economic framework of the aquaculture operation.

Off-setting the cost of compliance

Environmental compliance incurs labour and infrastructure costs which can be off-set to ensure that farms maximise profits. There are many measures to provide incentive for waste mitigation, such as internalizing costs, charging for ecosystem services, taxing carbon emissions or maximising recapture, reuse and recycling of valuable nutrients and biomass that are currently released. In the latter example, when the secondary product has commercial value, each of the extractive steps in best practice compliance can be considered as an income stream to maximise the utilisation of input costs which are lost in traditional management regimes (Bolton et al. 2009). Specifically, in sequential order, these are sludge, digestion energy products from settled fine solids and algal biomass. Relative to freshwater systems, reusing and recycling nutrients from marine and brackish water systems does have some limitations which will be described throughout this section.

Sludge is the first output from the waste cascade as it is collected in the primary stage of production. It is produced in large volumes (35–60 t ha^{-1} $harvest^{-1}$ within a single 1 ha shrimp grow-out pond) and potentially constitutes a valuable by-product for agriculture as it contains a range of micro-nutrients (Ca, K, Mg, Cd, Cu, Mn) and macro-nutrients (nitrogen and phosphorus) as well as organic carbon (Rosenani et al. 2004). These nutrients and trace elements can be used if sludge is processed to eradicate pathogens, provided nutrients remain bioavailable (Hossain et al. 2011). One approach to delivering nutrients and trace elements as a defined agri-fertiliser is through production of biochar. Biochar is produced through the slow pyrolysis of biomass, and feedstocks similar to sludge have been successfully converted to biochar. The pyrolysis of sewage sludge produces a biochar (Lehmann & Joseph 2009, Hossain et al. 2011) and soils amended with sewage sludge biochar have enhanced nutrient availability and electrical conductivity, resulting in an increase in horticulture crops (Hossain et al. 2010). Pyrolysis of aquaculture sludge and the use of the resulting biochar is a prospective technology, particularly in marine and brackish water systems, and the potential for biochar to off-set the cost of environmental compliance depends largely on the concentration and nature of sodium cations in the resulting biochar. Both Bird et al. (2011) and Grierson et al. (2011) noted that the sodic nature of biochar produced from marine algae (and presumably from saline sludge) may compromise the use of the product as a soil amendment unless it is applied in low concentrations to well-drained soil. In addition, feedstocks such as aquaculture sludge which are low in carbon but high in nutrients and trace elements could be enhanced through co-firing with carbon-rich lignocellulosic feedstocks.

Alternatively, suspended solids either from the culture or settlement ponds can be used for energy conversion through anaerobic digestion and biogas capture. Anaerobic digesters are used extensively in freshwater MWW treatment (Dereli et al. 2010) and are well established in the treatment of agriculture and household waste solids (Lanari & Franci 1998, Krzystek et al. 2001, Craggs et al. 2008, Raposo et al. 2012). Anaerobic digesters have been trialled in IRAWW treatment (Lanari & Franci 1998) and convert >60% of the solids from IRAWW to methane under a variety of salinity regimes, reducing outsourced energy requirements by a small but valuable 2 to 5% (Gebauer & Eikebrokk 2006, Mirzoyan et al. 2010). An experimental anaerobic digester, treating 2.8 l of trout farm faecal solids every 4 h, demonstrated high biogas yields with over 80% methane at a rate of 144 l d^{-1} (Lanari & Franci 1998). Similarly, Tal et al. (2009) demonstrated 60% conversion of solids to methane under high salinity in a pilot scale re-circulating gilthead seabream system, demonstrating the potential of biogas capture in saline systems. This broader 'industrial ecology' concept represents an innovation by industry in solid waste management based on an understanding of synergistic opportunities in the value-chain (Korhonen 2005). This integrated design has become a new industrial paradigm, led by China, where 50 eco-industrial parks, including aquaculture systems, are being constructed or have been approved for construction resulting in economic benefits and waste mitigation (Mathews & Tan 2011). Similar innovation can be expected once the co-products of intensive land-based aquaculture production are better evaluated (Castine et al. 2013).

Suspended solids can be transformed to provide substrates (dissolved nutrients) for new products through assimilation. The degradation of solids and conversion of nutrients by infauna (i.e. polychaetes and the associated microbial community) provides a biological technique by which complex organic molecules can be converted to simple DIN compounds (Kunihiro et al. 2008, Wada et al. 2008, Palmer 2010), facilitating further treatment with bacterial or algal communities. Palmer (2010) used brackish water sand beds stocked with polychaete worms to reduce suspended solid loads by >50% and to produce polychaete biomass at 300 to 400 g m^{-2} as an aquaculture feed. This process released and provided dissolved nutrients for algal remediation.

Nutrient sequestration into algae (micro and macro) is a proven approach to deliver an income driver from the remediation of dissolved nutrients through production of bio-products and energy (Chopin & Sawhney 2009, Park et al. 2011). Micro-algae, particularly marine micro-algae, contain high concentrations of fatty acids and are the target of a wide range of stand-alone aquaculture systems for nutraceuticals and biofuels (Brennan & Owende 2010, Huerlimann et al. 2010, Mata et al. 2010, Vílchez et al. 2011). Similarly, macro-algae have already generated net positive cash flow in commercial, integrated aquaculture systems (Bolton et al. 2009, Bunting & Shpigel 2009, Nobre et al. 2010). Algae offer economic returns through direct sales of commercial species (Chopin et al. 1999) in addition to reduction of feeding costs for herbivores and reduced pumping costs through recirculation and improvement of wastewater treatment capacity (Bolton et al. 2009, Nobre et al. 2010). More recently, there has been a renewed focus on bio-products from novel, resilient species of both micro- (Jung & Lovitt 2010) and macro-algae (de Paula Silva et al. 2008), demonstrating the flexibility to deliver products developed from site-specific bioremediation. For example, the production of algal biochar from green tide algae sequesters carbon and waste nutrients from aquaculture systems and results in a high-value biochar for use in soil amendments (Bird et al. 2011, 2012). Pyrolysis of microalgal biomass for the production of both bio-oil and biochar provides beneficial fatty acids in the resulting oil and high cation exchange capacity, nitrogen concentration and low C:N ratio in the char (Grierson et al. 2011, Castine et al. 2013).

The pyrolysis process also produces bioenergy (Abdullah et al. 2010), and thermal conversion of macro-algal biomass has the potential to deliver biofuels from species with high biomass productivities (Ross et al. 2008, Zhou et al. 2010). Given that dense micro- and macro-algal communities occur naturally in LBAS, and there is broad scope to enhance algal production to sequester waste nutrients (including the use of CO_2), these underutilised and easily implemented resources could provide an important driver for improvement of LBAWW treatment.

CONCLUSIONS

Solid and dissolved constituents are the 2 main waste sources which must be managed in LBAS. The current treatment of these waste sources in unmanaged settlement ponds is not optimised for efficient nutrient removal or reuse, and we describe 'best of' technologies that are tried and tested in other industries. These technologies offer off-the-shelf solutions to meet environmental compliance and enhance sustainability. By using multi-stage treatment plants with anaerobic digesters, sand filters and constructed wetlands, the difficulties associated with settling fine particles in dilute saline wastewater, and the complexities of enhancing beneficial microbial pathways for remediation of dissolved constituents, can be circumvented. Integration of algal and macrophyte cultures can also be optimised to increase wastewater treatment efficiency and profitability of the farms, and be tailored to local flora and regional requirements for specific end-products to engage with synergistic industrial ecology. Our conceptual model includes specific design parameters that form the basis for environmental compliance, enable the intensification of production through increased treatment efficiencies, and reduce water usage for LBAS. The potential for off-setting the costs of upgrading treatment systems through a suite of secondary products at each extractive stage, including bio-energy and agricultural applications, should be investigated further and optimised to the specific requirements of each farm.

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