Chapter 10
Aerobic and Anaerobic Treatments for Aquaponic Sludge Reduction and Mineralisation

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Abstract Recirculating aquaculture systems, as part of aquaponic units, are effective in producing aquatic animals with a minimal water consumption through effective treatment stages. Nevertheless, the concentrated sludge produced after the solid filtration stage, comprising organic matter and valuable nutrients, is most often discarded. One of the latest developments in aquaponic technology aims to reduce this potential negative environmental impact and to increase the nutrient recycling by treating the sludge on-site. For this purpose, microbial aerobic and anaerobic treatments, dealt with either individually or in a combined approach, provide very promising opportunities to simultaneously reduce the organic waste as well as to recover valuable nutrients such as phosphorus. Anaerobic sludge treatments additionally offer the possibility of energy production since a by-product of this process is biogas, i.e. mainly methane. By applying these additional treatment steps in aquaponic units, the water and nutrient recycling efficiency is improved and the dependency on external fertiliser can be reduced, thereby enhancing the sustainability of the system in terms of resource utilisation. Overall, this can pave the way for the economic improvement of aquaponic systems because costs for waste disposal and fertiliser acquisition are decreased.

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10.1   Introduction

The concept of aquaponics is associated with being a sustainable production system, as it re-utilises recirculating aquaculture system (RAS) wastewater enriched in macronutrients (i.e. nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg) and sulphur (S)) and micronutrients (i.e. iron (Fe), manganese (Mn), zinc (Zn), copper (Cu), boron (B) and molybdenum (Mo)) to fertilise the plants (Graber and Junge 2009; Licamele 2009; Nichols and Savidov 2012; Turcios and Papenbrock 2014). A much debated question is whether this concept can match its own ambition of being a quasi-closed-loop system, as high amounts of the nutrients that enter the system are wasted by discharging the nutrient-rich fish sludge (Endut et al. 2010; Naylor et al. 1999; Neto and Ostrensky 2013). Indeed, to maintain a good water quality in a RAS and aquaponic systems, the water has to be constantly filtrated for solid removal. The two main techniques for solid filtration are to retain the particles in a mesh (i.e. mesh filtration as drum filters) and to allow the particles to decant in clarifiers. In most conventional plants, sludge is recovered out of these mechanical filtration devices and is discharged as sewage. In the best cases, the sludge is dried and applied as fertiliser on land fields (Brod et al. 2017). Notably, up to 50% (in dry matter) of the feed ingested is excreted as solids by fish (Chen et al. 1997), and most of the nutrients that enter aquaponic systems via fish feed accumulate in these solids and so in the sludge (Neto and Ostrensky 2013; Schneider et al. 2005). Hence, effective solid filtration removes, for example, more than 80% of the valuable P (Monsees et al. 2017) that could otherwise be used for plant production. Therefore, recycling these valuable nutrients for aquaponic applications is of major importance. Developing an appropriate sludge treatment able to mineralise the nutrients contained in sludge for re-using them in the hydroponic unit seems to be a necessary process for contributing to close the nutrient loop to a higher degree and thus lowering the environmental impact of aquaponic systems (Goddek et al. 2015; Goddek and Keesman 2018; Goddek and Körner 2019).

It has been shown in experimental studies that complemented aquaponic nutrient solution (i.e. after addition of lacking nutrients) promotes plant growth compared to hydroponics (Delaide et al. 2016; Ru et al. 2017; Saha et al. 2016). Hence, sludge mineralisation is also a promising way to improve the aquaponic system performance as the nutrients recovered are used to complement the aquaponic solution. In addition, on-site mineralisation units can also increase the self-sufficiency of aquaponic facilities, especially with respect to finite resources as P which is essential for plant growth. P is produced by mining activities, whereby the deposits are not equally distributed around the world. In addition, its price has risen by up to 800% within the last decade (McGill 2012). Thus, mineralisation units applied in aquaponic systems are also likely to increase its future economic success and stability.

Sludge treatment in aquaponics needs to be approached differently than has been done in the past. Indeed, in conventional wastewater treatment, the main objective is to obtain a decontaminated and clean effluent. The treatment performances are expressed in terms of removal of contaminants (e.g. solids, nitrogen (N), phosphorus (P), etc.) out of the wastewater and by quantifying the effluents with respect to the achieved quality (Techobanoglous et al. 2014). Using this conventional approach, several studies have provided quantitative evidence that a consistent proportion of chemical oxygen demand (COD) and total suspended solids (TSS) can be removed by digesting the RAS wastewater under aerobic, anaerobic and sequential aerobic–anaerobic conditions (Goddek et al. 2018; Chowdhury et al. 2010; Mirzoyan et al. 2010; Van Rijn 2013). However, in aquaponic systems, the wastewater from fish is considered to be a valuable fertiliser source. Within a closed-loop approach, the solid part discharged needs to be minimised (i.e. organic reduction maximised), and the nutrient content in the effluents needs to be maximised (i.e. nutrient mineralisation maximised). Therefore, the wastewater treatment performance in aquaponics no longer needs to be expressed in terms of contaminants removal but in terms of its contaminant reduction and nutrient mineralisation ability.

A few studies have demonstrated the functional capability of digesting fish sludge with aerobic and anaerobic treatments for organic reduction purposes (Goddek et al. 2018; van Rijn et al. 1995). With anaerobic treatment in bioreactor, high dry matter (i.e. TSS) reduction performance (e.g. higher than 90%) can be achieved while methane can also be produced (van Lier et al. 2008; Mirzoyan and Gross 2013; Yogev et al. 2016).

The aerobic treatment of sludge is also a very effective way to reduce organic matter, which is oxidised to CO_2 during respiration (see Eq. 10.1). For example, reduction rates of 90% (here determined as suspended solids, COD and BOD reduction) were reported from a water resource recovery facility (Seo et al. 2017). Aerobic processes are faster than anaerobic, but they can be more expensive (Chen et al. 1997) as a constant aeration of the sludge–water mixture requires energy-intensive pumps or motors. Moreover, significant fractions of the nutrients are converted to microbial biomass and do not stay dissolved in the water.

Although these studies have shown the organic reduction potential of fish sludge, only a few authors have examined the release of specific nutrients (e.g. for N and P) from fish sludge. Most of these studies were for short in vitro batch experiments (Conroy and Couturier 2010; Monsees et al. 2017; Stewart et al. 2006) and from an operating RAS (Yogev et al. 2016), rather than an aquaponic setup. While discussed to some extent in theory (Goddek et al. 2016; Yogev et al. 2017), the research has to start now to systematically investigate the organic reduction and nutrient mineralisation performance of fish sludge for both aerobic and anaerobic reactors and its effect on the water composition and plant growth. Therefore, this chapter aims to give an overview on the diverse fish sludge treatments that can be integrated into aquaponic setups to achieve organic reduction and nutrient mineralisation. Some design approaches will be highlighted. The nutrient mass balance approach in the context of aquaponic sludge treatment will be discussed, and specific methodology to quantify the sludge treatment performance will be developed.

10.2   Wastewater Treatment Implementation in Aquaponics

In aquaponics, the wastewater charged with solids (i.e. the sludge) is a valuable source of nutrients, and appropriate treatments need to be carried. The treatment goals differ from conventional wastewater treatment because in aquaponics solids and water conservation is of interest. Moreover, regardless of the wastewater treatment applied, its aim should be to reduce solids and at the same time mineralise its nutrients. In other words, the aim is to obtain a solid-free effluent but rich in solubilised nutrients (i.e. anions and cations) that can be reinserted into the water loop in a coupled setup (Fig. 10.1a) or directly into the hydroponic grow beds in a decoupled setup (Fig. 10.1b). Fish sludge solids are mainly composed of degradable organic matter so that the solid reduction can be called organic reduction. Indeed, the complex organic molecules (e.g. proteins, lipids, carbohydrates, etc.) are principally composed of carbon and will be successively reduced to lower molecular weight compounds until the ultimate gaseous forms of CO_2 and CH_4 (in the case of anaerobic fermentation). During this degradation process, the macronutrients (i.e. N, P, K, Ca, Mg and S) and micronutrients (i.e. Fe, Mn, Zn, Cu, B and Mo) that were bound to the organic molecules are released into the water in their ionic forms. This phenomenon is called nutrient leaching or nutrient mineralisation. It can be assumed that when high organic reduction is achieved, high nutrient mineralisation would also be achieved. On the one hand, sludge contains a proportion of undissolved minerals, and on the other hand, some macro- and micronutrients are released during the mineralisation process. These can quickly precipitate together and form insoluble minerals. The state between ions and precipitated minerals of most of the macro- and micronutrients is pH dependent. The most well-known minerals that precipitate in bioreactors are calcium phosphate, calcium sulphate, calcium carbonate, pyrite and struvite (Peng et al. 2018; Zhang et al. 2016). Conroy and Couturier (2010) observed that Ca and P were released in anaerobic reactor when the pH dropped under 6. They showed that the release corresponded exactly to the mineralisation of calcium phosphate. Goddek et al. (2018) also observed the solubilisation of P, Ca and other macronutrients in upflow anaerobic sludge blanket reactor (UASB) that turned acidic. Jung and Lovitt (2011) reported a 90% nutrient mobilisation of aquaculture-derived sludge at a very low pH value of 4. In this condition, all the macro- and micronutrients were solubilised. There is thus an antagonism between organic reduction and nutrient mineralisation. Indeed, organic reduction is maximal when the microorganisms are active for degrading the organic compounds, and this happens at pH in a range of 6–8. Because nutrient leaching occurs at pH below 6, for optimal organic reduction and nutrient mineralisation, the most effective would be to divide the process in two steps, i.e. an organic reduction step at pH close to neutral and a nutrient leaching step under acidic conditions. To our knowledge, no operation using this two-step approach has been yet reported. This opens a new field in wastewater treatment and more research for implementation in aquaponics is needed.
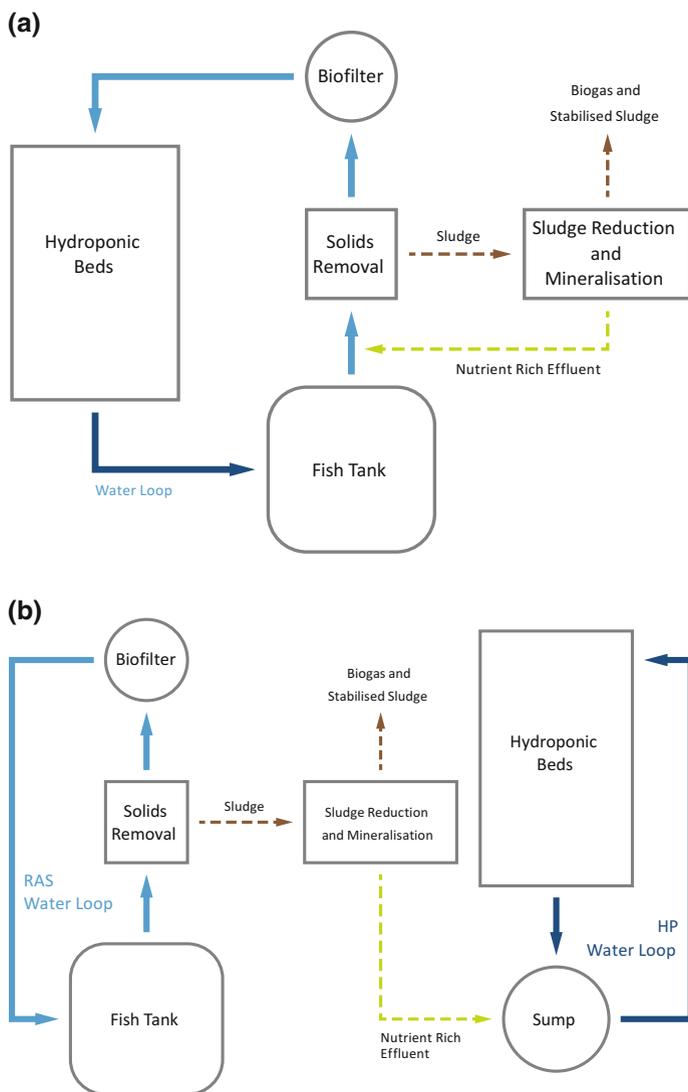
Fig. 10.1 Schematic implementation of sludge treatment in one loop aquaponic system (**a**) and in decoupled aquaponic system (**b**)

The choice of feed is also important in this context. In animal-based feeds where a major ingredient fraction is still based on animal sources (e.g. fishmeal, bone meal), bound phosphate, e.g. as apatite (derived from bone meal), is easily available under acidic conditions, whereas plant-based feeds contain phytate as a major phosphate source. Phytate in contrast to, e.g. apatite requires enzymatic (phytase) conversion (Kumar et al. 2012), and so the phosphate is not as easily available.

10.3   Aerobic Treatments

Aerobic treatment enhances the oxidation of the sludge by supporting its contact with oxygen. In this case, the oxidation of the organic matter is driven mainly by the respiration of heterotrophic microorganisms. CO_2, the end product of respiration, is released as is shown in Eq. (10.1).

$$C_6H_{12}O_6 + 6\,O_2 \rightarrow 6\,CO_2 + 6\,H_2O + \text{energy} \tag{10.1}$$

This process in aerobic reactors is mainly achieved by injecting air into the sludge–water mixture with air blowers connected to diffusers and propellers. Air injection also ensures a proper mixing of the sludge.

During this oxidative process, the macro- and micronutrients bound to the organic matter are released. This process is called aerobic mineralisation. Therefore, further nutrients can be recycled during the mineralisation process, whereas some nutrients, e.g. sodium and chloride, can also exceed their threshold for hydroponic application and must be monitored carefully before application (Rakocy et al. 2007). Aerobic mineralisation of organic matter, derived from the solid removal unit (e.g. clarifier or drum filter) in RAS, is an easy way to recycle nutrients for subsequent aquaponic application.

Moreover, during the aerobic digestion process, the pH drops and promotes the mineralisation of bound minerals trapped in the sludge. For example, Monsees et al. (2017) showed that P was released from RAS sludge due to this pH shift. This decrease in pH is mainly driven by respiration and to a lower extent probably by nitrification.

Due to a constant supply of oxygen via aeration of the mineralisation chamber and the abundance of organic matter, heterotrophic microorganisms find ideal conditions to grow. This results in an increase of respiration and the release of CO_2 that dissolves in water. CO_2 forms carbonic acid which dissociates and thereby lowers the pH of the process water as illustrated in the following equation:

$$CO_{2(g)} + 2\,H_2O \rightarrow H_3O^+ + HCO_3^- \tag{10.2}$$

RAS-derived wastewater often contains NH_4^+ and additionally is characterised by a neutral pH of around 7, because the pH in RAS needs to be kept at that level to ensure optimal microbial conversion of NH_4^+ to NO_3^- within the biofilter (i.e. nitrification). The nitrification process can contribute to the decrease in pH in aerobic reactors in the starting phase by releasing protons to the process water as can been seen in the following equation:

$$NH_4^+ + 2\,O_2 \rightarrow NO_3^- + 2\,H^+ + H_2O + \text{energy} \tag{10.3}$$
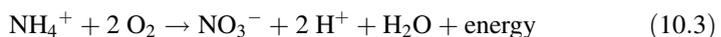
This is at least valid for the starting phase where the pH is still above 6. At a pH $\leq$ 6, nitrification might significantly slow down or even cease (Ebeling et al. 2006). However, this does not represent a problem for the mineralisation unit.

The general decrease of the pH in the aerobic mineralisation unit in the ongoing process is the main driver of the release of nutrients present under the form of precipitated minerals as calcium phosphates. Monsees et al. (2017) noted that around 50% of the phosphate in the sludge was acid soluble, derived from a tilapia RAS where a standard feed containing fishmeal was applied. Here, around 80% of the phosphate within the RAS was lost by the cleaning of the decanter and the discarding of the sludge–water mixture. Considering this fact, the big potential of mineralisation units for aquaponic applications becomes clears.

The advantages of aerobic mineralisation are the low maintenance with no need for skilled personnel and no subsequent reoxygenation. The enriched water can be used directly for plant fertilisation, ideally managed by an online system for the adequate preparation of the nutrient solution. A disadvantage compared to anaerobic mineralisation is that no methane is produced (Chen et al. 1997) and, as already mentioned, the higher energy demand due to the need for constant aeration.

10.3.1 Aerobic Mineralisation Units

A design example of an aerobic mineralisation unit is presented in Fig. 10.2. The inlet is connected to the solid removal unit via a valve, which allows discontinuous refilling of the mineralisation chamber with a mixture of sludge and water. The
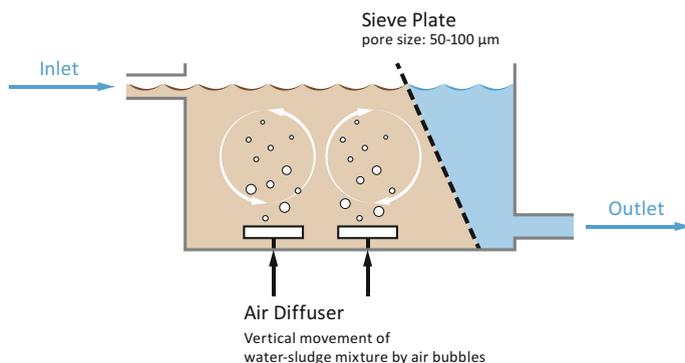

Fig. 10.2 Schematic example of an aerobic mineralisation unit operated in a batch mode. Mineralisation chamber (brown) is separated from the outlet chamber (blue) by a sieve plate that is covered by a solid cover plate during the mineralisation process (strong aeration) to prevent clogging and formation of fine particles. Organic-rich water from a clarifier or drum filter enters the mineralisation unit via the inlet. After a mineralisation cycle is completed, nutrient-rich, solid-free water exits the mineralisation unit via the outlet and is either directly transferred to the hydroponic unit or kept in a storage tank until needed

mineralisation chamber is aerated via compressed air to promote the respiration of heterotrophic bacteria and to keep anaerobic denitrification processes as minimal as possible. To prevent organic material from leaving the mineralisation chamber, a sieve plate could serve as a barrier. Ideally, a second, impermeable cover plate should be used to cover the sieve during the mineralisation process (during aeration). This should prevent the sieve plate from clogging as during the heavy aeration the organic material would be constantly moved against the sieve plate. Before transferring the nutrient-rich water from the mineralisation chamber to the hydroponic unit, aeration is stopped to allow the particles to settle. Subsequently, the cover plate is removed, and the nutrient-enriched water can pass through the sieve plate and leave the mineralisation chamber via the outlet as suggested in Fig. 10.2. Finally, the cover plate is put in place again, mineralisation chamber is refilled with RAS-derived sludge–water mixture, and the mineralisation process starts again (i.e. batch process).

The mineralisation unit should have at least twice the volume of the clarifier to allow for a continuous mineralisation. One mineralisation cycle can last for up to 5–30 days depending on the system, organic load and required nutrient profile and has to be elaborated for each individual system. For systems including a drum filter, as it is the case in most modern RAS, the mineralisation unit size has to be adjusted according to the daily or weekly sludge outflow of the drum filter. Since that has not been tested in an experimental setup so far, specific recommendations are not currently possible.

10.3.2   Implementation

An example of the implementation of an aerobic mineralisation unit into a decoupled aquaponic system is presented in Fig. 10.3. Since no pre- and post-treatment (e.g. re-oxygenation) is required, the mineralisation unit can be directly placed between the solid removal unit and the hydroponic beds. By installing a valve before and after the mineralisation unit, a discontinuous operation and nutrient delivery to the hydroponic unit on-demand are possible, but in many cases, an additional storage tank would be required. Ideally, after directing nutrient-rich water to the hydroponic unit, the displaced water is replaced with new sludge and water from the solid removal unit. Depending on the volume of the mineralisation unit, it is important to note that refilling with new sludge–water mixture can lead to an increase in pH again, and thus the mineralisation process could be interrupted. By increasing the size of the mineralisation unit, this effect would be buffered. In the study by Rakocy et al. (2007) investigating liquid organic waste from two aquaculture systems, a retention time of 29 days for aerobic mineralisation resulted in a substantial mineralisation success. Nevertheless, this also depends on the TS content within the mineralisation chamber, on the feed applied to the RAS, on the temperature and on the nutrient requirements of the plants that are produced within the hydroponic unit.
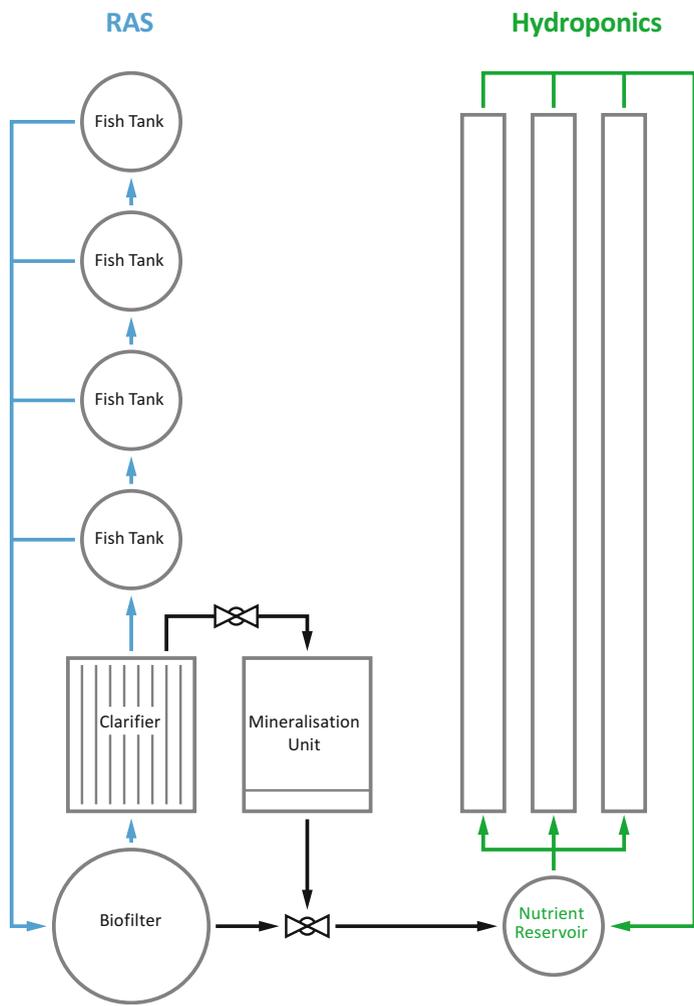
Fig. 10.3 Schematic picture of a decoupled aquaponic system including an aerobic mineralisation unit. Water can be transferred to the nutrient reservoir either from the RAS water loop or directly from the mineralisation unit

10.4 Anaerobic Treatments

Anaerobic digestion (AD) has long been used for the stabilisation and reduction of sludge mass process, mainly because of the simplicity of operation, relatively low costs and production of biogas as potential energy source. General stoichiometric representation of anaerobic digestion can be described as follows:

$$CnHaOb + (n - a/4 - b/2) \cdot H_2O \rightarrow (n/2 - a/8 + b/4) \cdot CO_2 \\ + (n/2 + a/8 - b/4) \cdot CH4 \qquad (10.4)$$
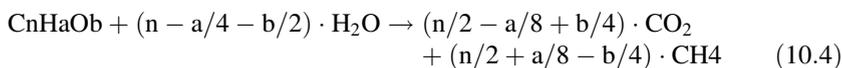
Equation 10.4 Biogas general mass balance (Marchaim 1992).

And the theoretical methane concentration can be calculated as follows:

$$[CH_4] = 0.5 + (\ a/4 + b/2\ )/2n \qquad (10.5)$$

Equation 10.5 Theoretical expected methane concentration in the biogas (Marchaim 1992).

The ultimate products from AD are mostly inorganic material (e.g. minerals), slightly degraded organic compounds and biogas which is typically composed of >55% methane (CH_4) and carbon dioxide (CO_2), with only small levels (<1%) of hydrogen sulphide (H_2S) and total ammonia nitrogen ($NH_3^+/NH4^+$) (Appels et al. 2008).

During the process of AD, the organic sludge undergoes considerable changes in its physical, chemical and biological properties and schematically can be divided into four stages (Fig. 10.4). The first stage is hydrolysis, where complex organic matter such as lipids, polysaccharides, proteins and nucleic acids degrade into soluble organic substances (sugars, amino acids and fatty acids). This step is generally considered rate-limiting (Deublein and Steinhauser 2010). In the second acidogenesis step, the monomers formed in the first step split further, and volatile fatty acids (VFA) are produced by acidogenic (fermentative) bacteria along with ammonia, CO_2, H_2S and other by-products. The third step is acetogenesis, where the VFA and alcohols are further digested by acetogens to produce mainly acetic acid as well as CO_2 and H_2. This conversion is controlled to a large extent by the partial pressure of H_2 in the mixture. The last step is methanogenesis where methane is mainly produced by two groups of methanogenic bacteria: acetotrophic archaea, which split acetate into methane and CO_2, and hydrogenotrophic archaea, which use hydrogen as an electron donor and carbon dioxide as electron acceptor to produce methane (Appels et al. 2008).

Various factors such as sludge pH, salinity, mineral composition, temperature, loading rate, hydraulic retention time (HRT), carbon-to-nitrogen (C/N) ratio and volatile fatty acid content influence the digestibility of the sludge and the biogas production (Khalid et al. 2011).

Anaerobic sludge treatment from RAS began about 30 years ago with reports on sludge from freshwater RAS (Lanari and Franci 1998) followed by reports on marine (Arbiv and van Rijn 1995; Klas et al. 2006; McDermott et al. 2001) and brackish water operations (Gebauer and Eikebrokk 2006; Mirzoyan et al. 2008). Recently, the use of UASB (Fig. 10.5) for AD of RAS sludge followed by biogas production as an alternative source of energy was suggested (Mirzoyan et al. 2010). The reactor is made of a tank, part of which is filled with an anaerobic granular sludge blanket containing the active microorganism species. Sludge flows upwards through a 'microbial blanket' where it is degraded by the anaerobic microorganisms and
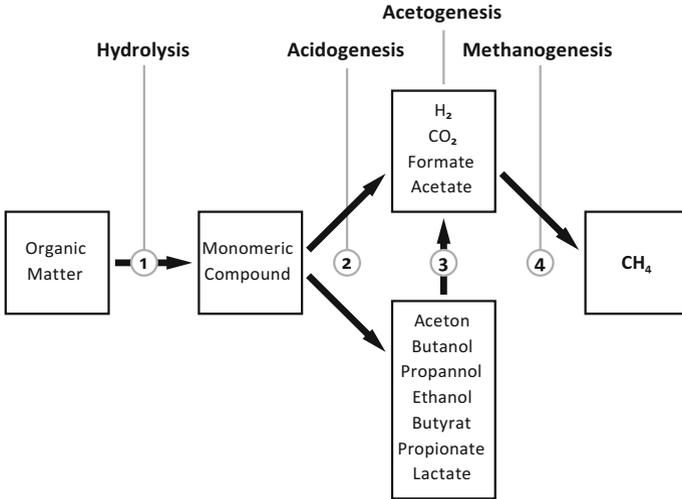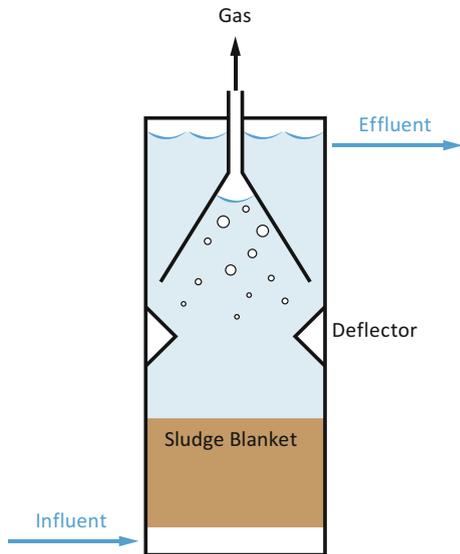
Fig. 10.4 Schematic diagram showing anaerobic degradation of organic matter based on Garcia et al. (2000)

Fig. 10.5 Scheme of an upflow anaerobic sludge blanket reactor (UASB)

biogas is produced. An inverted cone settler at the top of the digester allows gas–liquid separation. When the biogas is released from the floc, it is oriented into the cone by the deflectors to be collected. A slow mixing in the reactor results from the upwards flow coupled with the natural movement of the microbial flocs that are attached to biogas bubbles. At some point, the floc leaves the gas bubble and settles back down allowing for the effluent to be free from TSS, which can then be recycled

back to the system or released. The main advantages of the UASB are the low operational costs and simplicity of operation while providing high (>92%) solid-removal efficiency for wastes with low (1–3%) TSS content (Marchaim 1992; Yogev et al. 2017).

Two recent case studies demonstrated the use of UASB as a treatment for solids in pilot scale marine and saline RAS, which provide an example of the potential advantages of this unit in aquaponics (Tal et al. 2009; Yogev et al. 2017). A detailed look at the carbon balance suggested that about 50% of the introduced carbon (from feed) was removed by fish assimilation and respiration, 10% was removed by aerobic biodegradation in the nitrification bioreactor and 10% was removed in the denitrification reactor (Yogev et al. 2017). Therefore, overall about 25% carbon was introduced into the UASB reactor of which 12.5% was converted to methane, 7.5% to CO_2 and the rest (~5%) remained as nondegradable carbon in the UASB. In summary, it was demonstrated that the use of UASB allowed better water recirculation (>99%), smaller (<8%) production of sludge when compared with typical RAS that do not have on-site solid treatment, and recovery of energy that can account for 12% of the overall energy demand of the RAS. It should be noted that using UASB in aquaponics will also allow significant recovery of up to 50% more nutrients such as nitrogen, phosphorus and potassium since they are released into the water as a result of solid biodegradation (Goddek et al. 2018).

The anaerobic membrane bioreactor (AnMBR) is a more advanced technology. The main process consists of using a special membrane to separate the solids from the liquid instead of using a decanting process as in UASB. The sludge fermentation occurs in a simple anaerobic tank and the effluents leave it through the membrane. Depending on the membrane pore size (going down to 0.1–0.5 μm) even microorganisms can be retained. There are two types of membrane bioreactor design: one uses a side-stream mode outside the tank, and the other has the membrane unit submerged into the tank (Fig. 10.6), the latter being more favourable in AnMBR application due to its more compact configuration and lower energy consumption (Chang 2014). Membranes of different materials such as ceramic or polymeric (e.g. polyvinylidene fluoride (PVDF), polyethylene, polyethersulfone (PES), polyvinyl chloride (PVC)) may be configured as plate and frame, hollow fibre or tubular units (Gander et al. 2000; Huang et al. 2010). AnMBR has several significant advantages over typical biological reactors such as the UASB, namely, decoupling of (long) sludge retention time (SRT) and (short) hydraulic residence time (HRT), hence enabling the problem of the AD process's slow kinetics to be overcome; very high effluent quality in which most nutrients remain; and removal of pathogens and a small footprint (Judd and Judd 2008). In addition, efficient biogas production in the AnMBR can possibly result in a net energy balance.

While this technology deserves a lot of attention and research, it should be noted that since it is a fairly new technology, there are still several significant drawbacks that must be addressed before AnMBR would be adopted by the aquaculture industry. These are the high operational costs due to membrane maintenance to prevent biofouling, regular membrane exchange and high CO_2 fraction (30–50%) in the biogas which limits its utilisation and contributes to greenhouse gas (GHG)
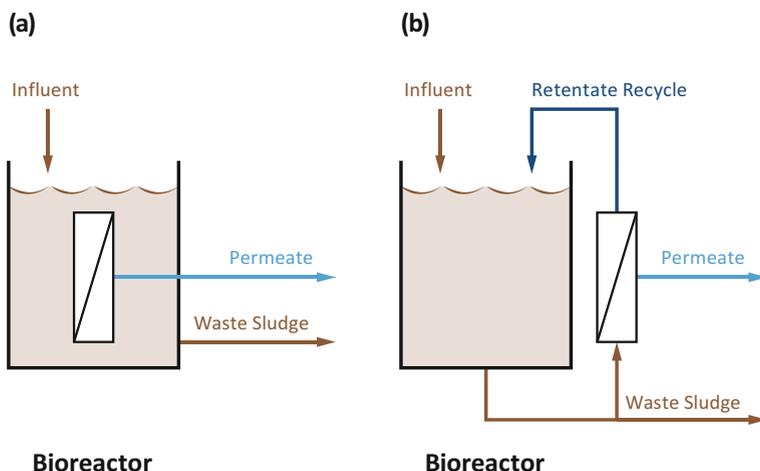
Fig. 10.6 (**a**) Side-stream MBR with a separate filtration unit with the retained fluid recycled back to the bioreactor; (**b**) submerged MBR: filtration unit integrated into the bioreactor. (Gander et al. 2000)

emission (Cui et al. 2003). On a positive note, in the near future, new biofouling prevention techniques will be developed while the membrane cost will certainly drop with the broader use of this technology. The combination of a UASB with a membrane reactor to filter the UASB effluent has been successfully studied to remove organic carbon and nitrogen (An et al. 2009). This combination seems a promising option for aquaponics for safe and sanitary use of UASB effluents.

10.4.1 Implementation

One possible solution of implementing anaerobic reactors is in a sequential manner (see also Chap. 8). A 'high pH–low pH' combination allows for harvesting methane (and thus reducing carbon) in the first high pH step and mobilising nutrients in the decarbonised sludge in a subsequent low pH environment. The advantage of this method is that the carbon reduction under high pH conditions results in less VFAs, which can occur during the low pH second step (Fig. 10.7). This approach also allows for co-digestion of green vegetative matter (i.e. from any harvesting of plants, there will be waste vegetative matter which could be put through such a digester) to increase both biogas production and nutrient recovery from the overall scheme.

Another technical integration possibility has been presented by Ayre et al. (2017). They propose to discharge the effluent of a high-pH anaerobic digester to an algal culture pond. Within that pond, algae are grown, whose biomass can be used for animal–aquaculture feed or biofertilisation (Fig. 10.8). More detailed information on this approach can be found in Chap. 11.
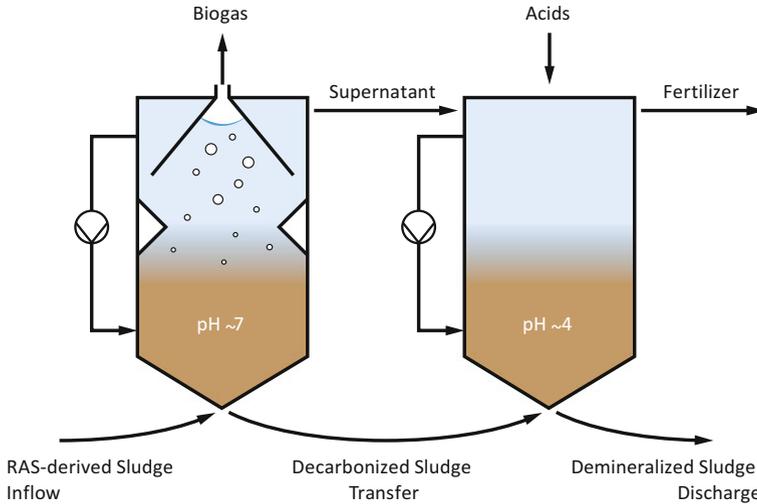
Fig. 10.7 Two-stage anaerobic system. In the first stage (high pH), the carbon will be removed from the sludge as biogas, whereas the low pH in the second stage allows nutrients that are trapped in the sludge do dissolve in the water. Usually, volatile fatty acids (VFA) would form in low pH environments. The removal of the carbon source in the first stage, however, limits VFA production in such a sequential setup

10.5 Methodology to Quantify the Sludge Reduction and Mineralisation Performance

To determine the digestion of aquaponic sludge treatment in aerobic and anaerobic bioreactors, a specific methodology needs to be followed. A methodology adapted for aquaponic sludge treatment purposes is presented in this chapter. Specific equations have been developed to precisely quantify their performance (Delaide et al. 2018), and these should be used to evaluate the performance of the treatment applied in a specific aquaponic plant.

In order to evaluate the treatment's performance, a mass balance approach needs to be achieved. It requires that TSS, COD and nutrient masses are determined for the all reactor inputs (i.e. fresh sludge) and outputs (i.e. effluents). The reactor content also needs to be sampled at the beginning and at the end of the studied period. The input, output and content of the reactors have to be perfectly mixed for sampling. Reactor input and output should basically be sampled every time the reactors are fed with fresh sludge.

Then, reactor sludge reduction performance (η) can be formulated as follows:

$$\eta_S = 100\%(1 - (\Delta S + S_{out})/S_{in})) \tag{10.6}$$
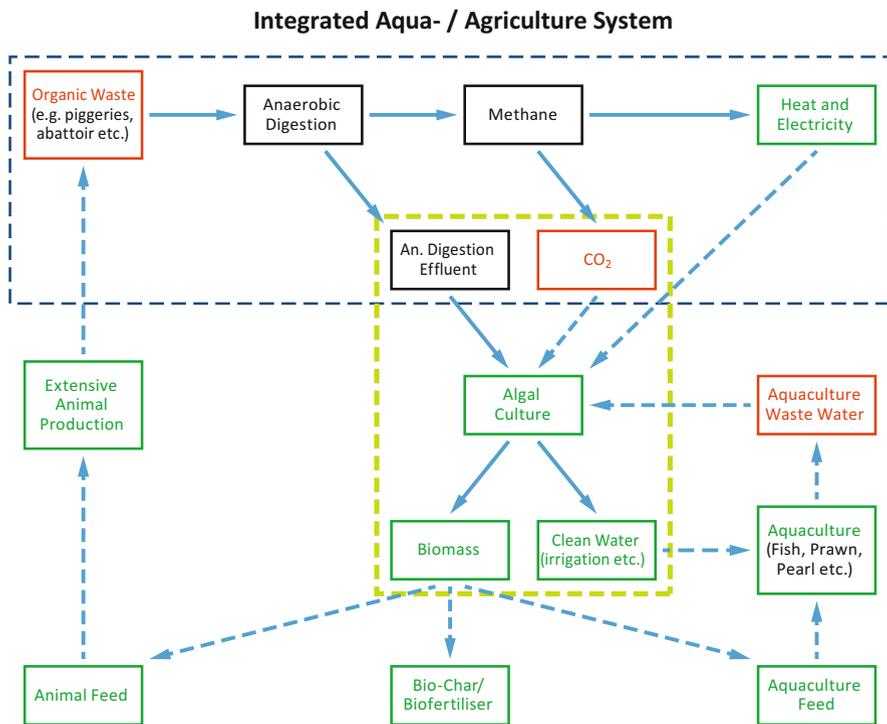
Integrated Aqua- / Agriculture System

Fig. 10.8 Anaerobic digestion system integrated with aquaculture and algal culture based on Ayre et al. (2017)

where ΔS is the sludge inside the reactor at the end of the studied period minus the one at the beginning of the period, S_{out} is the total sludge that left the reactor in the outflow, and S_{in} is the total sludge that entered the reactor via inflow.

For organic reduction, the sludge (i.e. the term S) can be characterised by the dry mass of sludge (i.e. TSS) or the mass of oxygen needed to oxidise the sludge (i.e. COD). Thus, for COD and TSS reduction performances, the smaller the accumulation and the smaller the quantity in the outflow, the higher the reduction performance (i.e. high percentage) and so the less solids discharged out of the loop.

Based on the same mass balance, the nutrient mineralisation performance of the treatment (ζ), i.e. the conversion into soluble ions of the macro- and micronutrients present in the sludge under undissolved forms, the following formula can be used:

$$\zeta_N = 100\%((DN_{out} - DN_{in})/(TN_{in} - DN_{in}))$$ (10.7)

where ζ is the recovery of N nutrient at the end of the studied period in percent, DN_{out} is the total mass of dissolved nutrient in the outflow, DN_{in} is the total mass of dissolved nutrient in the inflow and TN_{in} is the total mass of dissolved plus undissolved nutrients in the inflow.

   Thus, similar to the organic reduction performances, the smaller the accumulation inside the reactor and undissolved nutrient content in the outflow, the higher the mineralisation performance (i.e. high percentage) and so the dissolved nutrient recovered in the effluent (or outflow) for aquaponic crop fertilisation (see Example 10.1). The presented mass balance equations are used in the example box.

Example 10.1

The digestion performance of a 250-L anaerobic bioreactor has been evaluated for an 8-week period. It was fed once a day with 25 L of fresh sludge coming from a tilapia RAS system, and the equivalent supernatant volume (or output) was removed from the bioreactor. The fresh sludge (input) had a TSS of 10 g dry mass (DM) per litre or 1%, and the supernatant (output) had a TSS of 1 gDM/L or 0.1%. The TSS inside the bioreactor at the beginning and at the end of the period was 20 gDM/L. Consequently, the total DM inputs, outputs and inside the bioreactor during the evaluated period are calculated as follows:

DM in $= 0.01$ kg/Ld $\times$ 25 L $\times$ 7 days $\times$ 8 weeks $= 14$ kg
DM out $= 0.001$ kg/Ld $\times$ 25 L $\times$ 7 days $\times$ 8 weeks $= 1.4$ kg
DM to $=$ DM tf $= 250$ L $\times$ 0.02 kg/L $= 5$ kg

The TSS reduction performance (η_{TSS}) of the bioreactor can then be calculated as follows:

$$\eta_{TSS} = 100\%(1 - ((5 - 5) + 1.4)/14)) = 90\%$$

   The bioreactor P mineralisation performance can be evaluated knowing that the fresh sludge (input) had a concentration of dissolved P of 15 mg/L and a total P content of 90 mg/L. The concentration of dissolved P in the supernatant (output) was 20 mg/L. Consequently, the total P content in the input, the total dissolved P in the inputs and outputs during the evaluated period are calculated as follows:

TP in $= 0.090$ g/Ld $\times$ 25 L $\times$ 7 days $\times$ 8 weeks $= 126$ g
DP in $= 0.015$ g/Ld $\times$ 25 L $\times$ 7 days $\times$ 8 weeks $= 21$ g
DP out $= 0.020$ g/Ld $\times$ 25 L $\times$ 7 days $\times$ 8 weeks $= 28$ g

The P mineralisation performance (ζ_P) of the bioreactor can then be calculated as follows:

$$\zeta_P = 100\%((28 - 21)/(126 - 21)) = 6.67\%$$

10.6 Conclusions

Fish sludge treatment for reduction and nutrient recovery is in an early phase of implementation. Further research and improvements are needed and will see the day with the increased concern of circular economy. Indeed, fish sludge needs to be considered more as a valuable source instead of a disposable waste.

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