*Manuscript
Click here to view linked References

Process Safety and Environmental Protection, Volume 94, March 2015, Pages 285–306

Rotating biological contactors for wastewater treatment – a review

Abstract

Rotating biological contactors (RBCs) for wastewater treatment began in the 1970's. Removal of organic matter has been targeted within organic loading rates of up to 120 $g.m^{-2}d^{-1}$ with an optimum at around 15 $g.m^{-2}d^{-1}$ for combined BOD and ammonia removal. Full nitrification is achievable under appropriate process conditions with oxidation rates of up to 6 $g.m^{-2}d^{-1}$ reported for municipal wastewater. The RBC process has been adapted for denitrification with reported removal rates of up to 14 $g.m^{-2}d^{-1}$ with nitrogen rich wastewaters. Different media types can be used to improve organic/nitrogen loading rates through selecting for different bacterial groups. The RBC has been applied with only limited success for enhanced biological phosphorus removal and attained up to 70% total phosphorus removal. Compared to other biofilm processes, RBCs had 35% lower energy costs than trickling filters but higher demand than wetland systems. However, the land footprint for the same treatment is lower than these alternatives. The RBC process has been used for removal of priority pollutants such as pharmaceuticals and personal care products. The RBC system has been shown to eliminate 99% of faecal coliforms and the majority of other wastewater pathogens. Novel RBC reactors include systems for energy generation such as algae, methane production and microbial fuel cells for direct current generation. Issues such as scale up remain challenging for the future application of RBC technology and topics such as phosphorus removal and denitrification still require further research. High volumetric removal rate, solids retention, low footprint, hydraulic residence times are characteristics of RBCs. The RBC is therefore an ideal candidate for hybrid processes for upgrading works maximising efficiency of existing infrastructure and minimising energy consumption for nutrient removal. This review will provide a link between disciplines and discuss recent developments in RBC research and comparison of recent process designs are provided (section 2). The microbial features of the RBC biofilm are highlighted (section 3) and topics such as biological nitrogen removal and priority pollutant remediation are discussed (section 4 & 5). Developments in kinetics and modelling are highlighted (section 6) and future research themes are mentioned

Keywords: Bioaugmentation, biofilm, biological wastewater treatment, biological nitrogen removal, modelling, rotating biological contactor.

Author information: Francis Hassard, Jeremy Biddle, Elise Cartmell, Bruce Jefferson, Sean Tyrrel, Tom Stephenson.

a. Cranfield University, Cranfield, MK43 0AL, UK.
b. Bluewater Bio Ltd., Compass House, 22 Redan Place, London, W2 4SA, UK.

Corresponding author: Tom Stephenson, +44 (0)1234 754054, t.stephenson@cranfield.ac.uk.

Nomenclature

A_d = total disc surface area (L^2)

A_{exp} = area of exposed disc (L^2)

A_{sub} =area of submerged disc (L^2)

A_t = cross sectional area of tank (L^2)

C_{Bf} = compound concentration at the biofilm surface (ML^{-3})

C_{Lf} = compound concentration at liquid film surface (ML^{-3})

C_i = influent compound concentration (ML^{-3})

C_e = effluent compound concentration (ML^{-3})

C_T = compound concentration in the tank (ML^{-3})

C* = equilibrium compound concentration at a given temperature (ML^{-3})

D_L = diffusion coefficient of oxygen in water (L^2T^{-1})

e = distance from disc edge to the basin (L)

g = acceleration due to gravity (LT^{-2})

H = distance between the disc centre to the liquid free surface (L)

K_C = half saturation constant for compound (ML^{-3})

K_L^{air} = oxygen mass transfer coefficient film

K_L = overall oxygen mass transfer coefficient (LT^{-1})

$K_L a_t$, = volumetric oxygen mass transfer coefficient total (T^{-1})

N = number of discs

N_v = volume renewal number (T^{-1})

Q = reactor flow rate (L^3T^{-1})

r =radius of disc (L)

r_a = substrate removal rate ($ML^{-2}T^{-1}$)

s = half space between discs (L)

t_R =contact time per rotation (T)

V = wet volume of reactor (L^3)

V_{Lf} = volume of liquid film (L^3)

U_{max} =maximum substrate removal rate ($ML^{-2}T^{-1}$)

X_a = concentration of attached biomass (ML^{-3})

Y_a = yield coefficient for attached biomass

δ =liquid film thickness (L)

δ_{bf} = biofilm thickness (L)

μ = Absolute viscosity of a liquid (MLT^{-1})

μ_{max} = maximum specific growth rate (T^{-1})

ρ = fluid density (ML^{-3})

ω =rotational speed (RPM)

$\omega' = \omega/60$

□ =disc diameter (L)

$□_0$ =wetted disc diameter (L)

1. Introduction

Wastewater treatment processes should comply with standards that ensure environmental protection, whilst be efficient to minimise socio-economic burden (Ainger et al. 2009). The main priorities for wastewater treatment (WwT) are effluent quality, cost, energy efficiency and nutrient removal/recovery (STOWA, 2012). Regulatory agencies aim to improve local environmental health using advanced forms of WwT such as biological nutrient removal (BNR). To achieve tighter effluent standards, traditional biological treatment is largely reliant on increasing energy input through extended reactor aeration or retention time. Already, ~55% of the energy budget for sewage

treatment is used in aeration (Ainger et al. 2009). The development of wastewater treatment technology is critical to improve the long term sustainability of necessary treatment capacity (Hoyland et al. 2008; STOWA, 2012).

Rotating biological contactors (RBC) are called disc, surface, media and biofilm reactors and provide an alternative to the activated sludge (AS) process. The RBC has a solid media that encourages microbial growth in a static biofilm (Singh and Mittal, 2012). The RBC media is arranged in a series of plates or discs which are rotated on a shaft through a biozone trough by motor or air drive (Patwardhan 2003). The rotation leads to bulk fluid mixing, convection through media/biofilm pores, compound diffusion to the film and subsequent product exchange with the reactor and surroundings (Rittman and McCarty, 2001). Biological processes occur inside a fixed microbial biofilm, which contains components of active/non-active biomass, biofilm extracellular matrix and debris (Arvin and Harremoës, 1990). The RBC combines bacterial growth and substrate utilization with a natural biomass separation system; however effluent quality and process stability is contingent on a distal sedimentation zone. The principal advantage of biofilm processes, such as RBCs, is that the mean cell residence time (MCRT) is uncoupled from hydraulic residence time (HRT). This could allow higher organic loadings and resistance to toxic shocks than suspended culture systems (Najafpour et al. 2006; Cortez et al. 2008). Fixed RBC biofilms offer higher substrate affinity, resistance to traumatic events and exhibit quicker recovery from starvation than suspended counterparts (Batchelor et al. 1997; Bollmann et al. 2005). This could be due to differential gene expression, physical or chemical isolation and the presence of stronger diffusion gradients (Cohen 2001). The RBC biofilm is especially useful for the degradation of refractory agents due to high bacterial density and compound immobilisation (Singh et al. 2006). The presence of gradients can promote aerobic, anaerobic and anoxic conditions within a single amalgamated system, which promotes different removal regimes (Dutta et al. 2007).

The RBC biofilm can undertake biochemical oxygen demand (BOD) removal and BNR for domestic and high strength sewage (Hiras et al., 2004; Vlaeminck et al. 2009) and limited enhanced phosphorus recovery (Yun et al. 2004). Mounting evidence suggests that the RBC consortia can offer specific contaminant remediation for certain aromatics molecules including hydrocarbons, heavy metals, xenobiotics and pharmaceuticals / personal care products (PPCP) under appropriate process conditions (Novotný et al. 2011; Jeswani and Mukherji, 2012; Orandi et al., 2012; Simonich et al., 2002). Rotating biological contactors are used for wastewater treatment requiring low land area, maintenance, energy or start-up costs and can facilitate a more decentralised water treatment network (Hiras et al. 2004; Dutta et al. 2007). Traditional RBC design, maintenance and operation relied on process theory; however the biochemistry, biofilm modelling and microbial ecology have received increased attention recently (Wuertz et al. 2004). Patwardhan (2003), reviewed the process design aspects of RBCs and Cortez et al. (2008) highlighted some performance related process parameters. However despite investment and research in areas such as enhanced biological phosphorus removal, denitrification, cost and scale-up, the RBC is yet to achieve full potential.

1.1. Process development history

The RBC concept originated in Germany in 1920's where it was described as a 'rotating aerobic mass' fixed to a media support (Chan and Stenstrom, 1981), although the first plant was registered in the United States and was named the 'Contact Filter' or 'Biologic Wheel' consisting of partially submerged rotating plates (Doman, 1929). This device served as an alternative to the trickling filter with $1/10^{th}$ the land area, and lower power cost than AS (Allen, 1929). Commercial interest in RBCs was minimal, until the modern emergence of the so called 'drip body immersion systems' (Hartman, 1960). The design was patented (Hartmann, 1961) and the first recorded experimental pilot RBC was undertaken to test performance (Popel, 1964). This landmark study informed future RBC design which progressed in the 1960's. For example the surface BOD_5 loading from this study of ~3g.m^2.d^-

[1] is similar to modern overall organic surface loadings of 3-15 g.BOD_5.m^2d^{-1} that have been applied recently (Rittmann and Mcarty, 2001). The availability of stronger, lighter and affordable materials such as plastics increased the stability of media and increased the surface area available for microbiological growth, which improved treatment capacity. This allowed a plethora of capital ventures in the 1960 and 70's. The RBC was applied for biological treatment under a variety of influent types, organic and hydraulic regimes (Ritmann and Mcarty, 2001, Cortez et al. 2008). A Japanese company known as Kubota submitted a patent application for an RBC capable of simultaneous nitrification and denitrification, using variable submergence to facilitate multiple nutrient removal regimes (Sim 1988). A series of process failures have been noted for RBCs, many were due to inappropriate mechanical design which did not account for biomass growth, often leading to shaft, bearing and media malfunctions (Mba et al. 1999). A report suggested that equipment warranty should protect the owner from failure (Weston, 1985), however often liability contracts rarely exceeded 3 years which provided little stimulus to fix inherent mechanical issues (Griffin and Findlay, 2000). Another challenge was supplier competition led to an exaggeration of possible removal rates (Rittmann and McCarty, 2001); allowable loadings varied by a factor of 7 between suppliers (Ross et al. 1994). Unlike other major biological processes, designers were initially reliant on proprietors design criterion for process control (Ross et al. 1994). Hydraulic loading was previously applied as a design parameter, but was usually inappropriate by not considering organic strength; biodegradability, toxicity and temperature which impact microbial process performance (Steiner 1997). Design criteria should be used that incorporate fundamental parameters including microbial activity, organic loading and substrate utilisation rate.

2. Process engineering of RBCs

2.1. Types

There are two main types of RBC; integral and modular. Integral systems consist of a single unit combining primary settlement, RBC biozone and either a contained or separate final clarifier. (Fig. 1a). Integral units are usually contained within a package plant and have a treatment capacity of ≤250 population equivalents (PE) (Findlay et al. 1993). Conversely, modular systems have separate operations for primary, secondary, and solids treatment respectively and usually treat PE >1000 (Griffin and Findlay, 2000), which allows more flexible process configurations (Fig. 1b,c). However size and weight constraints generally limit RBCs to a size of 3.5 m disc diameter. Modular RBCs can be operated using parallel flow separation between units allowing operation within acceptable loading limits (Fig. 1b). In contrast, if effluent quality is of principal concern, RBCs are often operated in series, with an n^{th} RBC operating distal in the flow sheet (Fig. 1c). Typically a submergence of 40% (wet disc level), is used (Cortez et al. 2008). By increasing the submergence (Fig. 1d), the conditions in the reactor become increasingly anaerobic which could favour processes that require reduced oxygen levels such as denitrification (Teixeira and Oliveira, 2001). Hybrid systems operate a RBC combined with another unit operation to improve the stability of a process that has strong or variable loading, increase load capacity or improve the achievable effluent standard (Vesilind, 2003; Hoyland et al. 2010). Common configurations include a RBC/biofilm (Fig. 1e) or RBC/suspended growth combination which can be used for the upgrade of capacity (roughing) or provide tertiary treatment (Fig. 1f) (Vesilind, 2003, Upton et al. 1995). The RBC/wetland combination has been applied to improve discharge consents for small works and provide a storm flow buffer (Griffin, 2003) (Fig. 1e). For longevity, the RBC is protected using ultra violet light resistant media (e.g plastic with carbon black) or by covering the RBC within protective casing which can also reduce heat loss and flies/odour.

Note to publisher: insert fig 1(a-f).

2.2. Cost

The capital expenditure (CAPEX) and operational expenditure (OPEX) of RBCs has been minimised through reduced commissioning, monitoring and maintenance costs compared to AS processes. In the UK, half the CAPEX for RBCs is related to mechanical and electrical components. The CAPEX cost per head in RBCs is inversely proportional to the PE for treatment. At PE >1000 the CAPEX cost decreased by up to 50% (Upton et al. 1995). For example Labella et al. (1972) compared the cost of an activated sludge plant (ASP) and RBC system treating winery waste with a flow of 1.8 x 10^3 m^3d^{-1}. They noted that while capital expenditure were similar, estimated power consumption was less than half that of a concrete tank aerated ASP. An RBC was found to be on average 35% cheaper per PE per year compared to trickling filters due to lower land area and running costs (Upton et al. 1995). However other authors have suggested that the OPEX of an RBC are similar to suspended growth systems and savings are only apparent with CAPEX (Ware et al. 1990). Fountoulakis et al. (2009) identified that RBCs had 29% lower but 44% higher CAPEX than packed bed filters and horizontal surface flow wetlands respectively. In addition RBCs were shown to have five times the power consumption than packed bed filters when operated within the organic loading rate (OLR) range of 0.53-2.01 $kg.COD.m^{-3}d^{-1}$. The power efficiency of a RBC operated a 7.5 horse power motor ranged from 72-88% at 25-100% load capacity respectively (Brenner and Opaken 1984). However RBCs are appropriate for decentralised water treatment systems which generally have lower OPEX costs compared to a centralised approach which may require specialist labour and process control (Fountoulakis et al. 2009).

2.3. Substrate

Substrate dependent parameters in RBCs are staging, organic loading, recycle and flowsheet position. The hydraulic considerations include hydraulic residence time (HRT), tip speed, media specific surface area, compound transfer rate and submergence. However there is considerable overlap between these parameters, for example the inverse relationship between HRT and OLR (Patwardhan, 2003). Another example is the association between rotational speed, oxygen transfer rate (OTR) and biofilm thickness. A key criterion for RBC reactors is surface organic load which is defined as substrate (kg.COD/N/pollutant) applied per square metre (specific or nominal) of media per day. In RBCs, as the loading rate increases the removal rate increases in proportion until another parameter becomes limiting (Fig. 2). For example Hiras et al. (2004) operated a two stage predenitrification and aerobic RBC for the treatment of settled municipal sewage. A decrease in the percentage removal of COD with increasing OLR was observed from 50 to 35% at OLR of 90 and 360 $gm^{-2}d^{-1}$ respectively. This can be explained by biofilm oxygen transfer rate limiting the efficiency of substrate utilisation (Di Palma et al. 2009). However the organic removal rate increased from 45 to 125 $gm^{-2}d^{-1}$ suggesting that there was more capacity for bulk COD removal in the system. Therefore the highest substrate removal rate is achieved at the maximum loading before the transfer of rate limiting compound is exceeded (Fig. 2). In RBC biofilms mass transfer restrictions usually masks biological reaction kinetic limitations. As both substrates diffuse from the bulk fluid in the same direction and one or both will become limiting at a certain depth in the biofilm. In RBC biofilms there is an equilibrium between the rate of substrate consumption and diffusional transfer which influences the penetration depth (Stewart and Franklin, 2008). Under constant loading the microbial community will attain steady state based on available substrates and competition for electron acceptors and space. In the biofilm there is a layering of bacteria based on prevailing conditions with the lowest substrate redox state proximal to the media (Okabe et al. 1999).

Note to publisher: insert fig 2.

Staging is a physical barrier employed to separate the wastewater chemistry within or between reactors (Fig. 1b), which leads to a stepwise reduction in the bio-available substrate to the point where the reactor approaches plug-flow (Ayoub and Saikaly 2004). This localisation selects for microbial populations adapted to the physiochemical conditions within each stage. This

improves removal rate, process stability and permits autotrophic nitrification at higher organic loads than normally possible (Tawfik et al., 2002; Kulikowska et al. 2010; Najafpour et al., 2006). Staging can permit enhanced ability to manage shock loads providing the biomass has sufficient substrate. The positive impact of staging on RBC performance was found to be negligible after four stages (Adreadakis 1987), although this is dependent on wastewater load and composition. Step feeding can be used to reduce the initial effective substrate concentration. Ayoub and Saikaly (2004) showed that step feeding had minimal impact on removal of RBC bulk COD removal rate, however NH_4-N removal increased by 18%, by staggering the organic load which reduced the likelihood of oxygen limitation (Rittmann et al. 1983). Recycling effluent permits greater portions of the biofilm to nitrify by diluting the influent organic concentration (Ayoub and Saikaly 2004). The recycle can be either pre, post or from the clarifier depending on treatment aim (Fig. 1c). Recycling settled solids helps aid bacterial retention as sloughed biomass is returned to the reactor. Other biomass associated products like extracellular enzymes may be recycled which could aid the breakdown of complex polymers, which constitute roughly half of domestic wastewater (Confer and Logan, 1998).

2.4. Hydrodynamics

Understanding the hydrodynamics of RBCs is important to maintain appropriate biomass thickness, encourage compound mass transfer and prevent unequal biomass distribution (Di Palma et al. 2003; Griffin and Findlay, 2000). Rotation of media creates a head difference leading to convective air/water exchange. Increasing tip speed increases the total oxygen transfer rate in a pseudo-linear fashion (Rittmann et al. 1983). However the energy usage for motor drive increases exponentially with increasing rotational speed. For minimal OPEX the lowest rpm should be selected and rotor speeds of 0.7-2.0 rpm are common (Mba et al. 1999). However, some high rate systems are known to exceed this speed (Hoyland et al. 2008). Microscale biofilm structure can influence compound mass transfer into the RBC biofilm. For example high biofilm roughness influenced the RBC biofilm boundary layer thickness by changing hydraulics and flow velocity perpendicular to the biofilm. This increased the rate of diffusion through the boundary layer and DO concentration in the biofilm (De la Rosa and Yu 2005).

2.5. Media composition

The RBC media can be present as discs, mesh plates, saddles or rings in a packed bed reactor, which resembles a partially submerged, rotating, moving bed biofilm reactor (Ware et al 1990; Sirianuntapiboon and Chumlaong, 2013). The RBC media commonly has a specific surface area of 150-250 m^2m^{-3} for biofilm growth which supports high removal rates at low HRTs. Lower density media is normally applied at the front-end of the works which typically has high organic loads (Cortez et al. 2008). Support media should be insoluble, have high mechanical and biological stability, and be cost effective (Leenen et al. 1996). The media physicochemical composition and architecture both impact on the microbial biofilm and the removal rate of substrates (Tawfik and Klapwijk, 2010; Stephenson et al. 2013). A comparison between the oxygen transfer efficiency in RBCs was between 2-5 and 1-2 $kgkWh^{-1}$ for comparable packed bed and disc RBCs respectively (Mathure and Patwardhan, 2005). However previously it was noted that any performance gains from packed bed RBCs are usually offset by higher CAPEX costs and reliability issues (Ware et al. 1990). Polyurethane foam has been utilised to increase surface area for biofilm growth, reported specific surface areas range from 600-1000 m^2m^{-3} can provide greater solids retention, however careful management of biofilm thickness and pore clogging is required (Windey et al. 2005; Tawfik et al. 2010). Chen et al. (2006) used a ‘net-like’ media which increased the surface area of flat discs to facilitate a nitrification rate of 0.6 $g.N.m^{-2}d^{-1}$ (Table 1). Lui et al. (2008), utilised a pyridinium type polymer sprayed to a non-woven carrier. They demonstrated the feasibility of autotrophic anaerobic denitrification. It was claimed that surface properties of the pyridinium facilitated attachment of nitrifiers permitting a nitritification rate >26 $kgm^{-2}d^{-1}$. Whereas Hassard et al. (2014) studied the impact of OLR on removal rates of biofilm cultivated on a polyvinylcholoride-like mesh, polyester and polyurethane foam in RBC-like reactors operated concurrently. They identified that under high loading conditions macroscale media pore size was the most significant parameter governing

performance. As pore clogging leads to biomass inactivation and a decrease in effective surface area due to mass transfer restrictions.

Research suggests a link between the initial adhering community, subsequent established biofilm and reactor performance (Stephenson et al. 2013). Different media physicochemical properties have been suggested to select for different bacterial groups. For example media hydrophobicity influences adhesion, due to the difference in size of the aqueous boundary layer preventing bacterial contact. This effect is reduced in more hydrophobic materials, promoting adhesion (Khan et al. 2013). The surface roughness can impact bacterial adhesion. Singh et al. (2011) found that media with a roughness of >20 nm has high protein adsorption, which increases the effective media hydrophilicity, increasing the water layer and preventing adsorption of bacteria (Singh et al. 2011). In contrast a media of intermediate roughness gains the surface area benefit and high bacterial adsorption. Hassard et al. (2014), studied this phenomena using bench scale RBC-like reactors at OLR from 16-160 $g.sCOD.m^{-2}d^{-1}$ and identified that media with an average roughness <20 nm had 4.5 times more biomass on average compared to similar media with an average roughness of 35 nm. However the removal rates of sCOD were similar suggesting the high roughness biofilm had greater specific bacterial activity.

2.6. Scale up considerations

Appropriate scale up of RBCs is critical to validate whether performance will be comparable from bench/pilot to full scale (Arvin and Harremoës, 1990).For RBCs, scale up should incorporate parameters of hydrodynamics, media active surface area, flow, organic loading, oxygen transfer, bacterial growth rate, biofilm accumulation and detachment. However most models only accommodate one of these variables. For example Wilson et al. (1980) developed 'generalised design loadings' based on 12 months data at different scales. However, resulting models failed to consider operating/environmental conditions or process understanding (Harremoës and Gönenc, 1983).The use of tip speed is rarely a suitable parameter - as it increases (along with shear forces and mixer power) to the square of the diameter. To simulate full scale, bench scale reactors were previously operated at higher rotational speeds (to keep constant tip speed) which decreased the contact time per rotation (Spengel and Dzombak, 1992). This also resulted in different shear distributions influencing erosion and sloughing processes in the biofilm, greater mixing and improve substrate removal. The empirical approach to scale-up involves constructing reactors of different sizes and is popular but is generally expensive. After sufficient development a mechanistic model can be developed, reducing the need for extensive testing. However these models are usually appropriate only for identical operating and wastewater conditions. Dutta et al. (2010) constructed three different sized RBCs to characterise the oxygen transfer coefficient at different scales. The model was based around existing ones: the Activated Sludge Model No. 3 for biochemical reactions, a multiculture biofilm model and an RBC model. However the main limitation for this approach is that oxygen transfer should be suitably characterised on scale up, which is rarely the case. Alternatively, design such that large reactors have chemical, dynamic, geometric and kinematic homogeneity to bench scale trials (Spengel and Dzombak, 1992).The appreciation of scale up in RBCs is far from complete however models which based on fundamentals are less sensitive to scale up than empirical design parameters.

3. Microbiology of RBCs

The microbiology of RBC systems is governed by influent substrate conditions, seed population and hydrodynamic conditions. The biofilm which grows on RBC media is reliant on initial adhesion and the formation of glucoconjugate extracellular polymeric substance (EPS) matrix for stability (Möhle et al. 2007). The most influential variable to the microbiology of RBCs is the mass transfer of compounds, which is dependent on operational parameters, biofilm structure and attachment/detachment mechanisms, and boundary layer thickness which have profound impact on the chemistry and microbial community structure, function and activity (Wuertz et al. 2004).

3.1. Structure

The growth rate and yield govern the spatial location of groups within multispecies RBC biofilms (Wuertz et al. 2004). Organisms with the highest maximum specific growth rate will be located towards the outside of the biofilm whereas slower growing organisms will be located towards the inside (Okabe et al. 1996). Ouyang (1980) reported an RBC biofilm with 74% VS, 95% water content and a chemical composition of $C_{4.2}H_8N_{0.6}O_2$. However RBC biofilm communities also exhibit distinct three dimensional organisation, for example Zahid and Ganczarczyk (1994) found that early RBC biofilms are characterised by numerous fine pores, whereas mature biofilms have few large pores. This could reflect biofilm community regulation by quorum sensing (Strous et al. 1999). Pores influence the convective flow and diffusive mass transport within the biofilm itself. De la Rosa and Yu (2005) found that a mature RBC biofilm had highly heterogeneous surface DO concentration from 3.8 to 0 mgL^{-1} which suggested that the biofilm oxygen consumption exceeded the rate of mass transfer through the boundary layer. However, they identified pockets of high DO ($>1 \mu gL^{-1}$) at depths of 760 µm, which is attributed to convective water flow through pores within the biofilm (Zahid and Ganczarczyk, 1994). The surface microbiota will be exposed to shear forces and the biofilm as an entity is subject to erosion. It is important to minimise mass sloughing events which negatively impact biofilm sludge retention time and process performance can ultimately suffer. Biofilm density is important to reduce sloughing frequency. Cell density increased from 3.3 x 10^9 to 3.9 x 10^{10} cells.cm^3 with depth from 0 to 350 µm toward media surface (Okabe et al. 1996). The inner layers are protected from erosion and contain groups with a higher cell density (Arvin and Harremoës 1990). The rate of diffusion decreases with depth into the biofilm due to density, mineral formation and reduced mass driving force (Okabe et al. 1996; Stewart, 2003). Okabe et al. (1996) discovered that increasing the C:N ratio from 0 to 1.5 in an RBC biofilm created a distinct stratification in functional groups, where heterotrophs outcompeted nitrifiers for oxygen and space in the outer layers. Further increases in the carbon ratio decreased nitrification rate and enhanced the biofilm functional stratification. The biofilm thickness also influences the performance of RBC reactors by providing a barrier to mass transfer. Möhle et al. (2007) showed that RBC biofilm thickness increases with substrate concentration and decreases with surface shear forces. The cohesive strength of biofilms on RBC media was identified to be 6.1 and 7.7 Nm^{-2} at a biofilm thickness of 412 and 151 µm respectively, suggesting that biofilm stability is linked to thickness and density. Under high load and or low shear environments filamentous groups proliferate at the surface RBC biofilm boundary. Alleman et al. (1982) showed that a distinct redox layering exists where the *Desulfovibrio sp.* reduce of sulphate to sulphide in the anaerobic sublayer and the *Beggiatoa* species dominate the outer aerobic layer where they oxidise hydrogen sulphide. This was confirmed by Kinner et al. (1985) identified bacteria containing poly-β-hydroxybutyrate and elemental sulphur inclusions. This situation develops under high organic and low oxygen conditions in the biofilm which can result in reductions in RBC performance. Decreased OLR subsequently reduced the dominance of these organisms (Kinner et al. 1985).

3.2. Function and activity

Bacterial presence within an RBC biofilm does not necessitate functional activity. Satoh et al. (2003) studied the influence of bioaugmentation and biostimulation on the efficacy of nitrification by RBC biofilms. Addition of nitrifying bacteria into the RBC resulted in elevated bacteria cell numbers at the surface of the biofilm. This resulted in higher NH_4-N/NO_2-N removal rates and 0.33 and 3 times lower start-up required for AOBs and NOBs respectively compared to a control. Kindaichi et al. (2004) showed that a carbon limited RBC biofilm was comprised of 50% nitrifying bacteria composed of AOBs and NOBs consuming the influent ammonia and nitrite products respectively. However the remaining 50% were heterotrophic bacteria consuming soluble microbial products (SMPs) for nourishment from biofilm endogenous decay. A diverse heterotrophic community was present but sometimes inactive, however the majority of the carbohydrate and protein utilisation was by bacteria undertaking endogenous decay. Okabe et al. (2005) demonstrated that under substrate limitation the *Chloroflexi* group utilised ^{14}C labelled products derived from RBC biofilm endogenous decay. In contrast the *Cytophaga-Flavobacterium* group accumulated ^{14}C

labelled reaction products from nitrifying growth, which suggested that each group specialised in utilising products from different biofilm growth phases. Heterotrophic turnover of utilisation and biomass decay products formed an equal contribution to the cell number and a greater contribution to the total diversity within a nitrifying RBC biofilm suggesting a role in community regulation. Kulikowska et al. (2010) demonstrated that an integral RBCs can remove up to 99% of faecal coliforms from the influent. Tawfik et al. (2004) suggested that adsorption to the RBC biofilm could be a major mechanism for the removal of *Escherichia coli* although grazing by higher organisms or sedimentation could also contribute to pathogen removal in RBCs. Further research is warranted on the mechanisms of initial adhesion and bacterial incorporation in RBCs.

4. Biological nutrient removal in RBCs

4.1. Nitrification

Rotating biological contactors are used for nitrification and denitrification of a range of influent conditions (Cortez et al. 2007, De Clippeleir et al. 2011). Stringent rules govern nitrogen discharge and the energy cost/greenhouse gas emissions are a growing concern (Ainger et al. 2009). The RBC has potential benefits by reducing tank volume, HRT and aeration demand coupled with nitrogen removal at greater loadings compared to traditional treatments. Furthermore RBCs have been applied for refractory or contaminated wastes. For example Kulikowska et al. (2010) achieved a maximum nitrification rate of 4.8 $g.NH_4\text{-}N.m^2d^{-1}$ at a loading of 6.6 $g.NH_4\text{-}N.m^2d^{-1}$ (Table 1). Sequence analysis revealed microbial diversity decreased with time, suggesting a climax community was attained. Diversity indices were resistant to shock loading of >70% of normal flow and fluctuating performance, suggesting more sensitive measures of community change are required.

4.2. Denitrification

Denitrification is the dissimilarly reduction of nitrate to nitrite to dinitrogen gas under anoxic conditions (Paredes et al. 2007). Conventional heterotrophic denitrification is possible in wastewaters with a C/N ratio >2.5, without additional carbon sources (Hippen et al. 2001). As DO is consumed within a biofilm the community becomes oxygen limited. Thereby facilitating microenvironments where each consortia can develop. Helmer and Kunst, (1998) found that under low DO conditions RBCs can remove up to 90% of the nitrogen load from landfill leachate. Odegaard and Rusten (1980) found that the NO_x-N recycle ratio in RBCs improved denitrification rate. Batch testing revealed that nitrogen removal was carbon limited, suggesting autotrophic degradation satisfied the nitrogen deficit. Cortez et al. (2011a.) achieved almost complete nitrogen removal from landfill leachate using conventional denitrification in an anoxic RBC, they identified that preozonation was required to remove refractory carbon compounds. Gupta and Gupta (2001) augmented a myxotroph known as *Paracoccus denitrificans* to undertake simultaneous aerobic carbon oxidation, nitrification and denitrification. *P. denitrificans* removed a maximum of 26 and 1.9 $gm^{-2}d^{-1}$ of COD and nitrogen respectively in an RBC. However, the aerobic denitrification rate was slower than conventional denitrification. At high nitrate concentrations (>500 $mgL^{-}1$) inorganic phosphorus can limit denitrification. Cortez et al. (2011b) suggested that phosphorus improves overall biofilm denitrifying activity and nitrogen removal by promoting bacterial growth. Teixeira and Oliveira (2000) improved denitrification by 30% upon the addition of phosphorus. Hanhan et al. (2005) compared the nitrogen removal rates in full scale pre-denitrifying RBCs. The highest reported removal was ~2 $g.N.m^{-2}d^{-1}$ with a HRT of 0.2 d (Table 1). The nitrogen removal rate decreased with increasing rotational speed, suggesting oxygen inhibition led to suppression of the denitrification pathway. Teixeira and Oliveira (2001) demonstrated that increased disc submergence from 64.5 to 100% improved the TN removal by 63% but had delayed start-up.

The RBC is suitable for autotrophic denitrification as the anammox bacteria have low growth rates and therefore require reactors with a high MCRT (Siegrist et al. 1998). Initially the thin RBC biofilm is conductive for AOBs to proliferate and provide the colonisation matrix for slow growing

anammox bacteria; providing the biofilm is oxygen limited or NOBs are supressed (Pynaert et al. 2004). De Clippeleir et al. (2011) showed that decreasing HRT from 0.66 to 0.18 d stimulated a decrease in removal rate from 2.2 to 1.6 g.N $m^{-2}d^{-1}$ (Table 1). This was attributed to increased nitratation by *Nitrospira* sp. which proliferated at DO concentrations of >1.2 mgL^{-1}. Stepwise loading increases allowed removal rates in excess of 1.8 g.N $m^{-2}d^{-1}$ (Pynaert et al. 2004). Vlaeminck et al. (2009) tested the feasibility of an oxygen limited autotrophic nitrification and denitrification (OLAND) process to treat digestate from source separated black water and achieved a removal rate of 0.71 g.N $m^{-2}d^{-1}$. The nitrite oxidising bacteria were supressed at free ammonia levels >3 mgL^{-1}, however, DO levels <0.3 mgL^{-1} are required for process stability. The effluent from this reactor had a N/P ratio of 1 suggesting struvite production and therefore nutrient recovery is possible. However facilitating struvite accumulating organisms in RBC biofilms has not received any attention. Windey et al. (2005) showed that anammox bacteria could adapt to high salinity conditions of up to 30 gL^{-1}, providing the RBC biofilm acclimation was gradual. The removal rate of nitrogen decreased from 11.9 gL^{-1} using non-saline wastewater to 11.5, 9.6 and 9.6 at 5,10 and 30 gL^{-1} of salt respectively. A similar study by Kartal; et al. (2006), identified that 45 gL^{-1} of salt completely inhibited anammox bacteria. Liu et al. (2008) suggested that the anammox consortium on RBCs were relatively resistant to DO shocks. They found that a *Nitrosomonas eutropha*-like species protected the *Planctomycetes* by sequestrating potentially inhibiting DO levels.

Note to publisher: insert table 1

4.3. Biological phosphorus removal

Attaining biological phosphorus removal (BPR) is challenging in RBC systems, as it is difficult to control the sequential oxic and anaerobic conditions for growth of phosphorus accumulating organisms (PAO). Kenneth (1999) grew PAOs in a modified RBC setup with an anaerobic clarifier and carbon addition for PAO growth, with subsequent sludge recycle to the RBC. This solids recycle allowed oxygen conditions for enhanced BPR and increased the liquid phase MLSS improving organic removal rates. Simm (1988) varied the submergence in a RBC operated as a sequencing batch contactor. Initially full submergence and acetate addition created anaerobic conditions necessary for phosphorus release and fatty acid storage. Next half of the fluid was stored in a holding tank, the remaining liquid in the RBC was subjected to oxic conditions allowing enhanced phosphorus uptake. Yun et al. (2004) used a sequencing batch reactor (SBR) approach to undertake BPR without an additional carbon source. The authors demonstrated that the maximum biofilm phosphorus uptake was at a C:P range of 13 to 18 where P ranged from 3 to 8% of biofilm VS. The biofilm thickness appeared to determine the TP removal with a maximum removal efficiency of total phosphorus of 70% was attained at a biofilm thickness <1.8 mm. This limitation is not apparent in suspended growth SBR. This could be a mass transfer restriction preventing exchange of available phosphorus and organic substrates restricting TP uptake rate which is not present in suspended growth setups. Understanding mechanisms which govern BPR in RBC biofilms warrants further attention.

5. Priority pollutant remediation in RBCs

Priority pollutant remediation can require the bioaugmentation or retention of specialised strains. Bioaugmentation in RBC systems is usually achieved through addition of either suspended or freeze dried artificial cultures or freeze dried biomass to the RBC (Stephenson and Stephenson, 1992). Alternatively cultures of microbes can be grown in a side stream reactor prior to addition. The natural solids retention of the RBC biofilm permits microbe retention without additional separation or recirculation. Many systems require acclimatisation periods and are sensitive to shock/variable loadings or intermittent feeding of the pollutant which is of import for the removal of priority substances from wastewaters (Stephenson and Stephenson, 1992; Duque et al. 2011; Amorim et al. 2013).

5.1. Organic pollutants

Dye wastewater is a challenging form of organic pollutant as the dyes or breakdown products can be toxic or mutagenic (Malachova et al. 2013). The RBC is ideal for dye treatment due to high biomass retention, low startup costs, and appropriate technology level for developing countries (Robinson et al. 2001). Wastewater dyes are initially absorbed to the biofilm but a continually exposed biomass will eventually saturate. Most dyes do not penetrate bacteria as they have a high molecular weight and contain hydrophobic groups, which are a barrier to biocenocis (Pearce et al. 2003). The bioaugmentation of white rot funghi (WRF) e.g. *Phanerochaete* sp. has been undertaken in RBC systems as they excrete non-specific extracellular hydrolytic enzymes with dye decolouring capacity (Pakshirajan and Kheria, 2012). Novotný et al. (2011) found a surface decolourisation rate of 0.63, 0.19 and 0.01 $mg.m^{-2}h^{-1}$ for Remazol Brilliant Blue R, Methylene Blue and Azure B respectively by the augmented fungus *Dichomitus squalens* (Table 2). Dye degradation is often undertaken as a secondary metabolism so allochthonous carbon sources are required to maintain activity. Novotný et al. (2011) identified that *D. squalens* has a minimum glucose concentration of 0.018 gL^{-1} for effective dye decolourisation. Pakshirajan and Kheria, (2012) showed that the decolourisation rate of WRF *P. chrysosporium* is proportional to glucose concentrations to a limit of 10 gL^{-1}. The use of molasses dosing decreased the decolourisation rate of *P. chrysosporium* by 20% compared to glucose control (Pakshirajan and Kheria, 2012). Dye removal has been correlated with activity of manganese dependent peroxidase and lignin peroxidases. For full dye remediation from wastewater the dye should be decolourised and detoxified. Malachova et al. (2013) utilized an RBC bioaugmented with *Irpex lacteus* 931, and achieved a batch methyl blue decolourisation rate of 9.4 $mgm^{-2}d^{-1}$. Decolourisation resulted in reduced toxicity level of the wastewater. However the WRF are susceptible to bacterial stress which usually prevents application under real wastewater conditions. Nilsson et al. (2006) used an RBC augmented with *Trametes versicolor* to treat real textile wastewater and achieved 60-70% decolourisation efficiency. Research should identify if WRF can be utilized in RBCs with appropriate scale up.

Note to publisher insert table 2.

Duque et al. (2011) inoculated a strain capable of degrading 2-fluorophenol and demonstrated increased removal efficiency under constant pollutant loading. Under variable loading the pollutant removal decreased even though the community remained in the biofilm. Amorim et al. (2013) studied the impact of shock loadings of 4-fluorocinnamic acid (4-FCA) on an augmented RBC. The removal efficiency was increased from 8 to 46% at surface loadings of 73 to 168 $g.m^{-2}.d^{-1}$ respectively (Table 3). Isolation of biofilm strains and batch testing revealed that two strains completely mineralised 4-FCA. Sequence analysis revealed a 97% similarity to the original augmented *Rhodococcus* strain, suggesting horizontal gene transfer or genetic drift had occurred (Singh et al. 2006).. The RBC reactor has also been applied for removal of non-aqueous phase liquids (NAPL) (Mukherji and Chavan 2012). Chavan and Mukherji (2008.b) found that a mixed freshwater phototrophic community augmented with *Burkholderia cepacia* had a removal rate of $>26\ gm^{-2}d^{-1}$ for removal of diesel NAPL. The NAPL component of the wastewater was likely sorbed onto the biofilm for subsequent biodegradation of the aliphatic fraction (Mukherji and Chavan (2012). Operation with the co-contaminant phenol slightly reduced the removal efficacy of NAPL but resulted in complete phenol removal (Chavan and Mukherji, 2010). Under constant pollutant loading in RBCs it is therefore important to promote proliferation of the augmented community at functional levels.

Note to publisher insert table 3

In WWTPs micropollutants are usually eliminated through biotic degradation or abiotic sorption. Simonich et al. (2002) compared removal of fragrances in different WWTPs. Fragrances appeared to be removed typically in the biodegradable fraction of the wastewater. However sorptive non-biodegradable fragrance material removal is linked to solids disposal (Simonich et al. 2002). In

contrast micropollutants which are non-sorptive and non-readily biodegradable are of greatest concern. In this study the RBC achieved 99% removal efficiency of methyl dihydrojasmonate compared to 98, 93, 82% for an ASP, trickling filter and carousel setup respectively. The removal of 6-Acetyl-1,1,2,4,4,7-hexamethylteraline (AHTN) in the RBC was inferior compared to other secondary treatments which could be due to poor removal of particulate matter. Batt et al. (2007) compared four treatment works with similar influent concentrations of Ciprofloxacin (CP), Sulfamethoxazole (SM), Tetracycline (TC) and Trimethoprim (TM) and found that the RBC had comparable removal of antibiotics of between 52-95% removal of CP, TC and TM to an extended aeration ASP but with lower HRT and presumably treatment cost. In contrast the RBC demonstrated 43% lower SM removal compared to the ASP. The degradation behavior of this antibiotic could be due to physical differences between bacteria in biofilms and suspended growth..

5.2. Inorganic pollutants

Biological heavy metal removal relies on both the sorption of the metal species to biomass and the bioaccumulation by metabolic processes (Costley and Wallis, 2001). The RBC microbial biofilm is suitable for biosorption as there is a high contact area for sorption and a long MCRT. However the metal removal rate will decrease with time, as the attraction sites become saturated (Matheikal et al. 1991). For example Sirianuntapiboon and Chumlaong (2013) found that an RBC had a decreased removal efficiency of 64-45 and 80-85% with increased loading which corresponded to a removal rate of between 255-400 and 255-480 $mg.m^{-2}d^{-1}$ for Ni and Pb respectively (Table 4). This is similar to removal rates reported for Cu of ~450 $mg.m^{-2}d^{-1}$ using activated sludge consortia (Costley and Wallis, 2001) (Table 4). To prevent saturation it is necessary to remove the metal loaded biomass by suitable treatment. However this is costly and produces secondary waste issues (Costley and Wallis 2001). Costley and Wallis, (2001) showed that multiple cycles of sorption/desorption, using a dilute (<0.5 M) acid did not impact the adsorption efficiency of a mixed culture RBC biofilm, suggesting reuse was possible. The removal rates demonstrated by Costley and Wallis (2001) of ~ 640, 450 and 320 $mg.m^{-2}d^{-1}$ for Zn, Cu and Cd appeared dependent on loading and the availability of free sorption sites. Regression analysis reveals that the loading rate predicts removal rate between loads of 0.003-762.8 $mg.metal.m^{-2}d^{-1}$ ($R^2 = 0.9$, $P<0.001$) (Table 4).

Note to publisher: insert table 4

6. Modelling of RBC reactors

Process optimization and scale-up are challenges for the efficient use of RBCs (Spengel and Dzombak, 1992; Dutta et al. 2010). In contrast to most suspended growth processes, mass transfer can often mask the impact of biokinetics on the performance of RBCs (Famularo et al. 1978). This is because thick biofilms and unidirectional transfer limit the rate of compound exchange. Previously, the derivations of mass transfer were described within the context of penetration and surface renewal theory (Patwardhan et al. 2003). Then focus was placed on the relationship between oxygen transfer and substrate utilization biokinetics (Chavan and Mukherji, 2008). However usage of empirical approaches are limited to wastewater and operational conditions similar to the derivative source of the models (Di Palma et al. 2003). Models can also be based on reaction order, substrate diffusion, microbial growth biokinetics and the identification of different oxygen and nutrient conditions (Clarke et al. 1978; Patwardhan, 2003). Finally, multiple substrate and species models have been applied to RBCs using biofilm models based on description of transformation and transport processes (Gujer and Boller, 1990; Dutta et al. 2007). Historically RBC modeling has received significant research attention; however the inherent complexities of system hydrodynamics prevent application to other biological treatment processes.

6.1. Substrate utilization in RBCs

The substrate utilization in RBCs is separated into substrates and electron acceptors, model assumptions and output. Roberts (1973) developed a model incorporating substrate mass transfer

limitation to/from the biofilm and the kinetic considerations governing biodegradable substrate utilization. Alternatively the removal of soluble substances is determined by the boundary layer diffusion resistance, into the biofilm prior to microbial degradation within the interior (Arvin and Harremoës, 1990). An empirical relationship to predict effluent BOD_5 was determined by the US Environmental Protection Agency (Brenner and Opaken 1984) in which:

$$\frac{C_e}{C_i} = e^{K(0.000125\,V/Q)^{0.5}} \quad (1)$$

In which:

K= reaction rate constant (0.3) at 13°C.

V= media volume (m^3)

Q = hydraulic loading (Ls^{-1})

This model does not include parameters on microbial kinetics, substrate limitation or changes to influent / temperature. Clark et al. (1978) developed an RBC model where removal rate can be determined from influent/effluent conditions and microbial growth rate in which:

$$r_a = \left(\frac{\mu_{max}}{X_a}\right)/Y_a \quad (2)$$

A modified version of the Kincannon and Stover (1982) model for RBC systems of removal rate integrated over disc area in which:

$$r_a = \left(\frac{K_C}{U_{max}}\right).\left(\frac{A_d}{QC_i}\right) + \left(\frac{1}{\mu_{max}}\right) \quad (3)$$

The equations mentioned above are empirical or analytical in origin which predict removal rate per area as a function of a chosen suite of dependent variables. The removal rate constants and model coefficients are obtained by regression analysis with experimental data (Hansford et al. 1978). However providing the system has been adequately described more complex models allow application to different treatment scenarios (Wanner et al. 2006). An RBC model was one of the first to describe simultaneous BOD removal and nitrification. It was suggested that heterotrophic activity is the dominant process at earlier stages in RBC treatment and nitrification occurs once the BOD concentration is below the threshold selecting against autotrophic nitrification (Mueller et al. 1978). Wanner and Gujer (1984) demonstrated that competition for space and electron acceptors between heterotrophs and autotrophs occurs in biofilms. Biofilm modeling was previously based on Fick's Law of diffusion, however, Wanner and Gujer (1986) also accounted for biofilm behavior and internal microbial distribution in a dynamic model. This allowed the application of a modified version of Activated Sludge Model (ASM) 1 to permit true dynamic modeling of RBCs for aerobic and anoxic degradation of organic constituents (Gujer and Boller, 1990). The model revealed that the distal compartment of the RBC was substrate limited for nitrification, in which decay and inactivation outweighed growth (Dutta et al. 2007). This identified a potential risk to effluent quality under shock load scenarios. Model simulations demonstrated that periodic flow reversal restored the activity to the distal compartment by countering nitrifyer starvation. Dutta et al. (2007) developed a model incorporating the multi-species biofilm model after Gujer and Boller (1990) and the kinetics from the ASM 3 (Gujer et al. 1999) in which:

$$\frac{dC^{Lf}}{dt} = K_L^{air}\frac{A_{exp}}{V_{Lf}}(C^* - C^{Lf}) + K_L\frac{A_{sub}}{V_{Lf}}(C^T - C^{Lf}) - K_L\frac{A}{V_{Lf}}\left(C^{Lf} - C^{Bf}\Big|_{x=\delta_{Bf}}\right) \quad (4)$$

The terms on the right hand side describe the transfer from the air, from/to the tank and from the liquid film to the biofilm for each substrate/electron acceptor respectively. The model was implemented on a three stage RBC and calibrated using oxygen transfer data. Increased effluent

recycle rate from 0.25-2.0 improved the rate of nitrification in the first stage of an RBC due to dilution of influent BOD (Dutta et al. 2007). This model has the potential to describe biofilm development with multiple bacterial groups and removal rate of their substrates and electron acceptors. The hydrodynamics should be characterized and the model calibrated for oxygen transfer prior to application, the inherent complexity limits the application to experienced modelers.

6.2. Oxygen transfer in RBCs

The oxygen transfer rate (OTR) determines the biofilm oxygen concentration and hence the selected removal regime in RBCs. Initially, oxygen must diffuse from the bulk water/gas phase across the boundary layer, into the film layer and eventually into the biofilm itself. The rate of diffusion is dependent on the diffusion coefficient of oxygen and the distance according to Fick's Law (Stewart, 2003). Originally it was thought that the majority of transfer occurs with biofilm contact to the air phase and therefore bulk fluid concentration was less important (Hartman, 1960). Other models were developed with the assumption that substrate alone rather than oxygen is limiting in RBCs: these are now deemed unsuitable (Clark et al. 1978; Spengel and Dzombak, 1992). The OTR is related to the difference between the liquid phase and equilibrium concentration, in the liquid film and RBC biofilm (Chavan and Mukherji 2008.a). Hansford et al. (1978), presented one of the first attempts to include mass transport resistance to OTR. Initial models of OTR assumed that turbulence, wave generation and immersion dominate (Patwardhan, 2003). An alternative method is that oxygenation occurs during film breakup and renewal. This is caused by the air/water cycling involving the interaction with rotational derived forces, which overcome film layer surface tension. The rate of renewal is dependent on the rotational speed, disc diameter, position and half spacing (Table 5) (Chavan and Mukherji, 2008.a). A study suggested that the relationship between liquid film renewal and the OTR was linear under sterile conditions (Kim and Molof, 1982). Attached biofilm increases the OTR, by enhancing concentration gradients due to consumption in the film and adsorption to the biofilm (Kim and Molof, 1982; Zeevalkink et al. 1979). However biofilm growth can reduce OTR by clogging pores which reduces mass transfer, Friedman et al. (1979) related oxygen transfer coefficient to rpm alone. Rittmann et al. (1983), identified the import of adsorption for OT at high rotational speed (>25) whereas diffusive film transport dominated during operation at normal rotor speed. Kubsad et al. (2004), compared two forms of the Kim and Molof (1982) model to alternatives and found appropriate predictive fit providing the volume renewal number can be estimated effectively (Table 5). Di Palma et al. (2009) calibrated a previously defined model and found that the k_La increased in a linear fashion between the speeds of 3 and 10 rpm at bench scale. The majority of film renewal is thought to occur when the surface tension resistance is broken under the effect of gravity after the so called 'falling film' theory (Zhang et al. 2009).

Note to publisher: insert table 5

7. Novel applications of RBCs

The relatively simple engineering of RBC type systems promises to provide a platform for new energy generating processes that treat wastewater. There are a variety of RBC systems that have been applied for direct electricity generation or energy production through biogas and algae (Sayess et al. 2013; Cheng et al. 2011; Paule et al. 2011). Sayess et al. (2013) coupled an RBC with a microbial fuel cell configuration which allowed for contaminant removal and electricity production. This RBC achieved between 6.9 and 20.9% higher denitrification rates compared to a control RBC setup where electron generation by anodic oxidation was used by denitrifiers for nitrate reduction at the cathode. In a similar system it was shown that the optimum current for nitrogen removal is 0.2 $Amps.m^{-2}$ (Rodziewicz et al. 2011). Cheng et al. (2011), developed a bioelectrochemical RBC-type system for indirect energy generation. Each disc was split with regular 180° rotations which led to inversion of the anode and cathode allowing concurrent spatial acetate oxidation and methanogenesis respectively. Methane generation appeared proportional to electrical input with 80% energy recovery. Christenson and Sims (2012) developed a method for indirect energy generation

and removal of nitrogen and phosphorus utilising an algal RBC-type reactor. The reactor design consisted of a RBC drum with ropes and scraper blades which collected the algae. The maximum harvested biomass produced was ~30 $gm^{-2}d^{-1}$ of total solids. The algal RBC reactor achieved removal rates of ~14 and 2 $gm^{-2}d^{-1}$ of soluble nitrogen and phosphorus respectively. Paule et al. (2011) designed a vertical RBC with an intrinsic light source with removable polyethylene plates produced 0.007 $gm^{-2}d^{-1}$ of volatile solids which could be used for energy generation.

8. Conclusions

The use of RBCs for conventional biological wastewater treatment to remove BOD_5 and ammonia has been well established for the last three decades (Mueller et al. 1978). Application has largely been at the lower end of the WWT scale, usually for up to 2000 P.E. (Griffin and Findlay 2000). The limits of organic carbon renewal have been thoroughly investigated, with maximum OLRs of up to 120 g. $sCOD.m^{-2}d^{-1}$ through using improved media optimised disc immersion and adjusted rotational speeds (Teixeira and Oliveira 2001; Hanhan et al. 2005; Chen et al. 2008; Hassard et al. 2014). However, novel configurations of media – such as mesh types (Chen et al. 2008; Lui et al. 2008; Hassard et al. 2014) – and careful selection of media to enhance growth of certain bacterial populations could increase applied OLRs and nitrogen loading rates (NLRs) incrementally (Khan et al. 2012; Stephenson et al. 2013).

Recent research has demonstrated that the process can be adapted to remove nutrients, both nitrogen and phosphorus, as with other biological processes (Yun et al. 2004; Hahnhan et al. 2005). Novel RBC type processes, such as Hybrid Activated Sludge (HYBACS), has shown that new combinations of suspended growth and fixed film on rotating media can provide higher organic removal rates and efficient denitrification (Hoyland et al. 2008). Solid and liquid phase bacterial interactions have been mentioned previously (Wanner and Kos 1990; Kenneth 1999), a better understanding of these mechanisms merit further investigation in applying hybrid RBCs to energy efficient nutrient removal. Biofilm systems are suited to providing a range of redox environments, from wholly aerobic through anoxic to anaerobic conditions (Wuertz et al. 2004). Exploitation of this phenomenon in RBCs is in its infancy at full-scale: for example, anammox (Strous et al. 1999) has been demonstrated in RBCs (Siegrist et al. 1998; Vlaeminck et al. 2009; De Clippeleir et al. 2011). Control of disc immersion can be used to stimulate denitrification (Courtens et al. 2014). Enhanced BPR requires alternating anaerobic and aerobic conditions (Yun et al. 2004); however enforcing SBR type approaches in RBCs at full scale is challenging. Therefore manipulation of the gaseous headspace, submergence, rotational speed or recycle in RBCs could be explored to stimulate the enhanced BPR process.

Fully submerged processes such as Biological Aerated Filters use backwashing to remove excess bacterial growth to optimise performance, drawing analogies to mixed liquor wastage in activated sludge (Mendoza Espinosa and Stephenson 1999). Deliberate removal of RBC biofilm, either by mechanical means or air scouring, to control the biomass growth rate, and therefore performance, has not been directly employed. A full scale exception is the air scour used to remove biofilm in rotating biofilm SMART reactors, however, this is usually applied to prevent media clogging (Hoyland et al. 2008). Yun et al. (2004) suggested biofilm thickness should be controlled every 15 days to enable BPR in a SBR type RBC, although this would be dependent on biofilm accumulation rate. Christenson and Sims (2012) used scraper blades to remove algal biofilm for harvesting providing new surfaces for biomass growth. Manipulating microbial growth rate to determine performance could allow greater process control of RBCsThe mechanical engineering of RBCs has proven to be the most problematic issue when applied at full scale, specifically shaft material selection, media robustness and construction and design and maintenance of bearings (Mba et al. 1999). ‘Lightweighting’ of these components through use of new materials, e.g. composites, provide opportunities for re-engineering and allowing further scale-up. Application of low resistance bearings, e.g. air or ‘non-stick’ bearings, may allow for lower energy, higher rotational speeds that

could enhance treatment. The removal of dyes and other recalcitrant organic pollutants in RBCs appears linked to bioaugmentation and propagation of allochthonous microbial populations with pollutant degrading capacity (Novotny et al. 2012). The sensitivities and expense of these communities remains an issue for application under real scenarios with representative wastewater. Future research should focus on approaches suitable for scale-up or methods for upgrade or existing works which struggle to deal with organic pollutants containing wastewater. Costley and Wallis (2001) highlighted the potential of RBC biofilms for resource recovery, with the increasing price of metals and nutrient fertilizer new opportunities could be created for cost positive wastewater treatment (STOWA, 2012). The simplicity, adaptability, low land use and maintenance and high volumetric activity of the RBC suggest that it will continue to help meet our wastewater treatment requirements for years to come.

9. Acknowledgements

This work was funded by Bluewater Bio Ltd and the UK Engineering and Physical Sciences Research Council (EPSRC), through a Doctoral Training Award to Francis Hassard.

10. References

Ainger, C., Butler, D., Caffor, I., Crawford-Brown, D., Helm, D., Stephenson, T., 2009. A Low Carbon Water Industry in 2050 - Report: SC070010/R3. Bristol: Environment Agency.

Alleman, J., Veil, J., Canaday, J., 1982. Scanning electron microscope evaluation of rotating biological contactor biofilm. Water Res. 16, 543–550.

Allen, K., 1929. Sewage disposal bulletin, City of New York 14.

Amorim, C.L., Duque, A.F., M. Afonso, C.M., L. Castro, P.M., 2013. Bioaugmentation for treating transient 4-Fluorocinnamic acid shock loads in a rotating biological contactor. Bioresourc. Technol. 144, 554-562.

Andreadakis, A. 1987. Design of Multistage Rotating Biological Contactors. J. Environ. Eng., 113:1, 199–205.

Arvin, E., Harremoës, P., 1990. Concepts and models for biofilm reactor performance. Water Sci. & Technol. 22, 171–192.

Axelsson, J., Nilsson, U., Terrazas, E., Alvarez Aliaga, T., Welander, U., 2006. Decolorization of the textile dyes Reactive Red 2 and Reactive Blue 4 using *Bjerkandera* sp. Strain BOL 13 in a continuous rotating biological contactor reactor. Enzyme Microb. Technol. 39, 32–37.

Ayoub, G.M., Saikaly, P., 2004. The combined effect of step-feed and recycling on RBC performance. Water Res. 38, 3009–3016.

Batchelor, S.E., Cooper, M., Chhabra, S.R., Glover, L.A., Stewart, G.S., Williams, P., Prosser, J.I., 1997. Cell density-regulated recovery of starved biofilm populations of ammonia-oxidizing bacteria. Appl. Environ. Microbiol. 63, 2281–2286.

Bintanja, H.M., Vandererve, J.J.V.M., Boelhouwer, C., 1975.Oxygen transfer in a rotating disc treatment plant. Water Res. 9, 1147–1153.

763 Bollmann, A., Schmidt, I., Saunders, A.M., Nicolaisen, M.H., Bollmann, A., Schmidt, I., Saunders,
764 A.M., Nicolaisen, M.H., 2005. Influence of Starvation on Potential Ammonia-Oxidizing
765 Activity and amoA mRNA Levels of *Nitrosospira briensis.* Appl. Environ. Microbiol 63,
766 2281–2286.

767 Brar, S.K., Gupta, S., 2000. Biodegradation of trichloroethylene in a rotating biological contactor.
768 Water Res. 34, 4207–4214.

769 Brenner R.C. and Opaken A.J., 1984. Design Information on rotating biological contactor design
770 Technical Report No EPA-600/ 2-84-106 Municipal Environmental Research Laboratory,
771 Cincinnati, Ohio USA.

772 Chan, R., Stenstrom, M., 1981. Use of the rotating biological contactor for appropriate technology
773 wastewater treatment. Research Leaflet Series, University of California 1–70.

774 Chavan, A., Mukherji, S., 2008.a. Treatment of hydrocarbon-rich wastewater using oil degrading
775 bacteria and phototrophic microorganisms in rotating biological contactor: effect of N:P ratio.
776 J. Hazard. Mater. 154, 63–72.

777 Chavan, A., Mukherji, S., 2008.b. Dimensional analysis for modelling oxygen transfer in rotating
778 biological contactor. Bioresourc. Technol. 99, 3721–8.

779 Chavan, A., Mukherji, S., 2010. Effect of co-contaminant phenol on performance of a laboratory-
780 scale RBC with algal-bacterial biofilm treating petroleum hydrocarbon-rich wastewater. J.
781 Chem. Technol. Biotechnol. 85, 851–859.

782 Chen, Z., Wen, Q., Wang, J., Li, F., 2006. Simultaneous removal of carbon and nitrogen from
783 municipal-type synthetic wastewater using net-like rotating biological contactor (NRBC).
784 Process Biochem. 41, 2468–2472.

785 Cheng, K.Y., Ho, G., Cord-Ruwisch, R., 2011. Novel methanogenic rotatable bioelectrochemical
786 system operated with polarity inversion. Environ. Sci. Technol. 45, 796–802.

787 Christenson, L.B., Sims, R.C., 2012. Rotating algal biofilm reactor and spool harvester for
788 wastewater treatment with biofuels by-products. Biotechnol. Bioeng. 109, 1674–84.

789 Clark, J. H., Moseng, E. M. and Asano, T., 1978. Performance of a rotating biological contractor
790 under varying wastewater flow. J. Water Poll. Control Fed. 50, 896-911.

791 Confer, D., Logan, B., 1998. A conceptual model describing macromolecule degradation by
792 suspended cultures and biofilms. Water Sci. Technol. 37, 231–234.

793 Cortez, S., Teixeira, P., Oliveira, R., Mota, M., 2008. Rotating biological contactors: a review on
794 main factors affecting performance. Rev. Environ. Sci. Biotechnol.7, 155–172.

795 Cortez, S., Teixeira, P., Oliveira, R., Mota, M., 2011a. Mature landfill leachate treatment by
796 denitrification and ozonation. Process Biochem. 46, 148–153.

797 Cortez, S., Teixeira, P., Oliveira, R., Mota, M., 2011b. Denitrification of a landfill leachate with
798 high nitrate concentration in an anoxic rotating biological contactor. Biodegradation 22, 661–
799 71.

800 Costley, S.C., Wallis, F.M., 2001. Bioremediation of heavy metals in a synthetic wastewater using a
801 rotating biological contactor. Water Res 35, 3715–23.

802 Courtens, E.N.P., Boon, N., De Clippeleir, H., Berckmoes, K., Mosquera, M., Seuntjens, D.,
803 Vlaeminck, S.E., 2014. Control of nitration in an oxygen-limited autotrophic
804 nitrification/denitrification rotating biological contactor through disc immersion level variation.
805 Bioresourc. Technol. 155, 182-188.

806 De Clippeleir, H., Yan, X., Verstraete, W., Vlaeminck, S.E., 2011. OLAND is feasible to treat
807 sewage-like nitrogen concentrations at low hydraulic residence times. Appl. Environ.
808 Microbiol. 90, 1537–45.

809 De la Rosa, C., Yu, T., 2005. Three-dimensional mapping of oxygen distribution in wastewater
810 biofilms using an automation system and microelectrodes. Environ. Sci. Technol. 39, 5196–
811 202.

812 Di Palma, L., Merli, C., Paris, M., Petrucci, E., 2003. A steady-state model for the evaluation of disk
813 rotational speed influence on RBC kinetic: model presentation. Bioresourc. Technol. 86, 193–
814 200.

815 Di Palma, L., Verdone, N., 2009. The effect of disk rotational speed on oxygen transfer in rotating
816 biological contactors. Bioresourc. Technol. 100, 1467–70.

817 Doman, J., 1929. Results of operation of experiment contact filter with partially submerged plates.
818 Sewage Works Journal 1(5), 555–560.

819 Duque, A.F., Bessa, V.S., Carvalho, M.F., Castro, P.M.L., 2011. Bioaugmentation of a rotating
820 biological contactor for degradation of 2-fluorophenol. Bioresourc. Technol. 102, 9300–9303.

821 Dutta, S., Hoffmann, E., Hahn, H.H., 2007. Study of rotating biological contactor performance in
822 wastewater treatment using multi-culture biofilm model. Water Sci. and Technol. 55, 345-353.

823 Famularo, J., Muller, J.A., Mulligan T., 1978. Application of mass transfer to rotating biological
824 contactors. J. Water Pollut. Control Fed. 46, 653–671.

825 Findlay, G., Heist, J., 2003. Resolving the reliability issues of Rotating Biological Contractors. Proc.
826 Water Environ. Fed., 6, 142-161.

827 Fountoulakis, M.S., Terzakis, S., Chatzinotas, A, Brix, H., Kalogerakis, N., Manios, T., 2009. Pilot-
828 scale comparison of constructed wetlands operated under high hydraulic loading rates and
829 attached biofilm reactors for domestic wastewater treatment. Sci. Total Environ. 407, 2996–
830 3003.

831 Friedman, A.A., Robbins, L.E., Woods, R.C., 1979. Effect of disc rotational speed on biological
832 contactor efficiency. J.Water Pollut. Control Fed. 51, 2678–2690.

833 Gabet-Giraud, V., Miège, C., Choubert, J.M., Ruel, S.M., Coquery, M., 2010. Occurrence and
834 removal of estrogens and beta blockers by various processes in wastewater treatment plants. Sci
835 Total Environ. 408, 4257–69.

836 Ge, Y., Yan, L., Qinge, K., 2004. Effect of environment factors on dye decolorization by
837 *Phanerochaete sordida* ATCC90872 in an aerated reactor. Process Biochem. 39, 1401–1405.

838 Griffin, P., Findlay, G., 2000. Process and engineering improvements to rotating biological
839 contactor design. Water sci. and technol. 41, 137–144.

840 Gujer, W., Boiler, M., 1990. A mathematical model for rotating biological contactors. Water Sci.
841 Technol. 22, 53–73.

842 Gupta, A., and Gupta, S., 2001. Simultaneous carbon and nitrogen removal from high strength
843 domestic wastewater in an aerobic RBC biofilm. Water Res. 35, 1714–1722.

844 Hanhan, O., Orhon, D., Krauth, K., Günder, B., 2005. Evaluation of denitrification potential of
845 rotating biological contactors for treatment of municipal wastewater. Water sci. and Technol.
846 51, 131–9.

847 Hansford, G.S., Andrews, J.F., Grieves, C.G., Carr A.D., 1978. A steady-state model for the rotating
848 biological disc reactor. Water Res. 12, 855–868.

849 Harremoës, P. and Gönenc, I.E., 1983. The applicability of biofilm kinetics to rotating biological
850 contactors Paper presented at International EWPCA-IAWPRC Seminar, Fellbach, Germany.

851 Hartman, H., 1960 (Translation): Investigation on the biological treatment of waste water by using
852 drip-body immersion systems. Stuttgart reports for Urban Water Management Commission
853 Publisher, Munich, Germany.

854 Hartmann, H., 1961 Improvements in or relating to sewage plant, British patent no. 935162, T963.

855 Hassard. F., Cartmell, E., Biddle, J., Stepehnson T., 2014. Performance of permeable media rotating
856 reactors used for pretreatment of wastewaters. Water Sci. and Technol. 69, 1926-31.

857 Helmer, C., Kunst, S., 1998. Simultaneous nitrification/denitrification in an aerobic biofilm system.
858 Water Sci. and Technol. 37, 183-187.

859 Hippen, A, Helmer, C., Kunst, S., Rosenwinkel, K.H., Seyfried, C.F., 2001. Six years' practical
860 experience with aerobic/anoxic deammonification in biofilm systems. Water Sci. and Technol.
861 44, 39–48.

862 Hiras, D.N., Manariotis, I.D., Grigoropoulos, S.G., 2004. Organic and nitrogen removal in a two-
863 stage rotating biological contactor treating municipal wastewater. Bioresourc. Technol.. 93, 91–
864 98.

865 Hoyland, G., Vale, P., Rogalla, F., and Jones, M., 2010. A new approach to nutrient removal using
866 the HYBACS process. Proc. Water Environ. Fed., Presented at WEF/IWA Biofilm Reactor
867 Technology Conference, Portland, Oregon USA, 14: 81-94.

868 Jeswani, H., Mukherji, S., 2012. Degradation of phenolics, nitrogen-heterocyclics and polynuclear
869 aromatic hydrocarbons in a rotating biological contactor. Bioresourc. Technol..111, 12–20.

870 Kartal, B., Koleva, M., Arsov, R., van der Star, W., Jetten, M.S.M., Strous, M., 2006. Adaptation of
871 a freshwater anammox population to high salinity wastewater. J. Biotechnol. 126, 546–53.

872 Kenneth, E.N., 1999. Cost-effective upgrading of RBC plants to maximise performance and
873 improve effluent quality including biological nutrient removal. Proc. of IAWQ Conference on
874 Biofilm Systems, New York, USA.

Khan, M., Chapman, T., Cochran, K., Schuler, A., 2013. Attachment surface energy effects on nitrification and estrogen removal rates by biofilms for improved wastewater treatment. Water Res. 47, 2190-2198.

Kim, B., Molof, A., 1982. The scale-up and limitation of physical oxygen transfer in rotating biological contractors. Water Sci. Technol. 14, 569–579.

Kincannon D.F and Stover E. L. 1982 Design methodology for fixed film reaction RBCs and biological towers. In Civil Engineering for Practicing and Design Engineers, Vol. 2, pp.107-124.

Kinner, N.E., Maratea D., Bishop, P.L., 1985. An electron microscopic evaluation of bacteria inhabiting biological contactor biofilms during various loading conditions. Environ. Technol. Lett. 6, 455-466.

Kindaichi, T., Ito, T., Okabe, S., 2004. Ecophisiological interaction between nitrifying bacteria and heterotrophic bacteria in autotrophic nitrifying biofilms as determined by microautoradiography-fluorescence in situ hybridization. Applied and Environmental Microbiology 70, 1641–1650.

Kindaichi, T., Tsushima, I., Ogasawara, Y., Shimokawa, M., Ozaki, N., Satoh, H., Okabe, S., 2007. In situ activity and spatial organization of anaerobic ammonium-oxidizing (anammox) bacteria in biofilms. Applied and Environmental Microbiology 73, 4931–9.

Kubsad, V., Chaudhari, S., Gupta, S.K., 2004. Model for oxygen transfer in rotating biological contactor. Water Res. 38, 4297–304.

Kulikowska, D., Jóźwiak, T., Kowal, P., Ciesielski, S., 2010. Municipal landfill leachate nitrification in RBC biofilm - process efficiency and molecular analysis of microbial structure. Bioresour. Technol. 101, 3400–5.

Labella, S.A., Thaker, I.H., Tehan, J.E., 1972. Treatment of winery wastes by aerated lagoon, activated sludge, and rotating biological contactor. Proceedings of the 27th Industrial Waste Conference 41, 803–816.

Laopaiboon, L., Phukoetphim, N., Vichitphan, K., Laopaiboon, P., 2008. Biodegradation of an aldehyde biocide in rotating biological contactors. World J. Microbiol. Biotechnol. 24, 1633–1641.

Leenen, E.J.T.M., dos Santos, V.A.P., Grolle, K.C.F., Tramper, J., Wijffels, R.H., 1996. Characteristics of and selection criteria for support materials for cell immobilization in wastewater treatment. Water. Res., 30, 2985–2996.

Liu, S., Yang, F., Xue, Y., Gong, Z., Chen, H., Wang, T., & Su, Z. (2008). Evaluation of oxygen adaptation and identification of functional bacteria composition for anammox consortium in non-woven biological rotating contactor. Bioresourc. Technol., 99, 8273–9.

Lv, Y., Wang, L., Wang, X., Yang, Y., Wang, Z., Li, J., 2011. Macroscale and microscale analysis of Anammox in anaerobic rotating biological contactor. J. Environ. Sci. 23, 1679–1683.

Matheikal, J., Iyengar, L. and Venkobachar, C.1991 Sorption and desorption of Cu(II) by *Ganodeerma lucidium.*Water Pollution Research Journal Canada 26, 187–200.

914 Mathure, P., Patwardhan, A., 2005. Comparison of mass transfer efficiency in horizontal rotating
915 packed beds and rotating biological contactors. J. Chem. Technol. Biotechnol. 80, 413–419.

916 Mba, D., Bannister, R., Findlay, G., 1999. Mechanical redesign of the rotating biological contactor.
917 Water Res. 33, 3679–3688.

918 Möhle, R., Langemann, T., 2007. Structure and shear strength of microbial biofilms as determined
919 with confocal laser scanning microscopy and fluid dynamic gauging using a novel rotating disc
920 biofilm. Biotechnol, and Bioeng. 98, 747–755.

921 Mueller, J.A., Paquin, P., Famularo, J., 1978. Nitrification in rotating biological contactors. In: 51st
922 Annual Conference WPCF, Anaheim, CA.

923 Mukherji, S., Chavan, A., 2012. Treatment of aqueous effluents containing non-aqueous phase
924 liquids in rotating biological contactor with algal bacterial biofilm. Chem. Eng. J. 200-202, 459–
925 470.Najafpour, G.D., Zinatizadeh, A.A.L, Lee, L.K., 2006. Performance of a three-stage aerobic
926 RBC reactor in food canning wastewater treatment. Biochem. Eng. J. 30, 297–302.

927 Nilsson, I., Möller, a., Mattiasson, B., Rubindamayugi, M.S.T., Welander, U., 2006. Decolorization
928 of synthetic and real textile wastewater by the use of white-rot fungi. Enzyme Microb. Technol.
929 38, 94–100.

930 Novotný, C., Trošt, N., Šlušla, M., Svobodová, K., Mikesková, H., Válková, H., Malachová, K.,
931 Pavko, A., 2012. The use of the fungus *Dichomitus squalens* for degradation in rotating
932 biological contactor conditions. Bioresour. Technol. 114, 241–6.

933 Odegaard H., Rusten B., 1980. Nitrogen removal in rotating biological contactors without the use of
934 external carbon source. Proceedings of the First national symposium/workshop on rotating
935 biological contactor technology, Champion, PA, USA, Vol. II, 1301–1317.

936 Okabe, S., Hiratia, K., Ozawa, Y., Watanabe, Y., 1996. Spatial microbial distribution of nitrifiers
937 and heterotrophs in mixed-population biofilms. Biotechnol. and Bioeng. 50, 24–35

938 Okabe, S., Satoh, H., & Watanabe, Y. 1999. In situ analysis of nitrifying biofilms as determined by
939 in situ hybridization and the use of microelectrodes. Appl. and Environ. Microbiol., *65*(7),
940 3182–91.

941 Okabe, S., Kindaichi, T., Ito, T., 2005. Fate of 14C-Labeled Microbial Products Derived from
942 Nitrifying Bacteria in Autotrophic Nitrifying Biofilms Fate of C-Labeled Microbial Products
943 Derived from Nitrifying Bacteria in Autotrophic Nitrifying Biofilms. Appl. and Environ.
944 Microbiol. 71, 3987–3994.

945 Orandi, S., Lewis, D.M., Moheimani, N.R., 2012. Biofilm establishment and heavy metal removal
946 capacity of an indigenous mining algal-microbial consortium in a photo-rotating biological
947 contactor. J. Ind. Microbiol. and Biotechnol. 39, 1321–31.

948 Ouyang, C.F., 1980. The characteristics of rotating biological contactor sludge. Proceedings of the
949 First national symposium/workshop on rotating biological contactor technology, Champion,
950 PA, USA, Vol. II, 189.

951 Pakshirajan, K., Singh, S., 2010. Decolorization of Synthetic Wastewater Containing Azo Dyes in a
952 Batch-Operated Rotating Biological Contactor Reactor with the Immobilized Fungus
953 *Phanerochaete chrysosporium*. Ind. Eng. Chem. Res. 49, 7484–7487.

954 Pakshirajan, K., Kheria, S., 2012. Continuous treatment of coloured industry wastewater using
955 immobilized *Phanerochaete chrysosporium* in a rotating biological contactor reactor. J.
956 Environ. Manage. 101, 118–23.

957 Paredes, D., Kuschk, P., Mbwette, T.S.A., Stange, F., Müller, R. a., Köser, H., 2007. New Aspects
958 of Microbial Nitrogen Transformations in the Context of Wastewater Treatment –Rev. Eng.
959 Life Sci. 7, 13–25.

960 Patwardhan, A.W., 2003. Rotating Biological Contactors: A Review. Ind. Eng. Chem. Res. 42,
961 2035–2051.

962 Paule, a, Lauga, B., Ten-Hage, L., Morchain, J., Duran, R., Paul, E., Rols, J.L., 2011. A
963 photosynthetic rotating annular bioreactor (Taylor-Couette type flow) for phototrophic biofilm
964 cultures. Water Res. 45, 6107–18.

965 Pearce, C., Lloyd, J., Guthrie, J., 2003. The removal of colour from textile wastewater using whole
966 bacterial cells: a review. Dye Pigment 58, 179–196.

967 Popel 1964 (Translation: English) Assembly, dismantling, performance and design of rotating
968 biological contactor, Switzerland Hyfol.

969 Pynaert, K., Wyffels, S., Sprengers, R., Boeckx, P., Van Cleemput, O., Verstraete, W., 2002.
970 Oxygen-limited nitrogen removal in a lab-scale rotating biological contactor treating an
971 ammonium-rich wastewater. Water Sci. Technol. 45, 357–63.

972 Pynaert, K., Smets, B., Wyffels, S., 2003. Characterization of an autotrophic nitrogen-removing
973 biofilm from a highly loaded lab-scale rotating biological contactor. Appl. Environ. Microbiol.
974 69, 3626–3635.

975 Pynaert, K., Smets, B.F., Beheydt, D., Verstraete, W., 2004. Start-up of autotrophic nitrogen
976 removal reactors via sequential biocatalyst addition. Environ. Sci. Technol. 38, 1228–35.

977 Ramsay, J., Shin, M., Wong, S., Goode, C., 2006. Amaranth decoloration by *Trametes versicolor* in
978 a rotating biological contacting reactor. J. Ind. Microbiol. Biotechnol. 33, 791–5.

979 Rittmann, B. E.; McCarty, P. L. 2001, Environmental Biotechnology: Principles and Applications,
980 4th ed.; McGraw-Hill Higher Education: New York.

981 Roberts, J., 1973. Towards a better understanding of high rate biological film flow reactor theory.
982 Water Res. 7, 1561–1588.

983 Robinson, T., McMullan, G., Marchant, R., Nigam, P., 2001. Remediation of dyes in textile effluent:
984 a critical review on current treatment technologies with a proposed alternative. Bioresour.
985 Technol. 77, 247–55.

986 Rodziewicz, J., Filipkowska, U., Dziadkiewicz, E., 2011. Electrolytically aided denitrification on a
987 rotating biological contactor. Environ. Technol. 32, 93–102.

Ross, B., Lange, C., Barrett, R., 1994. Inordinate failures of rotating biological contactor (RBC) drive shafts in wastewater treatment plant service worldwide. Int. J. Press. Vessel. Pip. 59, 197–209.

Sahinkaya, E., Dilek, F.B., 2006. Biodegradation of 4-CP and 2,4-DCP mixture in a rotating biological contactor (RBC). Biochem. Eng. J. 31, 141–147.

Sant' Anna, J.R., 1980. Contribution à l'étude de l'hydrodynamique des réacteurs biologiques utilisés en traitement des eaux usées These, INSA Touslouse (France) (model form cited from Kubsad et al., 2004).

Sarayu, K., Sandhya, S., 2012. Rotating biological contactor reactor with biofilm promoting mats for treatment of benzene and xylene containing wastewater. Appl. Biochem. Biotechnol. 168, 1928–37.

Satoh, H., Okabe, S., Yamaguchi, Y., Watanabe, Y., 2003. Evaluation of the impact of bioaugmentation and biostimulation by in situ hybridization and microelectrode. Water Res 37, 2206–16.

Sayess, R.R., Saikaly, P.E., El-Fadel, M., Li, D., Semerjian, L., 2013. Reactor performance in terms of COD and nitrogen removal and bacterial community structure of a three-stage rotating bioelectrochemical contactor. Water Res. 47, 881–94.

Schmidt, I., Sliekers, O., Schmid, M., Bock, E., Fuerst, J., Kuenen, J.G., Jetten, M.S.M., Strous, M., 2003. New concepts of microbial treatment processes for the nitrogen removal in wastewater. FEMS Microbiol Rev. 27, 481–492.

Siegrist, H., Reithaar, S., Koch, G., Lais, P., 1998. Nitrogen loss in a nitrifying rotating contactor treating ammonium-rich wastewater without organic carbon. Water Sci. and Technol. 38, 241-248.

Sim, R., 1988. Enhanced Biological Phosphorus Removal Using a Sequencing Batch RBC. Master's thesis, University of British Columbia.

Simonich, S.L., Federle, T.W., Eckhoff, W.S., Rottiers, A., Webb, S., Sabaliunas, D., De Wolf, W., 2002. Removal of fragrance materials during U.S. and European wastewater treatment. Environ. Sci. and Technol. 36, 2839–47.

Singh, R., Paul, D., Jain, R.K., 2006. Biofilms: implications in bioremediation. Trends Microbiol. 14, 389–97.

Singh, A.V., Vyas, V., Patil, R., Sharma, V., Scopelliti, P.E., Bongiorno, G., Podestà, A., Lenardi, C., Gade, W.N., Milani, P., 2011. Quantitative characterization of the influence of the nanoscale morphology of nanostructured surfaces on bacterial adhesion and biofilm formation. PLoS One 6, e25029.

Singh, V., Mittal, A. K., 2012. Characterization of biofilm of a rotating biological contactor treating synthetic wastewater. Water Sci. and Technol. 66, 429–37.

Sirianuntapiboon, S., Chumlaong, S., 2013. Effect of Ni_2^+ and Pb_2^+ on the efficiency of packed cage rotating biological contactor system. Journ. Environ. Chem. Eng. 2–9.

Spengel, D., Dzombak, D., 1992. Biokinetic modeling and scale-up considerations for rotating biological contactors. Water Environ. Res. 64, 223–235.

Steiner, C., 1997. The biological approach to the rotating disc process. WSE publication no. 695 1–5.

Stephenson, D., Stephenson, T., 1992. Bioaugmentation for enhancing biological wastewater treatment. Biotechnol. Adv. 10, 549–559.

Stephenson, T., Reid, E., Avery, L.M., Jefferson, B., 2013. Media surface properties and the development of nitrifying biofilms in mixed cultures for wastewater treatment. Process Saf. Environ. Prot. 91, 321–324.

Stewart, P., 2003. Diffusion in biofilms. J. Bacteriol. 185, 1485–1491.

Stewart, P.S., Franklin, M.J. 2008. Physiological heterogeneity in biofilms Nat. Rev.Microbiol. 6, 199–210.

STOWA, 2010. STOWA News: The Dutch roadmap for the WWTP of 2030, STOWA report 2010-24.

Strous, M., Fuerst, J., Kramer, E., 1999. Missing lithotroph identified as new planctomycete. Nat. 400, 446-449.

Tawfik, A., Klapwijk, B., el-Gohary, F., Lettinga, G., 2002. Treatment of anaerobically pre-treated domestic sewage by a rotating biological contactor. Water Res. 36, 147–55.

Tawfik, A., Klapwijk, B., Van Buuren, J., El Gohary, F., & Lettinga, G. (2004). Physico-chemical factors affecting the *E. coli* removal in a rotating biological contactor (RBC) treating UASB effluent. Water Res., 38, 1081-1088.

Tawfik, a, Klapwijk, a, 2010. Polyurethane rotating disc system for post-treatment of anaerobically pre-treated sewage. J. Environ. Manage. 91, 1183–92.

Teixeira, P., Oliveira, R., 2000. Denitrification by *Alcaligenes denitrificans* in a closed rotating biological contactor. Biotechnol. Letters 1789–1792.

Teixeira, P., Oliveira, R., 2001. Denitrification in a closed rotating biological contactor: effect of disk submergence. Proc. Biochem. 37, 345–349.

Upton, J.E., Green, M.B. and Findley, G., 1995. Sewage treatment for small communities: the Severn Trent approach. Water and Environ. J. 9, 64–71.

Vesilind, A., 2003. In: Vesilind, P. (Ed.), Wastewater Treatment Plant Design IWA Publishing, London, Great Britain.

Vlaeminck, S.E., Terada, A., Smets, B.F., Van der Linden, D., Boon, N., Verstraete, W., Carballa, M., 2009. Nitrogen removal from digested black water by one-stage partial nitritation and anammox. Environ. Sci. and Technol. 43, 5035–41.

Wanner, O., Ebert, H.J., Morgenroth, E., Noguera, D., Picioreanu, C., Rittmann, B.E., Van Loosdrecht, M.C.M., 2006. Mathematical modeling of biofilms. IWA Scientific and Technical Report No. 18, IWA Task Group on Biofilm Modelling.

Ware, A., Pescod, M., Storch, B., 1990. Evaluation of alternatives to conventional disc support media for rotating biological contactors. Water Sci. Technol. 22, 113–117.

Weston R F, 1985 Review of Current RBC Performance and Design Procedures United States Technical Information Report PB85-180945 / AS.

Wilson, R.W., Murphy K.L and Stephenson J.P., 1980. Scaleup in rotating biological contactor design.Water Poll. Cont. Fed. 52, 3, 610-621.

Windey, K., De Bo, I., Verstraete, W., 2005. Oxygen-limited autotrophic nitrification-denitrification (OLAND) in a rotating biological contactor treating high-salinity wastewater. Water Res. 39, 4512–20.

Wuertz, S., Okabe, S., Hausner, M., 2004. Microbial communities and their interactions in biofilm systems: an overview. Water Sci. and Technol. 49, 327–36.

Wyffels, S., Pynaert, K., Boeckx, P., Verstraete, W., Van Cleemput, O., 2001. Identification and quantification of nitrogen removal in a rotating biological contactor by 15N tracer techniques. Water Res. 37,1252-1259.

Yun, Z., Lee, H., Choi, E., 2004. Enhanced biological phosphorus removal in RBC with SBR modification. Water Sci. Technol. 50, 121–30.

Zahid, W., Ganczarczyk, J., 1994. Fractal properties of the RBC biofilm structure. Water Sci. and Technol. 29, 271–279.

Zhang, H., Chen, G., Yue, J., Yuan, Q., 2009. Hydrodynamics and mass transfer of gas–liquid flow in a falling film microreactor. AIChE J. 55, 1110-1120.

Zheng, Z., Obbard, J.P., 2002. Removal of surfactant solubilized polycyclic aromatic hydrocarbons by *Phanerochaete chrysosporium* in a rotating biological contactor reactor. J. Biotechnol. 96, 241–9.

Zhevalkink, J., Kelderman, P., Boelhouwer, C., 1978. Liquid film thickness in a rotating disc gas-liquid contactor. Water Res. 12, 1–5.

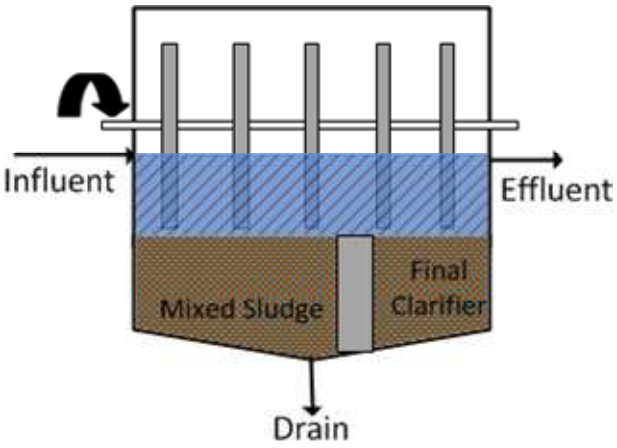
Zeevalkink, J., Kelderman, P., Visser, D., Boelhouwer, C., 1979. Physical mass transfer in a rotating disc gas-liquid contactor. Water Res. 13, 913–919.

1090

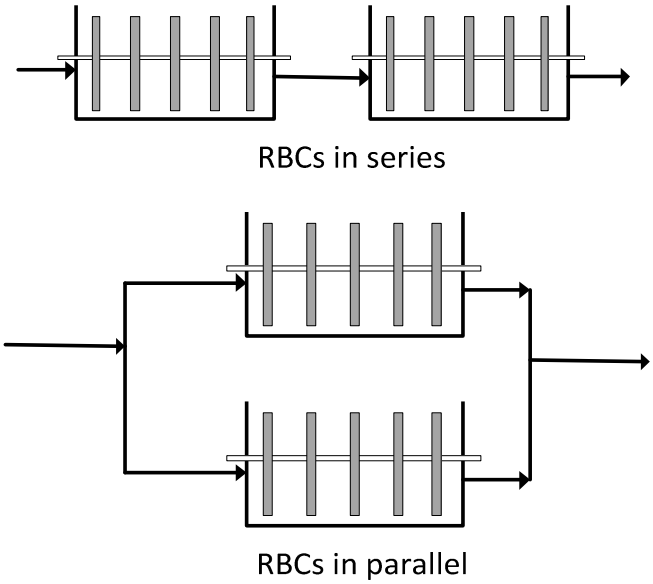
Figure

Rotating fixed film biological contactors for wastewater treatment – a review - figures

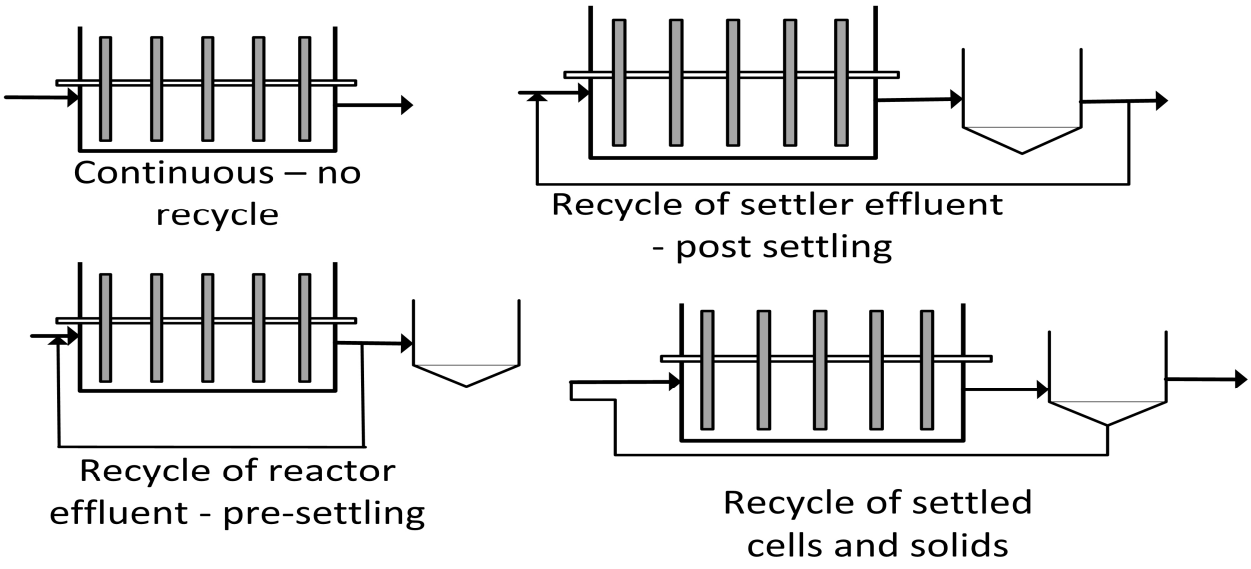
1.A - Integral RBC

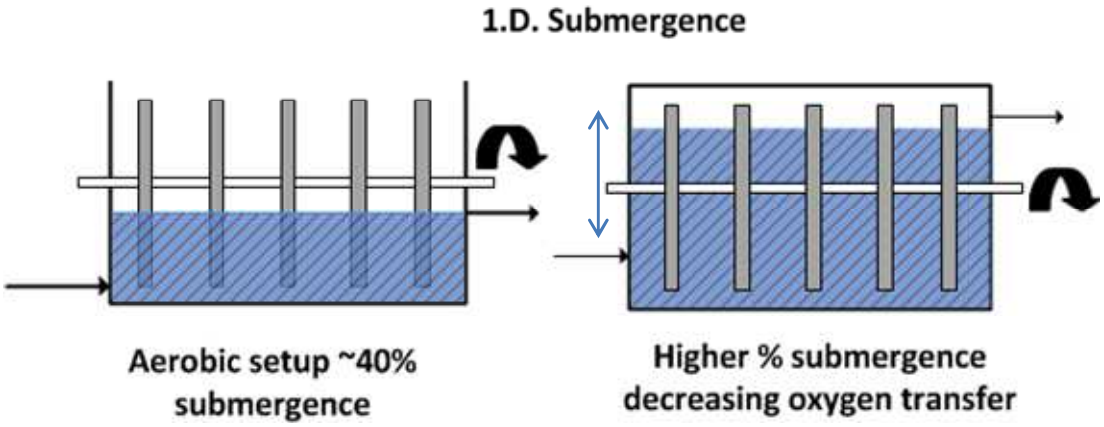


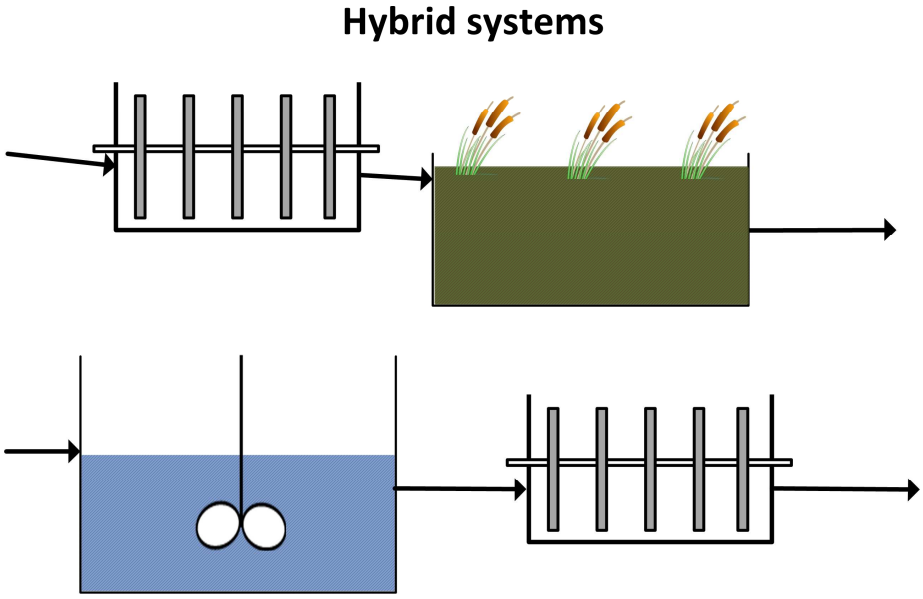
1.B – RBC flowsheet layout



1.C Reactor and recycle arrangements







1.E. Hybrid rbc/wetland operation 1.F Polishing step post AS tank.

Figure 1: Process configurations of RBC technology.

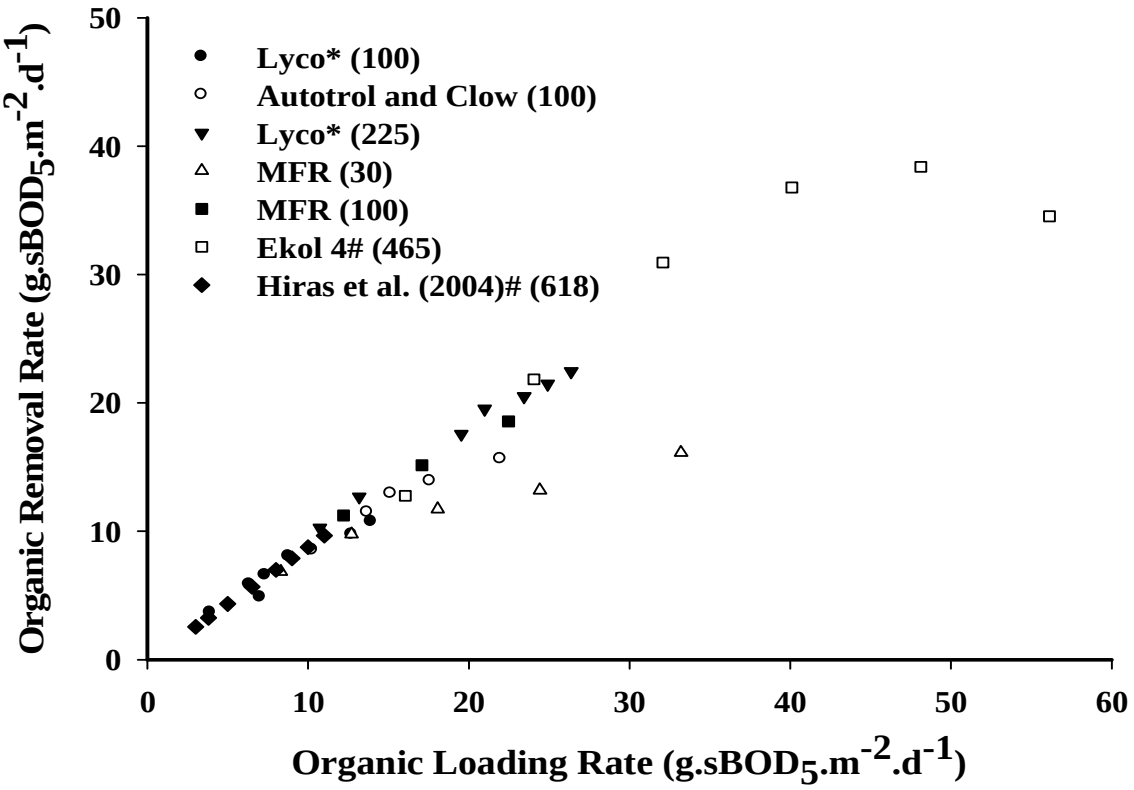
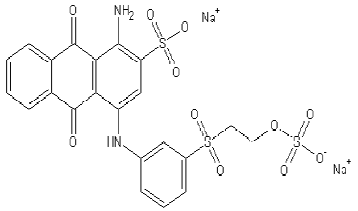
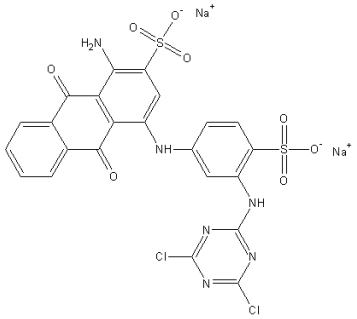
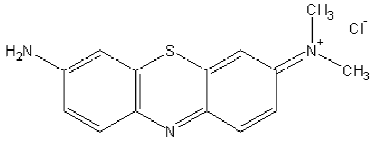
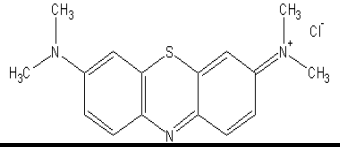


Figure 2: Organic removal rate with loading rate of RBCs from different manufacturers for soluble BOD, * total BOD, # total COD, numbers in brackets indicates influent concentration mgL^{-1} of (Brenner and Opaken 1984), Ekol 4 data adapted from Fountoulakis et al. (2009), data from Hiras et al. (2004) is an unspecified media/manufacturer.

Table 1 – Impact of N loading rate on NH_4-N removal

Wastewater	Influent N concentration	N loading rate ($g.N\ m^{-2}d^{-1}$)	N reaction rate ($g.N\ m^{-2}d^{-1}$)	HRT (day)	Process	Reference
Synthetic high nitrogen	450	5.7	4.8	1.70	Anammox	Wyffels et al. (2003)
Synthetic sewage like nitrogen	280	5.4	3.5	1.00	Anammox	Lv et al. (2011)
Saline high NH4+-N	770	12.9	11.9	0.77	OLAND	Windey et al. (2005)
Synthetic high nitrogen	750	9.6	6.4	0.38	OLAND	Pynaert et al. (2004)
	1300	16.7	14.4	0.38		
Synthetic high nitrogen	1150	11.5	10.3	0.70	OLAND	Pynaert et al. (2003)
Synthetic high nitrogen	400	1.7	1.6	1.70	OLAND	Pynaert et al. (2002)
Digested black water	537	2.2	2.2	0.66		
Synthetic high nitrogen	278	2.2	2.0	0.34	OLAND	De Clippeleir et al. (2011)
	146	2.2	1.6	0.18		
	66	2.1	1.9	0.08		
Digested black water	1023	0.9	0.71	1.14	OLAND	Vlaeminck et al. (2009)
Landfill leachate	209	0.4	0.67	0.55	OLAND	Hippen et al. (1997)
Synthetic high nitrogen	60	0.5	0.5	0.20	Nitrification	Jang et al. (2005)
Digested real sewage	43	1.9	1.6	10.00	Nitrification	Tawfik et al. (2002)
		3.8	2.9	5.00		
		7.6	1.5	2.50		
Landfill leachate	130	1.9	1.9	6.6	Nitrification	Kulikowska et al. (2010)
	244	3.6	3.6			
	332	4.8	3.6			
	451	6.6	4.8			
Real settled sewage	24	3.5	0.2	0.16	Nitrification	Hassard et al. (2014)
	36	10.3	6.3	0.08		
Synthetic high nitrogen	110	1.1	1.1	0.63	SND	Gupta and Gupta (2001)
Synthetic sewage	30	0.7	0.6	0.33	SND	Chen et al. (2006)
Real settled sewage	42	0.06	0.1	0.25	SND	Hiras et al. (2004)

Table 2 – Relationship between chemical structure and reactivity of decolourisation of dyes by bioaugmentation

Compound	Dye concentration mgL^{-1}	Structure	Medium composition	Organism	Removal (%)	Dye surface loading $mgm^{-2}h^{-1}$	Surface Decolourisation rate $mg.m^{-2}.h^{1}$	Referenc
Remazol Brilliant Blue R	50		Mineral medium 10 gL^{-1} glucose	*Dichomitus squalens*	95	0.66	0.63	Novotný al. 2011
Reactive Blue 4	200		Citric buffer 21.4 gL^{-1} glucose	*Trametes versicolor*	70	0.26	0.18	Nilsson e al. 2006
Reactive Blue 4	100		10 gL^{-1} glucose	*Bjerkandera sp.*	99	0.07	0.06	Axelsson al. 2006
Methylene Blue	50		Mineral medium 10 gL^{-1} glucose	*Dichomitus squalens*	85	0.22	0.19	Novotný al. 2011
	150		Malt extract glucose 10 gL^{-1} + 2% agar	*Irpex lacteus 931*	55	N/A	0.39	Malachov et al. 201
Azure B	50		Mineral medium 10 gL^{-1} glucose	*Dichomitus squalens*	42	0.03	0.01	Novotný al. 2011

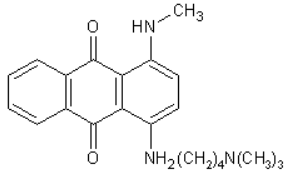
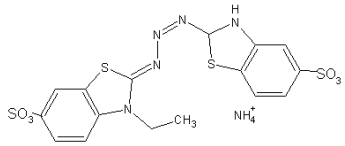
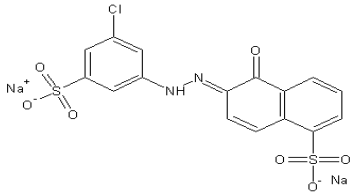
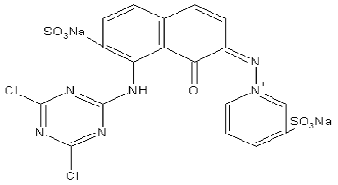
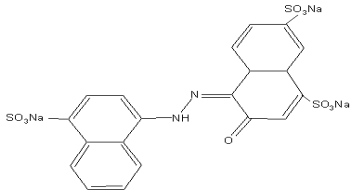
Compound	Dye concentration mgL^{-1}	Structure	Growth substrate supplied	Organism	Removal (%)	Dye surface loading $mgm^{-2}h^{-1}$	Surface Decolourisation rate $mgm^{-2}h^{1}$	Reference
Basic Blue 22	200		8 gL^{-1} glucose	*Phanerochaete sordida*	0.80	0.89	0.71	Ge et al. (2004)
Direct Red 80	200		13.46 gL^{-1} glucose	*Phanerochaete chrysosporium*	0.80	0.55	0.44	Pakshirajan and Singh 2010
Mordant Blue 9	200		13.46 gL^{-1} glucose	*Phanerochaete chrysosporium*	0.62	0.55	0.34	Pakshirajan and Singh 2010
Reactive Red 2	100		10 gL^{-1} glucose	*Bjerkandera sp.*	0.99	0.07	0.06	Axelsson et al. 2006
Acid Red 27	62.4		Kirk's medium 10.1 gL^{-1}	*Trametes versicolor*	0.58	1.64	0.96	Ramsay et al. 2006

Table 3 – Priority pollutant removal by RBC reactor communities

Organic pollutant	Initial pollutant concentration mgL^{-1}	Degrading species / consortia	Removal efficiency %	Pollutant surface loading rate $mg.pollutant.m^{-2}d^{-1}$	Maximum pollutant surface removal rate $mg.pollutant.m^{-2}d^{-1}$	HRT (d)	References
Benzene	1193	*Pseudomonas* sp., *Bacillus*, *Enterococcus* sp.	97.7	4.0	3.9	1.23	Sarayu and Sandhya 2012
Xylene	1226	*Pseudomonas* sp., *Bacillus*, *Enterococcus* sp.	98.5	4.1	4.1	1.23	Sarayu and Sandhya 2012
Phenol*#	250	*Exiguobacterium aurantiacum*	48.4	154.4	74.7	1.00	Jeswani and Mukherji, 2012
Pyridine*	280	*E. aurantiacum*	34.2	169.5	58.0	0.50	Jeswani and Mukherji, 2012
Quinoline*	280	*E. aurantiacum*	48.9	345.3	168.9	0.50	Jeswani and Mukherji, 2012
Benzene*	200	*E. aurantiacum*	35.0	246.7	86.3	0.50	Jeswani and Mukherji, 2012
Napthalene*	60	*E. aurantiacum*	59.8	36.3	21.7	0.50	Jeswani and Mukherji, 2012
Phenanthrene*	0.5	*E. aurantiacum*	53.2	0.3	0.2	0.50	Jeswani and Mukherji, 2012
Phenanthrene	1.73	*Phanerochaete chrysosporium*	41.0	2.5	1.0	12.16	Zheng and Obbard, (2002)
Fluoranthene*	0.2	*E. aurantiacum*	46.0	0.1	0.1	0.50	Jeswani and Mukherji, 2012

Organic pollutant	Initial pollutant concentration mgL^{-1}	Degrading species / consortia	Removal efficiency %	Pollutant surface loading rate $mg.pollutant.m^{-2}d^{-1}$	Pollutant surface removal rate $mg.pollutant.m^{-2}d^{-1}$	HRT (d)	References
Pyrene*	0.12	*E. aurantiacum*	80.0	0.1	0.1	0.50	Jeswani and Mukherji, 2012
Pyrene	1.23	*P. chrysosporium*	65.9	1.8	1.2	12.07	Zheng and Obbard, (2002)
Benzol(α)pyrene	0.21	*P. chrysosporium*	96.9	0.3	0.3	11.72	Zheng and Obbard, (2002)
Trichloroethylene	30	Mixed culture (MC) augmented with *Thiosphaera pantotropha*	98.7	202.8	200.1	2.00	Brar and Gupta (2000)
2-fluorophenol	100	MC augmented with 2-fluorophenol degrader (FP1)	82.0	4.8	3.9	0.78	Duque et al. (2011)
1,5-pentanedial (Glutaldehyde)	180	MC from RBC treating glutaldehyde and RAS	71.4	31468.5	22455.9	0.03	Laopaiboon et al. (2007)
4-chlorophenol	826	MC from settled sewage	51.3	37545.5	18300.0	0.35	Sahinkaya and Dilek,(2006)
2,4-dichlorophenol	424	MC from settled sewage	50.7	19272.7	9500.0	0.35	Sahinkaya and Dilek,(2006)
4-fluorocinnamic acid	80	*Rhodococcus* sp. S2	45.7	4660.3	2129.7	0.77	Amorim et al. 2013
4-fluorocinnamic acid	35	*Rhodococcus* sp. S2	7.9	2038.8	110	0.77	Amorim et al. 2013

*Mixed synthetic wastewater stream containing multiple organic pollutants, #removal from first stage only

Table 4 –Heavy metals and pollutant sequestration by RBC community

Trace pollutant	Initial metal concentration mgL^{-1}	Biosorbent species	Removal efficiency %	Metal loading rate $mg.metal.m^{-2}d^{-1}$	Metal removal rate $mg.metal.m^{-2}d^{-1}$	Adsorption capacity mg.metal. $biofilm.g^{-1}$	HRT d	References
Mn	45	*Ulothrix sp.*	36.7	18.243	6.695	-	1	Orandi et al. 2012
Co	0.5	*Ulothrix sp.*	5.7	0.203	0.012	-	1	Orandi et al. 2012
Cu	100	*Ulothrix sp.*	38	40.541	15.405	-	1	Orandi et al. 2012
Cu	100	Activated sludge consortium enriched by metal spiking	59	762.829	450.069	4484	1	Costley and Wallis, 2001
Pb	30	Sedimentation tank biomass	80 83 85	600 400 300	480 332 255	-	4 6 8	Sirianuntapiboon and Chumlaong (2013)
Zn	20	*Ulothrix sp.*	29	8.108	2.351	-	1	Orandi et al. 2012
Zn	100	Activated sludge consortium enriched by metal spiking	84	762.829	640.777	3454.1	1	Costley and Wallis, 2001
Se	0.04	*Ulothrix sp.*	35.2	0.016	0.006	-	1	Orandi et al. 2012
Sb	0.007	*Ulothrix sp.*	35.6	0.003	0.001	-	1	Orandi et al. 2012
Ni	3	*Ulothrix sp.*	35.7	1.216	0.434	-	1	Orandi et al. 2013
Ni	30	Sedimentation tank biomass	67 71 74	600 400 300	400 284 222	-	4 6 8	Sirianuntapiboon and Chumlaong (2013)
Cd	100	Activated sludge consortium enriched by metal spiking	42	762.829	320.388	1914.4	1	Costley and Wallis, 2001
Cyanide	40	Sedimentation tank consortium	90	0.408	0.367	-	0.33	Sirianuntapiboon and Chuamkaew, (2007)

Table 5 – Expressions for oxygen transfer in RBCs

Application / Derivation	Expression	Assumptions	Reference
Liquid film (LF) thickness	$\delta = \phi^{0.5}\omega^{1.5}s^{1}$		(Zhevalkink et al., 1978)
Overall oxygen transfer (OT) considering film theory	$K_L = -2(\frac{D_L}{\pi t_R})^{0.5}((1-4.21)\frac{\delta}{D_L T_R})^{0.5}$		Zeevalkink et al. (1979)
Overall OT to bulk	$lnK_L = 1.31\ln\omega + 14.78$	OT governed by disc rotation alone	Friedman et al. (1979)
	$K_L = 2(\frac{D_L}{\pi t_R})^{0.5}$	Where $\delta/D_L t_R \geq 1.7$	
Overall OT considering film theory	$K_L = 2\left(\frac{2\alpha}{\pi^{0.5}}\right).\frac{\delta}{t_R} \sim \frac{\delta}{t_R}$	Where $\delta/D_L t_R < 0.8$	Bintanja et al. (1975)
Overall OT	$K_L\frac{\emptyset}{D_L} = K\left(\frac{\omega'\emptyset^2\rho}{\mu}\right)^{l}\left(\frac{\omega'^2\emptyset}{g}\right)^{m}\left(\frac{\emptyset-\emptyset_0}{\emptyset}\right)^{n}$	K = 1.7, l = 0.8, m = 0.13, n =0.74	Sant' Anna (1980)
Volume renewal number	$K_L a = 0.0011\,(\phi^{0.5}\omega^{1.5}s^{-1})^{0.732}$	Sterile disks, e/r = 0.042 and H/t_R = 0.15	Kim and Molof (1982)
The OT dependence on volume renewal number	$K_L a = 0.0003\left(\frac{NA_d\delta\omega}{V}\right) + 0.0119$	Where N_v: <800	Kubsad et al. (2004)
	$K_L a = 0.0001\left(\frac{NA_d\delta\omega}{V}\right) + 0.1157$	Where Nv >800	
Non-dimensional model of $K_L\alpha$	$\frac{(K_L a\rho A_d)}{\mu} = \left(\frac{\emptyset}{A_d{}^{0.5}}\right)^{\psi}\left(\frac{\rho A_d\omega}{\mu}\right)^{\varepsilon}\left(\frac{A_d}{A_t}\right)^{\theta}\left(\frac{\delta}{V^{0.33}}\right)^{\lambda}$	Ψ=0.327, ε=1.018, θ=0.743, λ= 0.624	Chavan and Mukherji (2008)
Model of Oxygen transfer	$K_L a = \alpha.\omega^{1.5}.\phi^{0.5}.(\beta/\omega + \gamma)$	where α, β, γ are constants that need defining	Di Palma et al. (2003)
Experimentally verified model from above	$K_L a = 134.07.\omega^{1.5}.\phi^{0.5}.(2.15/\omega) + 0.006)$	Model only valid providing enhancement factor is described.	Di Palma et al. (2009)



https://dspace.org/

Rotating biological contactors for wastewater treatment - A review

Hassard, Francis; Biddle, Jeremy R.; Cartmell, Elise; Jefferson, Bruce; Tyrrel, Sean F. ...

2014-08-04
Elsevier
http://dspace.lib.cranfield.ac.uk/handle/1826/9746
Hassard F, Biddle J, Cartmell E, et al., (2015) Rotating biological contactors for wastewater treatment – A review. Process Safety and Environmental Protection, Volume 94, March 2015, pp. 285-306

Downloaded from a DSpace repository. https://dspace.org/