

1 Rotating biological contactors for wastewater treatment – a review

2 Abstract

3 Rotating biological contactors (RBCs) for wastewater treatment began in the 1970's. Removal of
4 organic matter has been targeted within organic loading rates of up to $120 \text{ g.m}^{-2}\text{d}^{-1}$ with an optimum
5 at around $15 \text{ g.m}^{-2}\text{d}^{-1}$ for combined BOD and ammonia removal. Full nitrification is achievable
6 under appropriate process conditions with oxidation rates of up to $6 \text{ g.m}^{-2}\text{d}^{-1}$ reported for municipal
7 wastewater. The RBC process has been adapted for denitrification with reported removal rates of up
8 to $14 \text{ g.m}^{-2}\text{d}^{-1}$ with nitrogen rich wastewaters. Different media types can be used to improve
9 organic/nitrogen loading rates through selecting for different bacterial groups. The RBC has been
10 applied with only limited success for enhanced biological phosphorus removal and attained up to
11 70% total phosphorus removal. Compared to other biofilm processes, RBCs had 35% lower energy
12 costs than trickling filters but higher demand than wetland systems. However, the land footprint for
13 the same treatment is lower than these alternatives. The RBC process has been used for removal of
14 priority pollutants such as pharmaceuticals and personal care products. The RBC system has been
15 shown to eliminate 99% of faecal coliforms and the majority of other wastewater pathogens. Novel
16 RBC reactors include systems for energy generation such as algae, methane production and
17 microbial fuel cells for direct current generation. Issues such as scale up remain challenging for the
18 future application of RBC technology and topics such as phosphorus removal and denitrification still
19 require further research. High volumetric removal rate, solids retention, low footprint, hydraulic
20 residence times are characteristics of RBCs. The RBC is therefore an ideal candidate for hybrid
21 processes for upgrading works maximising efficiency of existing infrastructure and minimising
22 energy consumption for nutrient removal. This review will provide a link between disciplines and
23 discuss recent developments in RBC research and comparison of recent process designs are
24 provided (section 2). The microbial features of the RBC biofilm are highlighted (section 3) and
25 topics such as biological nitrogen removal and priority pollutant remediation are discussed (section
26 4 & 5). Developments in kinetics and modelling are highlighted (section 6) and future research
27 themes are mentioned

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

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40 Nomenclature

- 41 A_d = total disc surface area (L^2)
- 42 A_{exp} = area of exposed disc (L^2)
- 43 A_{sub} = area of submerged disc (L^2)
- 44 A_t = cross sectional area of tank (L^2)
- 45 C_{Bf} = compound concentration at the biofilm
46 surface (ML^{-3})
- 47 C_{Lf} = compound concentration at liquid film
48 surface (ML^{-3})
- 49 C_i = influent compound concentration (ML^{-3})
- 50 C_e = effluent compound concentration (ML^{-3})
- 51 C_T = compound concentration in the tank (ML^{-3})
- 52 C^* = equilibrium compound concentration at a
53 given temperature (ML^{-3})
- 54 D_L = diffusion coefficient of oxygen in water (L^2T^{-1})
55
- 56 e = distance from disc edge to the basin (L)
- 57 g = acceleration due to gravity (LT^{-2})
- 58 H = distance between the disc centre to the liquid
59 free surface (L)
- 60 K_C = half saturation constant for compound (ML^{-3})
61
- 62 K_L^{air} = oxygen mass transfer coefficient film
- 63 K_L = overall oxygen mass transfer coefficient (LT^{-1})
64
- 65 $K_L a_t$ = volumetric oxygen mass transfer
66 coefficient total (T^{-1})
- 67 N = number of discs
- 68 N_v = volume renewal number (T^{-1})
- 69 Q = reactor flow rate (L^3T^{-1})
- 70 r = radius of disc (L)
- 71 r_a = substrate removal rate ($ML^{-2}T^{-1}$)
- 72 s = half space between discs (L)
- 73 t_R = contact time per rotation (T)
- 74 V = wet volume of reactor (L^3)
- 75 V_{Lf} = volume of liquid film (L^3)
- 76 U_{max} = maximum substrate removal rate ($ML^{-2}T^{-1}$)
- 77 X_a = concentration of attached biomass (ML^{-3})
- 78 Y_a = yield coefficient for attached biomass
- 79 δ = liquid film thickness (L)
- 80 δ_{bf} = biofilm thickness (L)
- 81 μ = Absolute viscosity of a liquid (MLT^{-1})
- 82 μ_{max} = maximum specific growth rate (T^{-1})
- 83 ρ = fluid density (ML^{-3})
- 84 ω = rotational speed (RPM)
- 85 $\omega' = \omega/60$
- 86 \square = disc diameter (L)
- 87 \square_0 = wetted disc diameter (L)

88 1. Introduction

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

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

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

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

135 **1.1. Process development history**

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137 The RBC concept originated in Germany in 1920's where it was described as a 'rotating aerobic
138 mass' fixed to a media support (Chan and Stenstrom, 1981), although the first plant was registered
139 in the United States and was named the 'Contact Filter' or 'Biologic Wheel' consisting of partially
140 submerged rotating plates (Doman, 1929). This device served as an alternative to the trickling filter
141 with 1/10th the land area, and lower power cost than AS (Allen, 1929). Commercial interest in RBCs
142 was minimal, until the modern emergence of the so called 'drip body immersion systems' (Hartman,
143 1960). The design was patented (Hartmann, 1961) and the first recorded experimental pilot RBC
144 was undertaken to test performance (Popel, 1964). This landmark study informed future RBC design
145 which progressed in the 1960's. For example the surface BOD₅ loading from this study of ~3g.m².d

146 ¹ is similar to modern overall organic surface loadings of 3-15 g.BOD₅.m².d⁻¹ that have been applied
147 recently (Rittmann and McCarty, 2001). The availability of stronger, lighter and affordable materials
148 such as plastics increased the stability of media and increased the surface area available for
149 microbiological growth, which improved treatment capacity. This allowed a plethora of capital
150 ventures in the 1960 and 70's. The RBC was applied for biological treatment under a variety of
151 influent types, organic and hydraulic regimes (Rittmann and McCarty, 2001, Cortez et al. 2008). A
152 Japanese company known as Kubota submitted a patent application for an RBC capable of
153 simultaneous nitrification and denitrification, using variable submergence to facilitate multiple
154 nutrient removal regimes (Sim 1988). A series of process failures have been noted for RBCs, many
155 were due to inappropriate mechanical design which did not account for biomass growth, often
156 leading to shaft, bearing and media malfunctions (Mba et al. 1999). A report suggested that
157 equipment warranty should protect the owner from failure (Weston, 1985), however often liability
158 contracts rarely exceeded 3 years which provided little stimulus to fix inherent mechanical issues
159 (Griffin and Findlay, 2000). Another challenge was supplier competition led to an exaggeration of
160 possible removal rates (Rittmann and McCarty, 2001); allowable loadings varied by a factor of 7
161 between suppliers (Ross et al. 1994). Unlike other major biological processes, designers were
162 initially reliant on proprietors design criterion for process control (Ross et al. 1994). Hydraulic
163 loading was previously applied as a design parameter, but was usually inappropriate by not
164 considering organic strength; biodegradability, toxicity and temperature which impact microbial
165 process performance (Steiner 1997). Design criteria should be used that incorporate fundamental
166 parameters including microbial activity, organic loading and substrate utilisation rate.

167 **2. Process engineering of RBCs**

168 **2.1. Types**

169 There are two main types of RBC; integral and modular. Integral systems consist of a single unit
170 combining primary settlement, RBC biozone and either a contained or separate final clarifier. (Fig.
171 1a). Integral units are usually contained within a package plant and have a treatment capacity of
172 ≤250 population equivalents (PE) (Findlay et al. 1993). Conversely, modular systems have separate
173 operations for primary, secondary, and solids treatment respectively and usually treat PE >1000
174 (Griffin and Findlay, 2000), which allows more flexible process configurations (Fig. 1b,c). However
175 size and weight constraints generally limit RBCs to a size of 3.5 m disc diameter. Modular RBCs
176 can be operated using parallel flow separation between units allowing operation within acceptable
177 loading limits (Fig. 1b). In contrast, if effluent quality is of principal concern, RBCs are often
178 operated in series, with an nth RBC operating distal in the flow sheet (Fig. 1c). Typically a
179 submergence of 40% (wet disc level), is used (Cortez et al. 2008). By increasing the submergence
180 (Fig. 1d), the conditions in the reactor become increasingly anaerobic which could favour processes
181 that require reduced oxygen levels such as denitrification (Teixeira and Oliveira, 2001). Hybrid
182 systems operate a RBC combined with another unit operation to improve the stability of a process
183 that has strong or variable loading, increase load capacity or improve the achievable effluent
184 standard (Vesilind, 2003; Hoyland et al. 2010). Common configurations include a RBC/biofilm (Fig.
185 1e) or RBC/suspended growth combination which can be used for the upgrade of capacity
186 (roughing) or provide tertiary treatment (Fig. 1f) (Vesilind, 2003, Upton et al. 1995). The
187 RBC/wetland combination has been applied to improve discharge consents for small works and
188 provide a storm flow buffer (Griffin, 2003) (Fig. 1e). For longevity, the RBC is protected using ultra
189 violet light resistant media (e.g plastic with carbon black) or by covering the RBC within protective
190 casing which can also reduce heat loss and flies/odour.

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

192 2.2. Cost

193 The capital expenditure (CAPEX) and operational expenditure (OPEX) of RBCs has been
194 minimised through reduced commissioning, monitoring and maintenance costs compared to AS
195 processes. In the UK, half the CAPEX for RBCs is related to mechanical and electrical components.
196 The CAPEX cost per head in RBCs is inversely proportional to the PE for treatment. At PE >1000
197 the CAPEX cost decreased by up to 50% (Upton et al. 1995). For example Labella et al. (1972)
198 compared the cost of an activated sludge plant (ASP) and RBC system treating winery waste with a
199 flow of $1.8 \times 10^3 \text{ m}^3\text{d}^{-1}$. They noted that while capital expenditure were similar, estimated power
200 consumption was less than half that of a concrete tank aerated ASP. An RBC was found to be on
201 average 35% cheaper per PE per year compared to trickling filters due to lower land area and
202 running costs (Upton et al. 1995). However other authors have suggested that the OPEX of an RBC
203 are similar to suspended growth systems and savings are only apparent with CAPEX (Ware et al.
204 1990). Fountoulakis et al. (2009) identified that RBCs had 29% lower but 44% higher CAPEX than
205 packed bed filters and horizontal surface flow wetlands respectively. In addition RBCs were shown
206 to have five times the power consumption than packed bed filters when operated within the organic
207 loading rate (OLR) range of $0.53\text{-}2.01 \text{ kg.COD.m}^{-3}\text{d}^{-1}$. The power efficiency of a RBC operated a
208 7.5 horse power motor ranged from 72-88% at 25-100% load capacity respectively (Brenner and
209 Opaken 1984). However RBCs are appropriate for decentralised water treatment systems which
210 generally have lower OPEX costs compared to a centralised approach which may require specialist
211 labour and process control (Fountoulakis et al. 2009).

212 2.3. Substrate

213 Substrate dependent parameters in RBCs are staging, organic loading, recycle and flowsheet
214 position. The hydraulic considerations include hydraulic residence time (HRT), tip speed, media
215 specific surface area, compound transfer rate and submergence. However there is considerable
216 overlap between these parameters, for example the inverse relationship between HRT and OLR
217 (Patwardhan, 2003). Another example is the association between rotational speed, oxygen transfer
218 rate (OTR) and biofilm thickness. A key criterion for RBC reactors is surface organic load which is
219 defined as substrate ($\text{kg.COD/N/pollutant}$) applied per square metre (specific or nominal) of media
220 per day. In RBCs, as the loading rate increases the removal rate increases in proportion until another
221 parameter becomes limiting (Fig. 2). For example Hiras et al. (2004) operated a two stage
222 predenitrification and aerobic RBC for the treatment of settled municipal sewage. A decrease in the
223 percentage removal of COD with increasing OLR was observed from 50 to 35% at OLR of 90 and
224 $360 \text{ gm}^{-2}\text{d}^{-1}$ respectively. This can be explained by biofilm oxygen transfer rate limiting the
225 efficiency of substrate utilisation (Di Palma et al. 2009). However the organic removal rate
226 increased from 45 to $125 \text{ gm}^{-2}\text{d}^{-1}$ suggesting that there was more capacity for bulk COD removal in
227 the system. Therefore the highest substrate removal rate is achieved at the maximum loading before
228 the transfer of rate limiting compound is exceeded (Fig. 2). In RBC biofilms mass transfer
229 restrictions usually masks biological reaction kinetic limitations. As both substrates diffuse from the
230 bulk fluid in the same direction and one or both will become limiting at a certain depth in the
231 biofilm. In RBC biofilms there is an equilibrium between the rate of substrate consumption and
232 diffusional transfer which influences the penetration depth (Stewart and Franklin, 2008). Under
233 constant loading the microbial community will attain steady state based on available substrates and
234 competition for electron acceptors and space. In the biofilm there is a layering of bacteria based on
235 prevailing conditions with the lowest substrate redox state proximal to the media (Okabe et al.
236 1999).

237 Note to publisher: insert fig 2.

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

242 improves removal rate, process stability and permits autotrophic nitrification at higher organic loads
243 than normally possible (Tawfik et al., 2002; Kulikowska et al. 2010; Najafpour et al., 2006). Staging
244 can permit enhanced ability to manage shock loads providing the biomass has sufficient substrate.
245 The positive impact of staging on RBC performance was found to be negligible after four stages
246 (Adreadakis 1987), although this is dependent on wastewater load and composition. Step feeding
247 can be used to reduce the initial effective substrate concentration. Ayoub and Saikaly (2004) showed
248 that step feeding had minimal impact on removal of RBC bulk COD removal rate, however $\text{NH}_4\text{-N}$
249 removal increased by 18%, by staggering the organic load which reduced the likelihood of oxygen
250 limitation (Rittmann et al. 1983). Recycling effluent permits greater portions of the biofilm to nitrify
251 by diluting the influent organic concentration (Ayoub and Saikaly 2004). The recycle can be either
252 pre, post or from the clarifier depending on treatment aim (Fig. 1c). Recycling settled solids helps
253 aid bacterial retention as sloughed biomass is returned to the reactor. Other biomass associated
254 products like extracellular enzymes may be recycled which could aid the breakdown of complex
255 polymers, which constitute roughly half of domestic wastewater (Confer and Logan, 1998).

256 **2.4. Hydrodynamics**

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

269 **2.5. Media composition**

270 The RBC media can be present as discs, mesh plates, saddles or rings in a packed bed reactor, which
271 resembles a partially submerged, rotating, moving bed biofilm reactor (Ware et al 1990;
272 Sirianuntapiboon and Chumlaong, 2013). The RBC media commonly has a specific surface area of
273 $150\text{-}250\text{ m}^2\text{m}^{-3}$ for biofilm growth which supports high removal rates at low HRTs. Lower density
274 media is normally applied at the front-end of the works which typically has high organic loads
275 (Cortez et al. 2008). Support media should be insoluble, have high mechanical and biological
276 stability, and be cost effective (Leenen et al. 1996). The media physicochemical composition and
277 architecture both impact on the microbial biofilm and the removal rate of substrates (Tawfik and
278 Klapwijk, 2010; Stephenson et al. 2013). A comparison between the oxygen transfer efficiency in
279 RBCs was between 2-5 and 1-2 kgkWh^{-1} for comparable packed bed and disc RBCs respectively
280 (Mathure and Patwardhan, 2005). However previously it was noted that any performance gains from
281 packed bed RBCs are usually offset by higher CAPEX costs and reliability issues (Ware et al. 1990).
282 Polyurethane foam has been utilised to increase surface area for biofilm growth, reported specific
283 surface areas range from $600\text{-}1000\text{ m}^2\text{m}^{-3}$ can provide greater solids retention, however careful
284 management of biofilm thickness and pore clogging is required (Windey et al. 2005; Tawfik et al.
285 2010). Chen et al. (2006) used a 'net-like' media which increased the surface area of flat discs to
286 facilitate a nitrification rate of $0.6\text{ g.N.m}^{-2}\text{d}^{-1}$ (Table 1). Lui et al. (2008), utilised a pyridinium type
287 polymer sprayed to a non-woven carrier. They demonstrated the feasibility of autotrophic anaerobic
288 denitrification. It was claimed that surface properties of the pyridinium facilitated attachment of
289 nitrifiers permitting a nitrification rate $>26\text{ kgm}^{-2}\text{d}^{-1}$. Whereas Hassard et al. (2014) studied the
290 impact of OLR on removal rates of biofilm cultivated on a polyvinylchloride-like mesh, polyester
291 and polyurethane foam in RBC-like reactors operated concurrently. They identified that under high
292 loading conditions macroscale media pore size was the most significant parameter governing

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

295 Research suggests a link between the initial adhering community, subsequent established
296 biofilm and reactor performance (Stephenson et al. 2013). Different media physicochemical
297 properties have been suggested to select for different bacterial groups. For example media
298 hydrophobicity influences adhesion, due to the difference in size of the aqueous boundary layer
299 preventing bacterial contact. This effect is reduced in more hydrophobic materials, promoting
300 adhesion (Khan et al. 2013). The surface roughness can impact bacterial adhesion. Singh et al.
301 (2011) found that media with a roughness of >20 nm has high protein adsorption, which increases
302 the effective media hydrophilicity, increasing the water layer and preventing adsorption of bacteria
303 (Singh et al. 2011). In contrast a media of intermediate roughness gains the surface area benefit and
304 high bacterial adsorption. Hassard et al. (2014), studied this phenomena using bench scale RBC-like
305 reactors at OLR from 16-160 g.sCOD.m⁻²d⁻¹ and identified that media with an average roughness
306 <20 nm had 4.5 times more biomass on average compared to similar media with an average
307 roughness of 35 nm. However the removal rates of sCOD were similar suggesting the high
308 roughness biofilm had greater specific bacterial activity.

309 **2.6. Scale up considerations**

310 Appropriate scale up of RBCs is critical to validate whether performance will be comparable from
311 bench/pilot to full scale (Arvin and Harremoës, 1990). For RBCs, scale up should incorporate
312 parameters of hydrodynamics, media active surface area, flow, organic loading, oxygen transfer,
313 bacterial growth rate, biofilm accumulation and detachment. However most models only
314 accommodate one of these variables. For example Wilson et al. (1980) developed 'generalised
315 design loadings' based on 12 months data at different scales. However, resulting models failed to
316 consider operating/environmental conditions or process understanding (Harremoës and Gönenc,
317 1983). The use of tip speed is rarely a suitable parameter - as it increases (along with shear forces
318 and mixer power) to the square of the diameter. To simulate full scale, bench scale reactors were
319 previously operated at higher rotational speeds (to keep constant tip speed) which decreased the
320 contact time per rotation (Spengel and Dzombak, 1992). This also resulted in different shear
321 distributions influencing erosion and sloughing processes in the biofilm, greater mixing and improve
322 substrate removal. The empirical approach to scale-up involves constructing reactors of different
323 sizes and is popular but is generally expensive. After sufficient development a mechanistic model
324 can be developed, reducing the need for extensive testing. However these models are usually
325 appropriate only for identical operating and wastewater conditions. Dutta et al. (2010) constructed
326 three different sized RBCs to characterise the oxygen transfer coefficient at different scales. The
327 model was based around existing ones: the Activated Sludge Model No. 3 for biochemical reactions,
328 a multiculture biofilm model and an RBC model. However the main limitation for this approach is
329 that oxygen transfer should be suitably characterised on scale up, which is rarely the case.
330 Alternatively, design such that large reactors have chemical, dynamic, geometric and kinematic
331 homogeneity to bench scale trials (Spengel and Dzombak, 1992). The appreciation of scale up in
332 RBCs is far from complete however models which based on fundamentals are less sensitive to scale
333 up than empirical design parameters.

334 **3. Microbiology of RBCs**

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

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3.1. Structure

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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 3.3×10^9 to 3.9×10^{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).

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3.2. Function and activity

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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

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

403 **4. Biological nutrient removal in RBCs**

404 **4.1. Nitrification**

405 Rotating biological contactors are used for nitrification and denitrification of a range of influent
406 conditions (Cortez et al. 2007, De Clippeleir et al. 2011). Stringent rules govern nitrogen discharge
407 and the energy cost/greenhouse gas emissions are a growing concern (Ainger et al. 2009). The RBC
408 has potential benefits by reducing tank volume, HRT and aeration demand coupled with nitrogen
409 removal at greater loadings compared to traditional treatments. Furthermore RBCs have been
410 applied for refractory or contaminated wastes. For example Kulikowska et al. (2010) achieved a
411 maximum nitrification rate of $4.8 \text{ g.NH}_4\text{-N.m}^2\text{d}^{-1}$ at a loading of $6.6 \text{ g.NH}_4\text{-N.m}^2\text{d}^{-1}$ (Table 1).
412 Sequence analysis revealed microbial diversity decreased with time, suggesting a climax community
413 was attained. Diversity indices were resistant to shock loading of >70% of normal flow and
414 fluctuating performance, suggesting more sensitive measures of community change are required.

415 **4.2. Denitrification**

416 Denitrification is the dissimilarly reduction of nitrate to nitrite to dinitrogen gas under anoxic
417 conditions (Paredes et al. 2007). Conventional heterotrophic denitrification is possible in
418 wastewaters with a C/N ratio >2.5, without additional carbon sources (Hippen et al. 2001). As DO is
419 consumed within a biofilm the community becomes oxygen limited. Thereby facilitating
420 microenvironments where each consortia can develop. Helmer and Kunst, (1998) found that under
421 low DO conditions RBCs can remove up to 90% of the nitrogen load from landfill leachate.
422 Odegaard and Rusten (1980) found that the $\text{NO}_x\text{-N}$ recycle ratio in RBCs improved denitrification
423 rate. Batch testing revealed that nitrogen removal was carbon limited, suggesting autotrophic
424 degradation satisfied the nitrogen deficit. Cortez et al. (2011a.) achieved almost complete nitrogen
425 removal from landfill leachate using conventional denitrification in an anoxic RBC, they identified
426 that preozonation was required to remove refractory carbon compounds. Gupta and Gupta (2001)
427 augmented a myxotroph known as *Paracoccus denitrificans* to undertake simultaneous aerobic
428 carbon oxidation, nitrification and denitrification. *P. denitrificans* removed a maximum of 26 and
429 $1.9 \text{ gm}^{-2}\text{d}^{-1}$ of COD and nitrogen respectively in an RBC. However, the aerobic denitrification rate
430 was slower than conventional denitrification. At high nitrate concentrations ($>500 \text{ mgL}^{-1}$) inorganic
431 phosphorus can limit denitrification. Cortez et al. (2011b) suggested that phosphorus improves
432 overall biofilm denitrifying activity and nitrogen removal by promoting bacterial growth. Teixeira
433 and Oliveira (2000) improved denitrification by 30% upon the addition of phosphorus. Hanhan et al.
434 (2005) compared the nitrogen removal rates in full scale pre-denitrifying RBCs. The highest
435 reported removal was $\sim 2 \text{ g.N.m}^{-2}\text{d}^{-1}$ with a HRT of 0.2 d (Table 1). The nitrogen removal rate
436 decreased with increasing rotational speed, suggesting oxygen inhibition led to suppression of the
437 denitrification pathway. Teixeira and Oliveira (2001) demonstrated that increased disc submergence
438 from 64.5 to 100% improved the TN removal by 63% but had delayed start-up.

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

442 anammox bacteria; providing the biofilm is oxygen limited or NOBs are suppressed (Pynaert et al.
443 2004). De Clippeleir et al. (2011) showed that decreasing HRT from 0.66 to 0.18 d stimulated a
444 decrease in removal rate from 2.2 to 1.6 g.N m⁻²d⁻¹ (Table 1). This was attributed to increased
445 nitrification by *Nitrospira* sp. which proliferated at DO concentrations of >1.2 mgL⁻¹. Stepwise
446 loading increases allowed removal rates in excess of 1.8 g.N m⁻²d⁻¹ (Pynaert et al. 2004). Vlaeminck
447 et al. (2009) tested the feasibility of an oxygen limited autotrophic nitrification and denitrification
448 (OLAND) process to treat digestate from source separated black water and achieved a removal rate
449 of 0.71 g.N m⁻²d⁻¹. The nitrite oxidising bacteria were suppressed at free ammonia levels >3 mgL⁻¹,
450 however, DO levels <0.3 mgL⁻¹ are required for process stability. The effluent from this reactor had
451 a N/P ratio of 1 suggesting struvite production and therefore nutrient recovery is possible. However
452 facilitating struvite accumulating organisms in RBC biofilms has not received any attention. Windey
453 et al. (2005) showed that anammox bacteria could adapt to high salinity conditions of up to 30 gL⁻¹,
454 providing the RBC biofilm acclimation was gradual. The removal rate of nitrogen decreased from
455 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
456 similar study by Kartal; et al. (2006), identified that 45 gL⁻¹ of salt completely inhibited anammox
457 bacteria. Liu et al. (2008) suggested that the anammox consortium on RBCs were relatively resistant
458 to DO shocks. They found that a *Nitrosomonas eutropha*-like species protected the *Planctomycetes*
459 by sequestering potentially inhibiting DO levels.

460 Note to publisher: insert table 1

461 **4.3. Biological phosphorus removal**

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

480 **5. Priority pollutant remediation in RBCs**

481 Priority pollutant remediation can require the bioaugmentation or retention of specialised strains.
482 Bioaugmentation in RBC systems is usually achieved through addition of either suspended or freeze
483 dried artificial cultures or freeze dried biomass to the RBC (Stephenson and Stephenson, 1992).
484 Alternatively cultures of microbes can be grown in a side stream reactor prior to addition. The
485 natural solids retention of the RBC biofilm permits microbe retention without additional separation
486 or recirculation. Many systems require acclimatisation periods and are sensitive to shock/variable
487 loadings or intermittent feeding of the pollutant which is of import for the removal of priority
488 substances from wastewaters (Stephenson and Stephenson, 1992; Duque et al. 2011; Amorim et al.
489 2013).

490 5.1. Organic pollutants

491 Dye wastewater is a challenging form of organic pollutant as the dyes or breakdown products can be
492 toxic or mutagenic (Malachova et al. 2013). The RBC is ideal for dye treatment due to high biomass
493 retention, low startup costs, and appropriate technology level for developing countries (Robinson et
494 al. 2001). Wastewater dyes are initially absorbed to the biofilm but a continually exposed biomass
495 will eventually saturate. Most dyes do not penetrate bacteria as they have a high molecular weight
496 and contain hydrophobic groups, which are a barrier to biocenosis (Pearce et al. 2003). The
497 bioaugmentation of white rot fungi (WRF) e.g. *Phanerochaete* sp. has been undertaken in RBC
498 systems as they excrete non-specific extracellular hydrolytic enzymes with dye decolouring capacity
499 (Pakshirajan and Kheria, 2012). Novotný et al. (2011) found a surface decolourisation rate of 0.63,
500 0.19 and 0.01 mg.m⁻².h⁻¹ for Remazol Brilliant Blue R, Methylene Blue and Azure B respectively by
501 the augmented fungus *Dichomitus squalens* (Table 2). Dye degradation is often undertaken as a
502 secondary metabolism so allochthonous carbon sources are required to maintain activity. Novotný et
503 al. (2011) identified that *D. squalens* has a minimum glucose concentration of 0.018 gL⁻¹ for
504 effective dye decolourisation. Pakshirajan and Kheria, (2012) showed that the decolourisation rate of
505 WRF *P. chrysosporium* is proportional to glucose concentrations to a limit of 10 gL⁻¹. The use of
506 molasses dosing decreased the decolourisation rate of *P. chrysosporium* by 20% compared to
507 glucose control (Pakshirajan and Kheria, 2012). Dye removal has been correlated with activity of
508 manganese dependent peroxidase and lignin peroxidases. For full dye remediation from wastewater
509 the dye should be decolourised and detoxified. Malachova et al. (2013) utilized an RBC
510 bioaugmented with *Irpex lacteus* 931, and achieved a batch methyl blue decolourisation rate of 9.4
511 mgm⁻².d⁻¹. Decolourisation resulted in reduced toxicity level of the wastewater. However the WRF
512 are susceptible to bacterial stress which usually prevents application under real wastewater
513 conditions. Nilsson et al. (2006) used an RBC augmented with *Trametes versicolor* to treat real
514 textile wastewater and achieved 60-70% decolourisation efficiency. Research should identify if
515 WRF can be utilized in RBCs with appropriate scale up.

516 Note to publisher insert table 2.

517 Duque et al. (2011) inoculated a strain capable of degrading 2-fluorophenol and
518 demonstrated increased removal efficiency under constant pollutant loading. Under variable loading
519 the pollutant removal decreased even though the community remained in the biofilm. Amorim et al.
520 (2013) studied the impact of shock loadings of 4-fluorocinnamic acid (4-FCA) on an augmented
521 RBC. The removal efficiency was increased from 8 to 46% at surface loadings of 73 to 168 g.m⁻².d⁻¹
522 respectively (Table 3). Isolation of biofilm strains and batch testing revealed that two strains
523 completely mineralised 4-FCA. Sequence analysis revealed a 97% similarity to the original
524 augmented *Rhodococcus* strain, suggesting horizontal gene transfer or genetic drift had occurred
525 (Singh et al. 2006).. The RBC reactor has also been applied for removal of non-aqueous phase
526 liquids (NAPL) (Mukherji and Chavan 2012). Chavan and Mukherji (2008.b) found that a mixed
527 freshwater phototrophic community augmented with *Burkholderia cepacia* had a removal rate of
528 >26 gm⁻².d⁻¹ for removal of diesel NAPL. The NAPL component of the wastewater was likely sorbed
529 onto the biofilm for subsequent biodegradation of the aliphatic fraction (Mukherji and Chavan
530 (2012). Operation with the co-contaminant phenol slightly reduced the removal efficacy of NAPL
531 but resulted in complete phenol removal (Chavan and Mukherji, 2010). Under constant pollutant
532 loading in RBCs it is therefore important to promote proliferation of the augmented community at
533 functional levels.

534 Note to publisher insert table 3

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

539 contrast micropollutants which are non-sorptive and non-readily biodegradable are of greatest
540 concern. In this study the RBC achieved 99% removal efficiency of methyl dihydrojasmonate
541 compared to 98, 93, 82% for an ASP, trickling filter and carousel setup respectively. The removal of
542 6-Acetyl-1,1,2,4,4,7-hexamethylteraline (AHTN) in the RBC was inferior compared to other
543 secondary treatments which could be due to poor removal of particulate matter. Batt et al. (2007)
544 compared four treatment works with similar influent concentrations of Ciprofloxacin (CP),
545 Sulfamethoxazole (SM), Tetracycline (TC) and Trimethoprim (TM) and found that the RBC had
546 comparable removal of antibiotics of between 52-95% removal of CP, TC and TM to an extended
547 aeration ASP but with lower HRT and presumably treatment cost. In contrast the RBC demonstrated
548 43% lower SM removal compared to the ASP. The degradation behavior of this antibiotic could be
549 due to physical differences between bacteria in biofilms and suspended growth..

550 **5.2. Inorganic pollutants**

551 Biological heavy metal removal relies on both the sorption of the metal species to biomass and the
552 bioaccumulation by metabolic processes (Costley and Wallis, 2001). The RBC microbial biofilm is
553 suitable for biosorption as there is a high contact area for sorption and a long MCRT. However the
554 metal removal rate will decrease with time, as the attraction sites become saturated (Matheikal et al.
555 1991). For example Sirianuntapiboon and Chumlaong (2013) found that an RBC had a decreased
556 removal efficiency of 64-45 and 80-85% with increased loading which corresponded to a removal
557 rate of between 255-400 and 255-480 mg.m⁻²d⁻¹ for Ni and Pb respectively (Table 4). This is similar
558 to removal rates reported for Cu of ~450 mg.m⁻²d⁻¹ using activated sludge consortia (Costley and
559 Wallis, 2001) (Table 4). To prevent saturation it is necessary to remove the metal loaded biomass by
560 suitable treatment. However this is costly and produces secondary waste issues (Costley and Wallis
561 2001). Costley and Wallis, (2001) showed that multiple cycles of sorption/desorption, using a dilute
562 (<0.5 M) acid did not impact the adsorption efficiency of a mixed culture RBC biofilm, suggesting
563 reuse was possible. The removal rates demonstrated by Costley and Wallis (2001) of ~ 640, 450 and
564 320 mg.m⁻²d⁻¹ for Zn, Cu and Cd appeared dependent on loading and the availability of free sorption
565 sites. Regression analysis reveals that the loading rate predicts removal rate between loads of 0.003-
566 762.8 mg.metal.m⁻²d⁻¹ (R² = 0.9, P<0.001) (Table 4).

567 Note to publisher: insert table 4

568 **6. Modelling of RBC reactors**

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

584 **6.1. Substrate utilization in RBCs**

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

587 limitation to/from the biofilm and the kinetic considerations governing biodegradable substrate
 588 utilization. Alternatively the removal of soluble substances is determined by the boundary layer
 589 diffusion resistance, into the biofilm prior to microbial degradation within the interior (Arvin and
 590 Harremoës, 1990). An empirical relationship to predict effluent BOD₅ was determined by the US
 591 Environmental Protection Agency (Brenner and Opaken 1984) in which:

$$592 \frac{C_e}{C_i} = e^{K(0.000125 V/Q)^{0.5}} \quad (1)$$

593 In which:

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

595 V = media volume (m³)

596 Q = hydraulic loading (Ls⁻¹)

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

$$600 r_a = \left(\frac{\mu_{max}}{X_a} \right) / Y_a \quad (2)$$

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

$$603 r_a = \left(\frac{K_C}{U_{max}} \right) \cdot \left(\frac{A_d}{QC_i} \right) + \left(\frac{1}{\mu_{max}} \right) \quad (3)$$

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

$$624 \frac{dC^{Lf}}{dt} = K_L^{air} \frac{A_{exp}}{V_{Lf}} (C^* - C^{Lf}) + K_L \frac{A_{sub}}{V_{Lf}} (C^T - C^{Lf}) - K_L \frac{A}{V_{Lf}} (C^{Lf} - C^{Bf} \Big|_{x=\delta_{Bf}}) \quad (4)$$

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

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

633 **6.2. Oxygen transfer in RBCs**

634 The oxygen transfer rate (OTR) determines the biofilm oxygen concentration and hence the selected
635 removal regime in RBCs. Initially, oxygen must diffuse from the bulk water/gas phase across the
636 boundary layer, into the film layer and eventually into the biofilm itself. The rate of diffusion is
637 dependent on the diffusion coefficient of oxygen and the distance according to Fick's Law (Stewart,
638 2003). Originally it was thought that the majority of transfer occurs with biofilm contact to the air
639 phase and therefore bulk fluid concentration was less important (Hartman, 1960). Other models
640 were developed with the assumption that substrate alone rather than oxygen is limiting in RBCs:
641 these are now deemed unsuitable (Clark et al. 1978; Spengel and Dzombak, 1992). The OTR is
642 related to the difference between the liquid phase and equilibrium concentration, in the liquid film
643 and RBC biofilm (Chavan and Mukherji 2008.a). Hansford et al. (1978), presented one of the first
644 attempts to include mass transport resistance to OTR. Initial models of OTR assumed that
645 turbulence, wave generation and immersion dominate (Patwardhan, 2003). An alternative method is
646 that oxygenation occurs during film breakup and renewal. This is caused by the air/water cycling
647 involving the interaction with rotational derived forces, which overcome film layer surface tension.
648 The rate of renewal is dependent on the rotational speed, disc diameter, position and half spacing
649 (Table 5) (Chavan and Mukherji, 2008.a). A study suggested that the relationship between liquid
650 film renewal and the OTR was linear under sterile conditions (Kim and Molof, 1982). Attached
651 biofilm increases the OTR, by enhancing concentration gradients due to consumption in the film and
652 adsorption to the biofilm (Kim and Molof, 1982; Zeevalkink et al. 1979). However biofilm growth
653 can reduce OTR by clogging pores which reduces mass transfer, Friedman et al. (1979) related
654 oxygen transfer coefficient to rpm alone. Rittmann et al. (1983), identified the import of adsorption
655 for OT at high rotational speed (>25) whereas diffusive film transport dominated during operation at
656 normal rotor speed. Kubsad et al. (2004), compared two forms of the Kim and Molof (1982) model
657 to alternatives and found appropriate predictive fit providing the volume renewal number can be
658 estimated effectively (Table 5). Di Palma et al. (2009) calibrated a previously defined model and
659 found that the k_{LA} increased in a linear fashion between the speeds of 3 and 10 rpm at bench scale.
660 The majority of film renewal is thought to occur when the surface tension resistance is broken under
661 the effect of gravity after the so called 'falling film' theory (Zhang et al. 2009).

662 Note to publisher: insert table 5

663 **7. Novel applications of RBCs**

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

677 and removal of nitrogen and phosphorus utilising an algal RBC-type reactor. The reactor design
678 consisted of a RBC drum with ropes and scraper blades which collected the algae. The maximum
679 harvested biomass produced was $\sim 30 \text{ gm}^{-2}\text{d}^{-1}$ of total solids. The algal RBC reactor achieved
680 removal rates of ~ 14 and $2 \text{ gm}^{-2}\text{d}^{-1}$ of soluble nitrogen and phosphorus respectively. Paule et al.
681 (2011) designed a vertical RBC with an intrinsic light source with removable polyethylene plates
682 produced $0.007 \text{ gm}^{-2}\text{d}^{-1}$ of volatile solids which could be used for energy generation.

683 **8. Conclusions**

684 The use of RBCs for conventional biological wastewater treatment to remove BOD₅ and ammonia
685 has been well established for the last three decades (Mueller et al. 1978). Application has largely
686 been at the lower end of the WWT scale, usually for up to 2000 P.E. (Griffin and Findlay 2000). The
687 limits of organic carbon renewal have been thoroughly investigated, with maximum OLRs of up to
688 $120 \text{ g. sCOD.m}^{-2}\text{d}^{-1}$ through using improved media optimised disc immersion and adjusted rotational
689 speeds (Teixeira and Oliveira 2001; Hanhan et al. 2005; Chen et al. 2008; Hassard et al. 2014).
690 However, novel configurations of media – such as mesh types (Chen et al. 2008; Lui et al. 2008;
691 Hassard et al. 2014) – and careful selection of media to enhance growth of certain bacterial
692 populations could increase applied OLRs and nitrogen loading rates (NLRs) incrementally (Khan et
693 al. 2012; Stephenson et al. 2013).

694 Recent research has demonstrated that the process can be adapted to remove nutrients, both
695 nitrogen and phosphorus, as with other biological processes (Yun et al. 2004; Hahnhan et al. 2005).
696 Novel RBC type processes, such as Hybrid Activated Sludge (HYBACS), has shown that new
697 combinations of suspended growth and fixed film on rotating media can provide higher organic
698 removal rates and efficient denitrification (Hoyland et al. 2008). Solid and liquid phase bacterial
699 interactions have been mentioned previously (Wanner and Kos 1990; Kenneth 1999), a better
700 understanding of these mechanisms merit further investigation in applying hybrid RBCs to energy
701 efficient nutrient removal. Biofilm systems are suited to providing a range of redox environments,
702 from wholly aerobic through anoxic to anaerobic conditions (Wuertz et al. 2004). Exploitation of
703 this phenomenon in RBCs is in its infancy at full-scale: for example, anammox (Strous et al. 1999)
704 has been demonstrated in RBCs (Siegrist et al. 1998; Vlaeminck et al. 2009; De Clippeleir et al.
705 2011). Control of disc immersion can be used to stimulate denitrification (Courstens et al. 2014).
706 Enhanced BPR requires alternating anaerobic and aerobic conditions (Yun et al. 2004); however
707 enforcing SBR type approaches in RBCs at full scale is challenging. Therefore manipulation of the
708 gaseous headspace, submergence, rotational speed or recycle in RBCs could be explored to
709 stimulate the enhanced BPR process.

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

726 could enhance treatment. The removal of dyes and other recalcitrant organic pollutants in RBCs
727 appears linked to bioaugmentation and propagation of allochthonous microbial populations with
728 pollutant degrading capacity (Novotny et al. 2012). The sensitivities and expense of these
729 communities remains an issue for application under real scenarios with representative wastewater.
730 Future research should focus on approaches suitable for scale-up or methods for upgrade or existing
731 works which struggle to deal with organic pollutants containing wastewater. Costley and Wallis
732 (2001) highlighted the potential of RBC biofilms for resource recovery, with the increasing price of
733 metals and nutrient fertilizer new opportunities could be created for cost positive wastewater
734 treatment (STOWA, 2012). The simplicity, adaptability, low land use and maintenance and high
735 volumetric activity of the RBC suggest that it will continue to help meet our wastewater treatment
736 requirements for years to come.

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740 **10. References**

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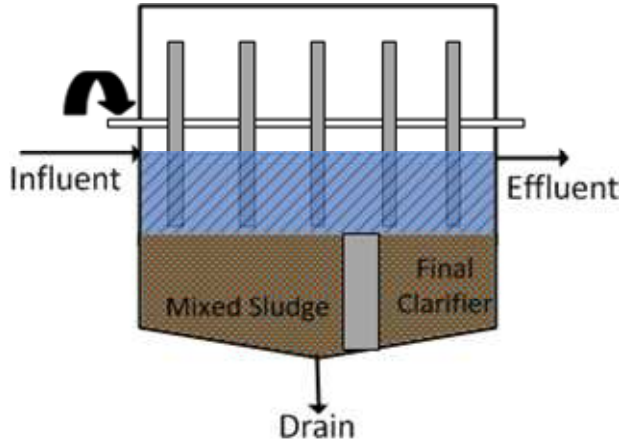
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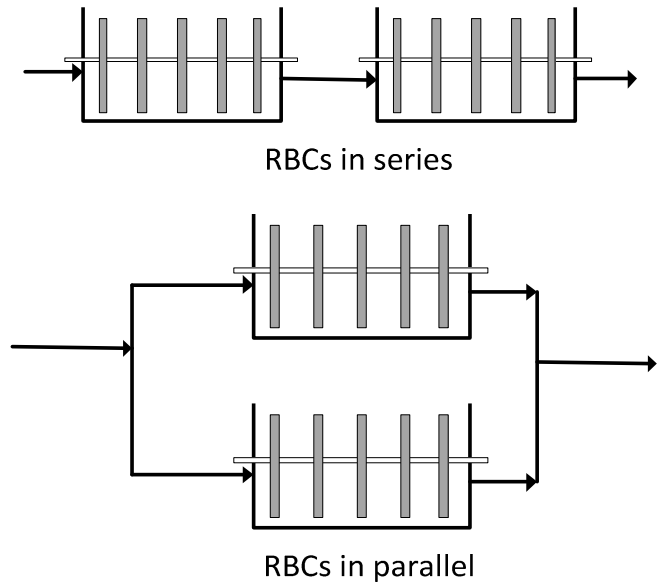
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Rotating fixed film biological contactors for wastewater treatment – a review - figures

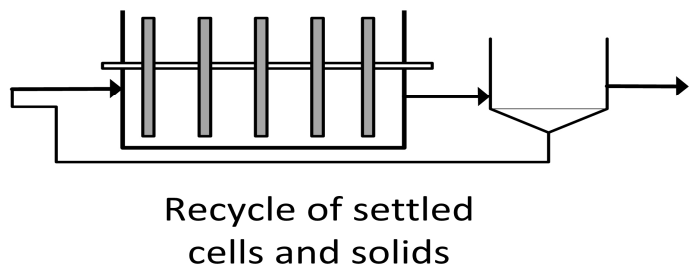
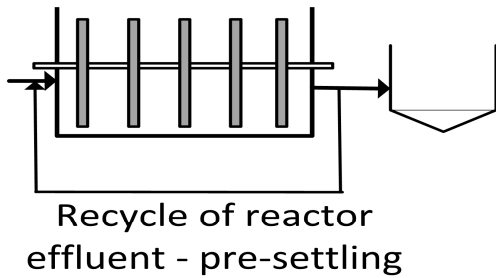
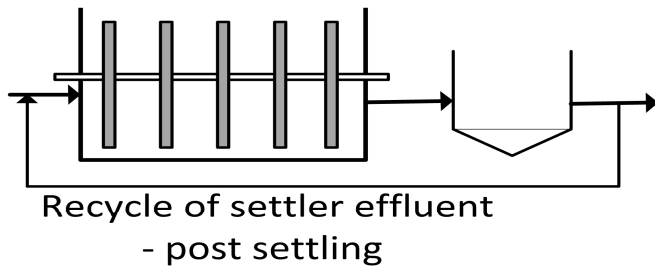
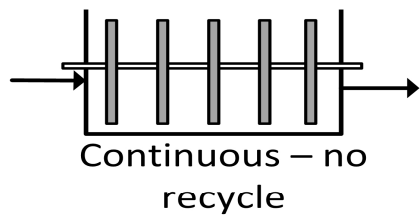
1.A - Integral RBC



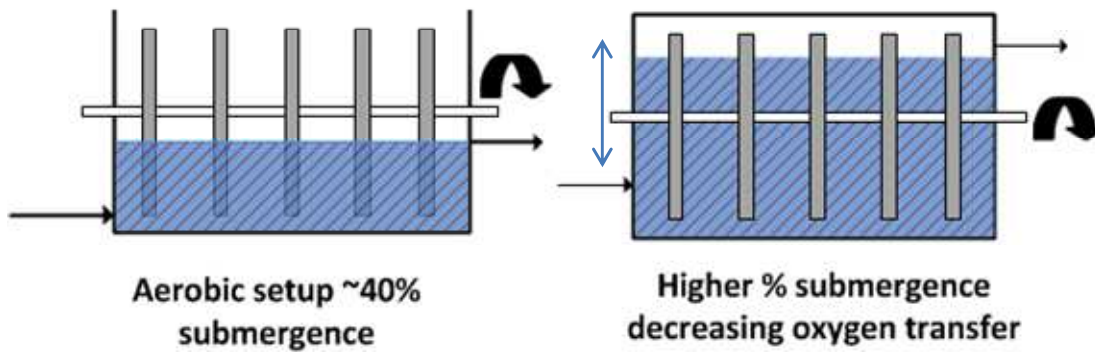
1.B – RBC flowsheet layout



1.C Reactor and recycle arrangements



1.D. Submergence



Hybrid systems

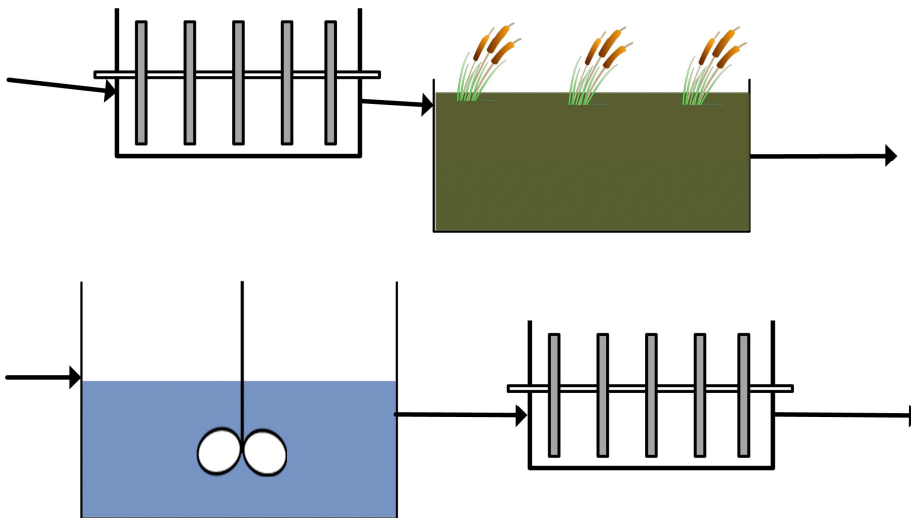


Figure 1: Process configurations of RBC technology.

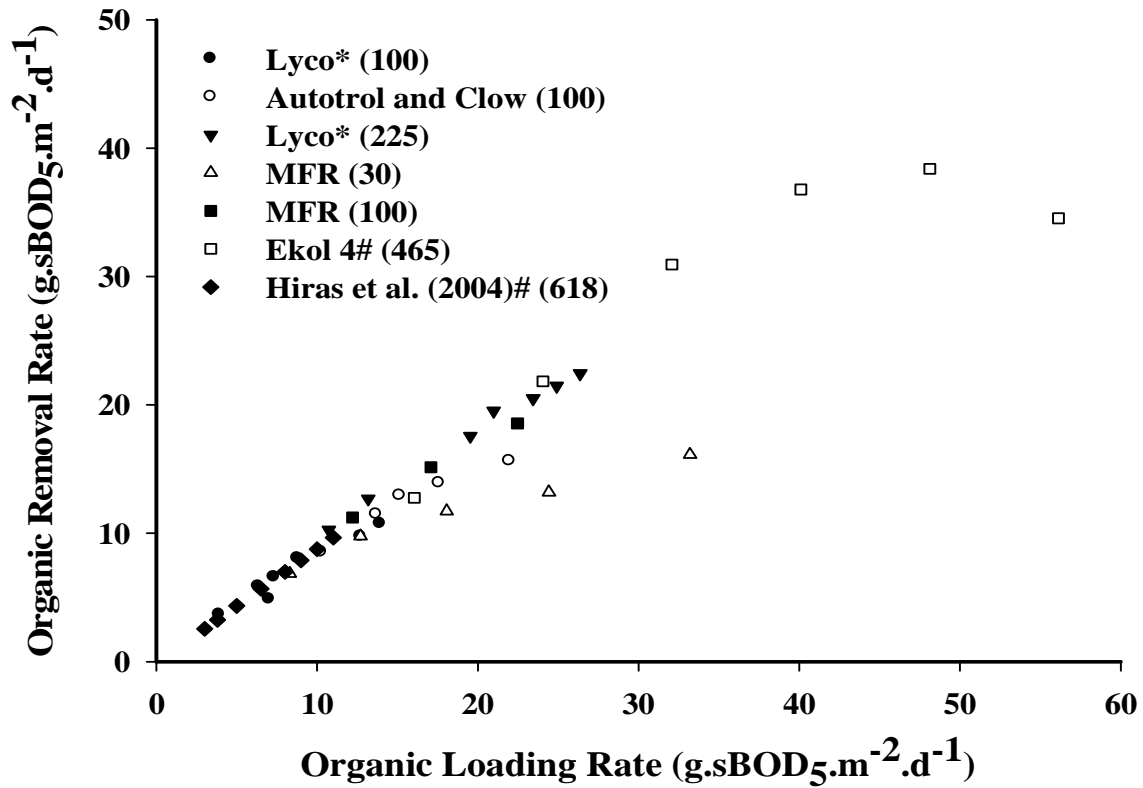
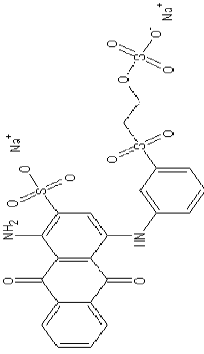
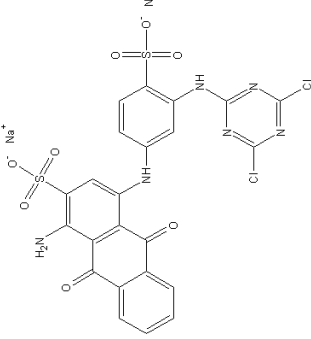
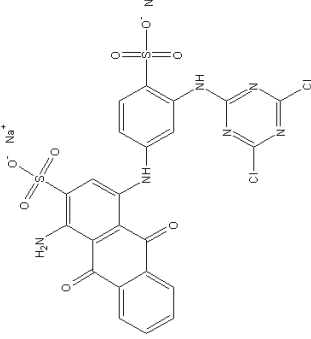
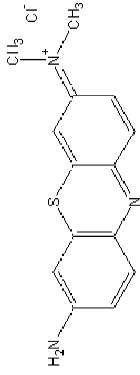
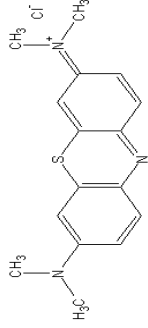


Figure 2: Organic removal rate with loading rate of RBCs from different manufacturers for soluble BOD, * total BOD, # total COD, numbers in brackets indicates influent concentration mgL⁻¹ of (Brenner and Opaken 1984), Ekol 4 data adapted from Fountoulakis et al. (2009), data from Hiras et al. (2004)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

Table 1 – Impact of N loading rate on NH₄-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 NH ₄ -N	770	12.9	11.9	0.77	OLAND	Windey et al. (2005)
	750	9.6	6.4	0.38		
Synthetic high nitrogen	1300	16.7	14.4	0.38	OLAND	Pynaert et al. (2004)
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		
	278	2.2	2.0	0.34		
Synthetic high nitrogen	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		
		3.8	2.9	5.00	Nitrification	Tawfik et al. (2002)
		7.6	1.5	2.50		
Landfill leachate	130	1.9	1.9			
	244	3.6	3.6	6.6	Nitrification	Kulikowska et al. (2010)
	332	4.8	3.6			
	451	6.6	4.8			
	24	3.5	0.2	0.16		
Real settled sewage	36	10.3	6.3	0.08	Nitrification	Hassard et al. (2014)
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 Decolourisat ion rate mg.m ⁻² .h ⁻¹	Referenc
Remazol Brilliant Blue R	50		Mineral medium 10 gL ⁻¹ glucose	<i>Dichomitus squalens</i>	95	0.66	0.63	Novotný et al. 2011
Reactive Blue 4	200		Citric buffer 21.4 gL ⁻¹ glucose	<i>Trametes versicolor</i>	70	0.26	0.18	Nilsson et al. 2006
Reactive Blue 4	100		10 gL ⁻¹ glucose	<i>Bjerkandera sp.</i>	99	0.07	0.06	Axelsson et al. 2006
Methylene Blue	50		Mineral medium 10 gL ⁻¹ glucose Malt extract glucose 10 gL ⁻¹ + 2% agar	<i>Dichomitus squalens</i> <i>Irpex lacteus 931</i>	85 55	0.22 N/A	0.19 0.39	Novotný et al. 2011 Malachov et al. 2011
Azure B	50		Mineral medium 10 gL ⁻¹ glucose	<i>Dichomitus squalens</i>	42	0.03	0.01	Novotný et al. 2011

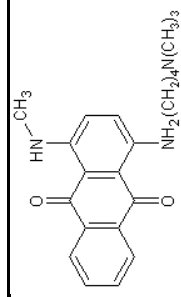
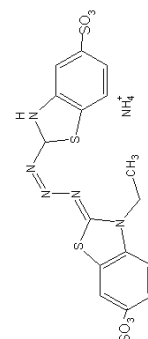
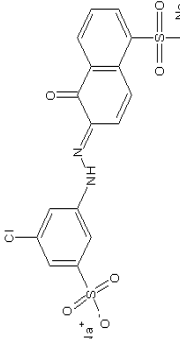
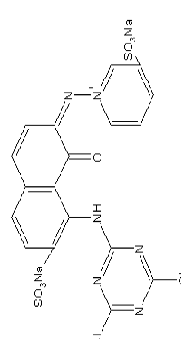
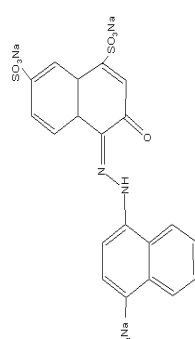
Compound	Dye concentration mgL ⁻¹	Structure	Growth substrate supplied	Organism	Removal I (%)	Dye surface loading mgm ⁻² h ⁻¹	Surface Decolourisation rate mgm ⁻² h ⁻¹	Refere nce
Basic Blue 22	200		8 gL ⁻¹ glucose	<i>Phanerochaete sordida</i>	0.80	0.89	0.71	Ge et al. (2004)
Direct Red 80	200		13.46 gL ⁻¹ glucose	<i>Phanerochaete chrysosporium</i>	0.80	0.55	0.44	Pakshira jan and Singh 2010
Mordant Blue 9	200		13.46 gL ⁻¹ glucose	<i>Phanerochaete chrysosporium</i>	0.62	0.55	0.34	Pakshir ajan and Singh 2010
Reactive Red 2	100		10 gL ⁻¹ glucose	<i>Bjerkandera sp.</i>	0.99	0.07	0.06	Axelsson et al. 2006
Acid Red 27	62.4		Kirk's medium 10.1 gL ⁻¹	<i>Trametes versicolor</i>	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	<i>Pseudomonas</i> sp., <i>Bacillus</i> , <i>Enterococcus</i> sp.	97.7	4.0	3.9	1.23	Sarayu and Sandhya 2012
Xylene	1226	<i>Pseudomonas</i> sp., <i>Bacillus</i> , <i>Enterococcus</i> sp.	98.5	4.1	4.1	1.23	Sarayu and Sandhya 2012
Phenol**	250	<i>Exiguobacterium aurantiacum</i>	48.4	154.4	74.7	1.00	Jeswani and Mukherji, 2012
Pyridine*	280	<i>E. aurantiacum</i>	34.2	169.5	58.0	0.50	Jeswani and Mukherji, 2012
Quinoline*	280	<i>E. aurantiacum</i>	48.9	345.3	168.9	0.50	Jeswani and Mukherji, 2012
Benzene*	200	<i>E. aurantiacum</i>	35.0	246.7	86.3	0.50	Jeswani and Mukherji, 2012
Napthalene*	60	<i>E. aurantiacum</i>	59.8	36.3	21.7	0.50	Jeswani and Mukherji, 2012
Phenanthrene*	0.5	<i>E. aurantiacum</i>	53.2	0.3	0.2	0.50	Jeswani and Mukherji, 2012
Phenanthrene	1.73	<i>Phanerochaete chrysosporium</i>	41.0	2.5	1.0	12.16	Zheng and Obbard, (2002)
Fluoranthene*	0.2	<i>E. aurantiacum</i>	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	<i>E. aurantiacum</i>	80.0	0.1	0.1	0.50	Jeswani and Mukherji, 2012
Pyrene	1.23	<i>P. chryso sporium</i>	65.9	1.8	1.2	12.07	Zheng and Obbard, (2002)
Benzol(o)pyrene	0.21	<i>P. chryso sporium</i>	96.9	0.3	0.3	11.72	Zheng and Obbard, (2002)
Trichloroethylene	30	Mixed culture (MC) augmented with <i>Thiosphaera pantotropa</i> MC augmented with 2-fluorophenol degrader (FP1) MC from RBC treating glutaldehyde and RAS	98.7	202.8	200.1	2.00	Brar and Gupta (2000)
2-fluorophenol	100	MC augmented with 2-fluorophenol degrader (FP1) MC from RBC	82.0	4.8	3.9	0.78	Duque et al. (2011)
1,5-pentanedial (Glutaldehyde)	180	MC from settled sewage	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	<i>Rhodococcus</i> sp. S2	45.7	4660.3	2129.7	0.77	Amorim et al. 2013
4-fluorocinnamic acid	35	<i>Rhodococcus</i> 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	<i>Ulothrix sp.</i>	36.7	18.243	6.695	-	1	Orandi et al. 2012
Co	0.5	<i>Ulothrix sp.</i>	5.7	0.203	0.012	-	1	Orandi et al. 2012
Cu	100	<i>Ulothrix sp.</i>	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	<i>Ulothrix sp.</i>	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	<i>Ulothrix sp.</i>	35.2	0.016	0.006	-	1	Orandi et al. 2012
Sb	0.007	<i>Ulothrix sp.</i>	35.6	0.003	0.001	-	1	Orandi et al. 2012
Ni	3	<i>Ulothrix sp.</i>	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	$\delta = \phi^{0.5} \omega^{1.5} s^1$		(Zeevalkink et al., 1978)
Overall oxygen transfer (OT) considering film theory	$K_L = -2 \left(\frac{D_L}{\pi t_R} \right)^{0.5} \left((1 - 4.21) \frac{\delta}{D_L T_R} \right)^{0.5}$		Zeevalkink et al. (1979)
Overall OT to bulk	$\ln K_L = 1.31 \ln \omega + 14.78$	OT governed by disc rotation alone	Friedman et al. (1979)
	$K_L = 2 \left(\frac{D_L}{\pi t_R} \right)^{0.5}$	Where $\delta/D_L t_R \geq 1.7$	
Overall OT considering film theory	$K_L = 2 \left(\frac{2\alpha}{\pi^{0.5}} \right) \cdot \frac{\delta}{t_R} \sim \frac{\delta}{t_R}$	Where $\delta/D_L t_R < 0.8$	Bintanja et al. (1975)
Overall OT	$K_L \frac{\phi}{D_L} = K \left(\frac{\omega' \phi^2 \rho}{\mu} \right)^l \left(\frac{\omega'^2 \phi}{g} \right)^m \left(\frac{\phi - \phi_0}{\phi} \right)^n$	$K = 1.7, l = 0.8, m = 0.13, n = 0.74$	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	$K_L a = 0.0003 \left(\frac{N A_d \delta \omega}{V} \right) + 0.0119$	Where $N_v : < 800$	Kubsad et al. (2004)
	$K_L a = 0.0001 \left(\frac{N A_d \delta \omega}{V} \right) + 0.1157$	Where $N_v > 800$	
Non-dimensional model of $K_L a$	$\frac{(K_L a \rho A_d)}{\mu} = \left(\frac{\phi}{A_d^{0.5}} \right)^\psi \left(\frac{\rho A_d \omega}{\mu} \right)^\varepsilon \left(\frac{A_d}{A_t} \right)^\theta \left(\frac{\delta}{V^{0.33}} \right)^\lambda$	$\Psi = 0.327, \varepsilon = 1.018, \theta = 0.743, \lambda = 0.624$	Chavan and Mukherji (2008)
Model of Oxygen transfer	$K_L a = \alpha \cdot \omega^{1.5} \cdot \phi^{0.5} \cdot (\beta/\omega + \gamma)$	where α, β, γ are constants that need defining	Di Palma et al. (2003)
Experimentally verified model from above	$K_L a = 134.07 \cdot \omega^{1.5} \cdot \phi^{0.5} \cdot (2.15/\omega) + 0.006$	Model only valid providing enhancement factor is described.	Di Palma et al. (2009)

Rotating biological contactors for wastewater treatment - A review

Hassard, Francis

2014-08-04

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

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