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Title

OPTIMIZATION OF TWO-STAGE HIGH-RATE ANAEROBIC REACTORS COUPLED WITH BAFFLED POND AND WET-DRY SAND FILTERS FOR DOMESTIC WASTEWATER TREATMENT IN A WARM-DRY CLIMATE (OUAGADOUGOU, BURKINA FASO)

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Thesis

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This research was conducted under the framework of Sanitation for the Urban Poor project: ''**Stimulating Local Innovation on Sanitation for the Urban Poor in Sub-Saharan Africa and South-East Asia**'' under the auspices of the project Director **Prof. Damir Brdjanovic** of UNESCO-IHE Institute for Water Education, Delft, the Netherlands.

Dedication

This thesis is dedicated to my late father Moumouni Ali
May God bless and rest your soul in peace

Abstract

Over recent decades, there is renewed interest in optimizing and innovating wastewater treatment technologies (WTTs) in sub-Saharan Africa, to reduce the impact of domestic and industrial sewage on the environment. However, poor city-dwellers need low-cost, reliable WTTs that allow for the safe reuse of the effluent in water scare context. This research focuses on the design, implementation, evaluation and optimization of two options for domestic wastewater treatment in the warm, dry sub-Saharan Africa climate of Ouagadougou, Burkina Faso. The first option consisted of two-stage high-rate Anaerobic Reactors followed by a Baffled Pond (AR-BP) with recycled plastic media as a medium for attached growth. The three vertical plastic baffles (with plastic bottle caps affixed to them to increase their surface area) formed four compartments in the baffled pond (BP). The second option included the same two-stage high-rate Anaerobic Reactors but followed them with wet-dry Sand Filters (AR-SF). The research was conducted on the pilot scale, by applying a design flow of 1 m^3/day, which was later increased to 1.5 m^3/ day. A peristaltic pump was used to provide an intermittent flow three times a day (at 8:00 am, 1:00 pm and 5:00 pm) from the buffer tank to the system.

After two years of operation, COD, BOD_5 and TSS mean removal efficiencies were achieved by significant difference in both systems : 79%, 81% and 72% for AR-BP; 84%, 88% and 88% for AR-SF respectively. It was also found out that high pathogen removal efficiencies were achieved in both treatment options with 6 and 5 log units for AR-BP and AR-SF respectively. In addition, the AR-SF option presented a high rate of nitrification, while the BP was more efficient in removing ammonia nitrogen (84%) and *E. coli* (6 log units). Furthermore, no *E.coli* were ever detected in the BP effluent, nor did clogging occur in the SF, during the entire study. *E-coli* were, however, found in the effluent of a control pond (CP) that had no baffles. In fact, it was found that *E. coli* concentrations were lower in the upper layers of all four compartments of the BP, with an undetectable level in the last compartment down to a depth of 0.60 m. A tracer test with salt results showed actual mean hydraulic retention times of 4.1 and 3.2 days for BP and CP respectively. Also, it was found that the volume of the pond was more efficiently used for wastewater treatment in the BP, since more half of the volume of the CP was estimated to be inactive. The tracer experiment also showed that there was better mixing in the BP, thus treatment would be more predictable. Consequently, incorporating three verticals baffles in a pond, under Sahelian climate, not only improved the hydrodynamics and the performance of the pond, but also reduced costs and the amount of land that is required.

Another important aspect revealed by this research was the dense and rich biodiversity on both the attached media and in the water column of the BP. The biofilm was thick and green on the upper parts of both sides of all three at the top of the two sides of the baffles (on both the plastic sheets that form the baffles and the plastic bottles caps affixed to it). The biomass attached on the media constituted 35.5 times of that in the water column. Three major groups of diverse zooplankton were found in the water column at 15-90 cm depth, which included Cladocera, Copepoda and rotifers. The latter group was dominant with 13 identified species, which are attracted to a wide spectrum of food items. In addition, the Principal Components Analysis (PCA) carried out to examine the interactions between biotic and abiotic components of BP further revealed the symbiotic algal-bacterial activity and abiotic parameters, such as pH, dissolved oxygen and temperature interdependences in the course of organic matter degradation in the top layer of the BP. Furthermore, the very strong negative correlation between zooplankton and phytoplankton associated with abiotic parameters corroborates their predatory relationship. As a result, the predatory symbiosis distributions of phytoplankton and zooplankton have shown that the baffles had an effect on water quality which in turn has affected the ecology of the BP. Moreover, this dense and abundant presence of the zooplankton community could play an important role in the control of bacterial and algal populations in BP.

Lastly, the two-stage high-rate anaerobic reactors (R1 and R2) produced ample amounts of valuable biogas under the warm Sahelian climate, with 9.7 L/m^2 per day of biogas and with methane content of 54%. More importantly, very low sludge yields were recorded in R1, R2, and BP (0.0006, 0.0002 and 0.0014 m^3/capita/year respectively), thus reducing the cost of its extraction and management.

Both treatment options can be recommended as an alternative low-cost wastewater treatment technologies, separately or in tandem, for African cities, with the final effluent being used for restricted irrigation in peri-urban agriculture. To contribute even more to the alleviation of hunger in poor neighborhoods, further investigations may look at the use of this effluent in aquaculture, before its use in irrigation.

Keywords: anaerobic reactor; baffled pond; biofilm; biogas; domestic wastewater; low-cost technology; wastewater treatment; sustainable sanitation for urban poor; water recycling; wet-dry sand filter

List of Publications

Dissertation submitted for the degree

I. Title

Optimization of two-stage High-rate Anaerobic Reactors coupled with Baffled Pond and Wet-dry Sand Filters for domestic wastewater treatment in a warm-dry climate (Ouagadougou, Burkina Faso)

II. Published Papers:

D.A. Moumouni, H.A. Andrianisa, Y. Konaté, A. Ndiaye and A.H. Maïga (2015): Inactivation of *Escherichia coli* in a baffled pond with attached growth: treating anaerobic effluent under the Sahelian climate, Environmental Technology, DOI: 10.1080/09593330.2015.1098732

III. Submitted Papers and Manuscripts:

- D.A. Moumouni, H.A. Andrianisa, Y. Konaté, A. Ndiaye and A.H. Maïga (…): Alternative low-cost wastewater treatments for sub-Saharan Africa urban poor, Environmental Engineering Science (submitted manuscript ID EES-2015-0435)

- D.A. Moumouni, H.A. Andrianisa, Y. Konaté, A. Ndiaye and A.H. Maïga (…): Effects of baffles on biofilm characteristics and zooplankton composition in a Baffled Pond in the Sahel Region of Africa, (manuscript)

- D.A. Moumouni, H.A. Andrianisa, Y. Konaté, A. Ndiaye and A.H. Maïga (…): Hydraulic regimes of baffled and unbaffled, treating anaerobic effluent under the Sahelian climate, (manuscript)

IV. Conference Papers:

- D.A. Moumouni, H.A. Andrianisa, Y. Konaté and A.H. Maïga (2013): Performance estimation of two-stage high rate anaerobic reactors coupled with baffled pond and wet-dry sand filters for domestic wastewater treatment in a warm climate: the case of Ouagadougou. 7th2iE International Scientific Days, 1-4 April 2013, Ouagadougou, Burkina Faso.
- D.A. Moumouni, H.A. Andrianisa, Y. Konaté and A.H. Maïga (2013): Performance estimation of Two Stage High-Rate Anaerobic Reactors coupled with Baffled Pond and Wet-Dry Sand Filters for Domestic Wastewater treatment In a Warm climate. 3rdIWA DevelopmentCongress and Exhibition, 14-17 October 2013, Nairobi, Kenya.

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Résumé substantiel en Français

Optimisation de deux Réacteurs Anaérobies à haut rendement, suivis de deux options de post-traitement des eaux usées domestiques sous un climat sahélien chaud et sec: Bassin Lamellé et Filtre à Sable (Ouagadougou, Burkina Faso)

Introduction

Au cours des dernières décennies, les techniques de traitement des eaux usées par les procédés du système extensif, et particulièrement le lagunage à microphytes, ont connu une véritable évolution dans les pays tropicaux où le climat est favorable. Ces technologies de traitement des eaux usées à faible coût sont non seulement fiables, efficaces, durables mais aussi adaptées, aux populations à faible revenu vivant dans les zones urbaines et péri-urbaines de l'Afrique Subsaharienne. Les déficits hydriques sont récurrents dans ces zones où la rareté des ressources en eau a des répercussions importantes sur l'économie, l'alimentation et la santé des populations. Pour pallier ce manque, les eaux usées sont utilisées et réutilisées en agriculture avec ou sans traitement préalable entrainant ainsi des problèmes de santé publique. Pour réduire les risques dus à la réutilisation de ces eaux, plusieurs techniques innovantes de traitement des rejets domestiques et urbains ont été développées. La méconnaissance des conditions d'usage et de maintenance de ces systèmes remet en cause leur viabilité. Il est donc nécessaire de proposer des systèmes de traitement qui tiennent compte au mieux les réalités de la zone d'étude. Ces systèmes à moindre coût de conception et de maintenance, ne doivent pas avoir une forte emprise sur le sol. C'est dans cette optique que cette étude a été menée sur la conception, la mise en œuvre, l'évaluation et l'optimisation de deux options de traitement des eaux usées domestiques sous le climat sahélien de Ouagadougou au Burkina Faso. Le choix de ces deux options est basé sur les concepts de technologies extensives à faible coût, tant au niveau de la collecte qu'à celui de l'épuration des eaux usées.

La première option comporte deux Réacteurs Anaérobies à haut rendement connectés en série, puis suivis par un Bassin Lamellé avec des bouchons en plastique fixés aux chicanes (RA-BL). Le principe de cette option est basé sur le fonctionnement à trois étages de bassins de lagunage à microphytes. Le premier bassin qui est le bassin anaérobie a été modifié pour former deux réacteurs anaérobies, où le biogaz est collecté. Le second bassin, dit bassin facultatif a été omis afin de minimiser la zone d'emprise du système de traitement. Enfin, pour optimiser l'efficacité hydraulique et augmenter la biomasse épuratrice, trois chicanes munies des bouchons de bouteilles en plastique usagées ont été introduites verticalement à contrecourant dans le troisième bassin dit bassin de maturation, formant ainsi quatre cloisons. Cette configuration a été prévue pour améliorer l'efficacité d'élimination des matières organiques, des nutriments (azote et phosphore), des agents pathogènes et permettre une valorisation de la production d'énergie (biogaz).

La seconde option est composée de deux Réacteurs Anaérobies à haut rendement couplés à deux filtres à sable (RA-FS) à fonctionnement alterné. Cette alternance d'alimentation du filtre vise à éviter le colmatage. Cette filière de traitement est similaire à la fosse septique conventionnelle mais en diffère en raison de la collecte du biogaz et de la qualité potentielle de ses effluents. Par conséquent, cette technologie pourrait être une meilleure variante de la fosse septique standard.

Objectifs de l'étude

L'objectif général de cette thèse est d'optimiser l'efficacité épuratoire du système de traitement des eaux usées domestiques sous le climat sahélien chaud et sec de Ouagadougou par deux Réacteurs Anaérobies à haut rendement suivis de deux options de post-traitement : Bassin Lamellé et Filtre à Sable. Ainsi, cette étude contribuera à atténuer l'impact des rejets des ouvrages d'assainissement autonome sur la qualité des ressources en eau et celle de l'environnement, tout en remediant aux problèmes de santé publique.

Objectifs spécifiques

De façon plus spécifique, il s'agit :

- de concevoir, mettre en œuvre et évaluer la performance épuratoire des technologies alternatives novatrices et durables à une échelle pilote, en terme d'élimination des matières organiques, des nutriments et des agents pathogènes pour les communautés à revenues limitées du Sahel;
- d'évaluer la performance hydraulique du bassin lamellé comparée au bassin sans lamelle;
- de comprendre l'absence d'*Escherichia coli* dans l'effluent du bassin lamellé ;
- d'évaluer la biodiversité algale et zooplanctonique développée sur les bouchons de bouteilles en plastique fixés aux lamelles et dans la colonne d'eau du bassin lamellé ;
- d'estimer le potentiel de production du biogaz, sa composition et le taux d'accumulation des boues dans les deux réacteurs anaérobies à haut rendement.

Portée de l'étude

Le document de thèse est structuré en 7 parties :

La première partie est une introduction générale sur l'importance du traitement des eaux usées et leurs réutilisations pour une gestion durable de l'environnement dans le contexte Africain. Elle présente un aperçu des principales options de traitement des eaux usées, avec un accent particulier sur les options à faible coût, leurs limites et les différentes combinaisons de procédés anaérobie et aérobie. En outre, elle décline les objectifs et justifie le choix du thème.

La deuxième partie détaille les différents aspects pris en compte dans la conception et la mise en œuvre du projet pilote. Cette partie présente également les résultats de l'évaluation de la performance épuratoire de deux options de traitement, en faisant varier les temps de séjour hydraulique des deux réacteurs anaérobies à haut rendement. Les résultats sont analysés et comparés à ceux obtenus dans la littérature.

La troisième partie compare les résultats de la caractérisation hydraulique du bassin lamellé à ceux du bassin sans lamelle (témoin) à travers un test de traçage au chlorure de sodium. En outre, elle se concentre sur les caractéristiques et les modèles hydrauliques qui pourraient être appliqués dans la prédiction de la performance des bassins en termes d'élimination de la matière organique et des agents pathogènes.

La quatrième partie met en évidence les mécanismes d'élimination des *Escherichia coli* au niveau du bassin lamellé. En outre, les taux d'inactivation d'*E. coli* dans les deux bassins ont été déterminés sur la base du modèle hydraulique obtenue à partir de l'essai de traçage (**partie 3**). L'importance des lamelles est également décrite dans cette partie.

La cinquième partie se concentre principalement sur les caractéristiques du biofilm qui s'est formé dans le bassin lamellé, en termes de répartition de la biomasse algale, la diversité microbienne, et la composition des espèces de zooplancton et leur distribution dans la colonne d'eau et sur les médias des lamelles. En outre, un outil statistique d'analyse en composantes principales, a été utilisé pour mettre en exergue les corrélations entre les phytoplanctons, les zooplanctons, les bactéries et les matières en suspension.

La sixième partie présente et discute de la possibilité de la production de biogaz, sa qualité, et le taux d'accumulation des boues dans les deux réacteurs anaérobies à haut rendement.

Enfin, la septième partie de cette thèse se termine par quelques conclusions générales, ainsi que des perspectives pour l'avenir de l'assainissement décentralisé en Afrique.

Méthodologie

➢ **Conception et mise en œuvre des deux filières de traitement**

Les principes de base qui ont guidé à la conception des deux étages de réacteurs anaérobies à haut rendement suivis de deux options de post-traitement (Bassin Lamellé et Filtre à Sable) sont ainsi résumés:

- une combinaison optimale des procédés de traitement à faible coût ; anaérobie et aérobie, à partir de laquelle des effluents de haute qualité peuvent être obtenus permettant également la récupération de l'eau, des nutriments et la valorisation énergétique ;
- les unités de traitement choisies devraient être adaptées au contexte local tout en considérant des matériaux de construction disponibles localement;
- le système anaérobie a été conçu sur la base des concepts de bassin anaérobie, réacteur anaérobie à flux ascendant (UASB : upflow anaerobic sludge blanket) et de fosses septiques, puis l'option optimale a été adoptée;
- la station pilote a été conçue pour servir la communauté urbaine à faible revenu avec environ 50 équivalents habitants et chaque individu pourrait générer 40 litres d'eau usée par jour (Maiga et al. 2014) ;

- initialement, un débit journalier de 1 m^3 a été considéré, puis il a été augmenté progressivement à 1,5 m^3 afin de déterminer l'état optimal de fonctionnement de la station pilote ;
- une température moyenne pour le mois le plus froid de 25 ^{0}C a été adoptée;
- une concentration dans l'influent brut en coliformes fécaux de 10^6 UFC / 100 mL et la demande biochimique en oxygène (DBO) de 250 mg / L ont été supposées (Maiga et al. 2006);
- trois lamelles verticales avec 70 % de la longueur de la profondeur du bassin ont été adoptées ;
- une vitesse d'infiltration de 0,02 m par heure pour une superficie maximale de lit de sable de 1 m^2 a été adoptée pour la conception du filtre à sable ;

La combinaison de ces critères a abouti à la mise au point de deux filières de technologies à faible coût destinées aux communautés aux revenus limités dans un climat Sahélien. Cette technologie est dénommée : Deux-étages de réacteurs anaérobies à haut rendement suivis d'une part, par un Bassin Lamellé avec des bouchons en plastique fixés aux chicanes et d'autre part, par deux lits de filtration à sable à fonctionnement alterné. Les filières de traitement ainsi conçues ont été installées au sein du campus de l'Institut International d'Ingénierie de l'Eau et de l'Environnement (2iE) à Ouagadougou au Burkina Faso.

- **Suivi de la performance épuratoire des deux filières de traitement**

La recherche est menée à l'échelle pilote. La station pilote a été exploitée sous deux conditions dénommées période 1 (P1) et période 2 (P2). Au cours de P1, du 8 mai 2013 au 6 mai 2014, un débit journalier de 1 m^3 était pompé de façon intermittente en trois moments (à 8 h: 00, 13 h : 00 et 17 h : 00). Les temps de rétention hydraulique théorique (TRH) du premier réacteur anaérobie (R1), du second réacteur anaérobie (R2) et du bassin lamellé (BL) pendant P1 étaient respectivement de 1,5 ; 1,5 et 7 jours, tandis que le temps d'infiltration du filtre à sable (FS) était d'environ 5 minutes. Il convient de noter qu'au cours de P1, le bassin témoin (BT) n'a pas encore été construit. Durant P2, du 13 mai 2014 au 12 mai 2015, le débit journalier de l'influent des eaux usées a été porté à 1,5 m^3 par jour en 3 fois (à 8h : 00, 13 h : 00 et 17 h : 00). Par conséquent, les TRH de R1 et R2 ont été réduits à 1 jour chacun, alors que celui de BL a été maintenu à 7 jours. D'autre part, le BT a été mis en service, tandis que la surface filtrante du filtre à sable a été réduite de moitié. L'objectif principal visé par l'augmentation de débit de l'influent était de réduire la surface du lit du filtre à sable mais aussi d'estimer les conditions optimales de fonctionnement de la station pilote.

Les prélèvements ont été effectués de façon ponctuelle entre 8 h et 9 h. Des mesures d'indicateurs de qualité des eaux ont été effectuées pendant deux ans sur des échantillons prélevés suivant une fréquence hebdomadaire. Les échantillons ont été prélevés au point de l'entrée de l'influent noté EB (eaux brutes), à la sortie de R1, R2, BL, FS et BT afin d'évaluer la performance épuratoire à chaque étape de traitement, de même que l'ensemble des deux options. Les échantillons destinés aux analyses physico-chimiques ont été collectés dans des flacons de 500 ml en polyéthylène et les échantillons réservés pour les analyses bactériologiques ont été prélevés dans des flacons en verre borosilicaté de 500 ml préalablement stérilisés à 150 °C pendant une heure. Les échantillons prélevés ont été immédiatement rangés dans une glacière et conservés à une température de 4 °C, puis sont transportés au laboratoire 2iE pour les analyses avant 24 heures selon la méthode standard APHA (2012).

- **Etude de la stratification de la biomasse du bassin lamellé**

Pour étudier la stratification qui se développe dans le bassin lamellé, quelques paramètres physico-chimiques et bactériologiques ont été analysés *in situ* et au laboratoire en utilisant la méthode standard APHA (2012). Le pH, l'oxygène dissous, la température, la conductivité électrique, la DCO, la Chlorophylle (a) et les coliformes fécaux dont *E. coli* étaient les paramètres considérés. Les paramètres *in situ* (pH, température oxygène dissous, et la conductivité) ont été mesurés à sept (7) différents niveaux (15, 30, 45, 60, 75, 90 et 105 cm de profondeur) par compartiment et ceux-ci trois (3) fois par jour. Les échantillons destinés aux analyses du laboratoire ont été prélevés respectivement à 15 ; 60 et 105 cm de profondeur dans chaque compartiment du bassin une fois dans la semaine.

- **Quantification et distribution de la biomasse planctonique et sessile sur les chicanes et les bouchons du bassin lamellé**

Après deux années de fonctionnement, l'échantillonnage pour la quantification et la distribution de la biomasse planctonique et sessile a été fait de façon ponctuelle entre 08h : 00 et 09h : 00 à trois profondeurs différentes (15, 60 et 90 cm) dans la colonne d'eau et dans chaque compartiment (A, B, C et D) du bassin lamellé. Ensuite, une seconde étape d'échantillonnage a été menée immédiatement après avoir soigneusement vidé le bassin lamellé. La biomasse amassée sur les parois (longitudinales et transversales) du bassin et les trois lamelles immergées (les deux faces avec les bouchons) a été collectée par raclage minutieux à l'aide des spatules sur une surface de 0.01 m^2 à des profondeurs de 15 cm, 60 cm et 90 cm. Les échantillons ont été recueillis et conditionnés dans des bouteilles en verre borosilicaté puis transportés au laboratoire. En plus des paramètres *in situ* (pH, température oxygène dissous, et la conductivité), le poids sec, le poids humide du biofilm, la chlorophylle a, les matières en suspension, la biomasse microbienne (bactérienne et fongique) et la biomasse zooplanctonique ont été analysés.

Les échantillons de plancton récoltés ont été analysés au Laboratoire de Biologie et Ecologie Animale (LBEA) de l'Université de Ouagadougou (Burkina Faso). Les observations ont été faites dans un volume de 0,5 ml d'échantillon et analysés entre lame et lamelle au microscope optique. L'opération de comptage a été répétée quatre (04) fois pour le même échantillon pour optimiser la qualité des résultats. Des clés de détermination et des catalogues d'identification ont été utilisés pour identifier les spécimens rencontrés. Ce sont ceux de Koste et Voigt (1978) ; Pontin (1978) ; de Pourriot (1980) et Hamadi et al, (2011) pour les Rotifères. Dussart (1980) pour les copépodes et les ouvrages de références: Korinek

(1984), Notemboomram (1981), Rey et Saint-Jean (1980), Amoros (1984) pour les Cladocères.

- **Accumulation de boues dans R1, R2 et BL**

La répartition des boues accumulées au fond des réacteurs anaérobies à haut rendement (R1 et R2) et du bassin lamellé (BL) a été définie par la méthode de serviette blanche dite « White Towel ». La méthode consiste à introduire verticalement au fond des réacteurs, une tige en bois enrobée d'une serviette blanche. L'épaisseur de boue mesurée est clairement visible sur le tissu de la serviette après l'avoir doucement retiré des eaux usées. Un décamètre a été utilisé pour mesurer l'épaisseur de boue correspondante (Llyod & Vorkas, 1999; Mara, 2004; Konate et al, 2013). Grace aux coordonnées des points d'échantillonnage et la version 8 du logiciel Surfer, la distribution spatiale des boues en 3D au fond du bassin a été reproduite.

- **Collecte et analyse de biogaz**

La production du biogaz a été mesurée quotidiement à partir d'un dispositif de collecte de biogaz de forme géométrique assimilé à un cylindre ayant un volume de 0,024 m^3, lequel ètait soutenu par une tige coulissant verticalement à l'aide d'un ressort. Ainsi, le collecteur remonte à la surface de l'eau grâce à la pression qu'exerce le biogaz sur le ressort sensible à cette poussée de gaz. Une fois le collecteur est en surface de l'eau, la lecture du volume de biogaz peut se faire à l'aide de la graduation sur le collecteur. Ainsi, le volume du gaz collecté après 24 h a été mesuré avec une échelle graduée qui a été établie au-dessus du collecteur. De plus, le volume du biogaz collecté a été corrigé à 20°C et à la température de 1 atm suivant la formule des gaz parfaits **PV= nRT.**

GA 5000 est l'appareil utilisé pour mesure la composition du biogaz dans les deux réacteurs R1 et R2. En effet, les analyseurs de gaz de la série 5000 (GA 5000) sont conçus pour mesurer la qualité des gaz des sites d'enfouissement et d'autres sources (digesteurs anaérobies), et le matériel est certifié uniquement pour une utilisation à température ambiante comprise entre -10 ºC et +50 ºC et ne doit pas être utilisé en dehors de cette plage. Il convient de noter que la pression d'entrée ne doit pas dépasser +/- 500 mbar par rapport à la pression atmosphérique et la pression de sortie ne doit pas dépasser +/- 100 mbar par rapport à la pression atmosphérique. La calibration de l'appareil a été effectuée par différents types de gaz que l'on peut lire à l'écran, ce sont le méthane (CH_4), le dioxyde de carbone (CO_2), l'hydrogène sulfurique (H_2S) et autres gaz.

Résultats et Discussion

Caractéristiques des eaux usées brutes et des effluents traités

Les caractéristiques de l'influent brut admis en tête de la station pilote et les effluents de chaque procédé de traitement, ainsi que les charges organiques volumiques / surfaciques qui ont été analysées pour les 2 ans de fonctionnement sont conformes aux données de littérature (Metcalf et Eddy, 2003; von Sperling et Chernicharo, 2005; Henze et al. 2008; Khan et al. 2013).

Les valeurs d'*E. coli* en moyenne 10^7 UFC /100 ml dans les eaux usées brutes pour les périodes 1 et 2 restent dans la gamme des valeurs de référence pour les eaux usées d'origine domestique (Metcalf et Eddy, 2003). Cependant, durant ces deux périodes de suivi, une forte variabilité d'*E. coli* et des coliformes fécaux a été observée dans l'influent brut et dans les effluents de chaque procédé de traitement, à l'exception de l'effluent du Bassin lamellé où aucune souche d'*E. coli* n'été détectée. Cette situation de forte variabilité pourrait s'expliquer par l'incidence des personnes infectées dans la communauté (campus 2iE), la saison de l'année (chaud ou froid), la période et la méthode d'échantillonnage (échantillonnage ponctuel aux heures de pointe entre 8 h : 00 et 9 h : 00), le statut socio-économique des populations qui contribuent à la production d'eaux usées, la faible consommation d'eau par habitant, tel que discuté amplement par Oliveira et von Sperling (2006) et Henze et al. (2008).

Dans le même temps, les valeurs moyennes de DCO, DBO_5 et MES des eaux usées brutes pour les périodes 1 et 2 ont été estimées à 424 et 425 mg/l, à 252 et 255 mg/l, et à 148 et 134 mg/l respectivement. Il en résulte que ces valeurs sont en deçà des gammes habituellement espérées dans les eaux usées domestiques des pays en voie de développement (Metcalf et Eddy, 2003 ; von Sperling et Chernicharo, 2005; Henze et al, 2008; Khan et al, 2013). Cela pourrait s'expliquer par l'effet de dilution des eaux usées du campus de 2iE, car aucun dispositif économiseur d'eau n'était en cours d'utilisation. Cependant, ces résultats montrent un rapport de $DCO/DBO_5 < 2$, d'où ces eaux usées d'origine domestique sont facilement biodégradables (Metcalf et Eddy, 2003). Par ailleurs, de forte variations de DCO, DBO_5 et MES a été observée à chaque niveau du processus de traitement et pour les deux périodes, ce qui reflète une bonne réponse de la station pilote. Par exemple, les coefficients de variation de la DCO, DBO_5 et MES des eaux usées brutes au cours de la période 1 étaient respectivement de 47 %, 34 % et 55%. Ce constat est confirmé par des études antérieures menées par Maiga et al. (2006) et Konaté et al. (2013) sur le même site d'étude. Toutefois, cette situation peut s'expliquer par les activités au sein du campus, marquée par la mobilité du personnel et des étudiants.

La variation des valeurs de la température, de l'oxygène dissous et du pH dans les eaux brutes et à la sortie de point des unités de traitement, sont dans la gamme favorable au bon développement des microorganismes épurateurs de la matière organique (Metcalf et Eddy, 2003). En outre, les pH des deux réacteurs anaérobies sont dans l'intervalle favorable au développement des bactéries méthanogènes (Peña, 2002; Foresti et al. 2006). Cependant, le pH reste élevé (entre 8 et 9,8 pour les périodes 1 et 2) dans l'effluent du bassin lamellé, qui est le résultat de la symbiose algues microorganismes (Curtis et al. 1992; Kayombo et al. 2002). Contrairement au bassin lamellé, des faibles valeurs de pH (entre 3,9 et 6,7 durant P1 et P2) ont été obtenus dans l'effluent du filtre à sable. Cela peut être dû à l'alimentation intermittente et à la libération des H+ qui acidifient le milieu et réduit le pH durant de la forte nitrification observée dans le filtre à sable (Metcalf et Eddy, 2003).

Par ailleurs, la valeur de la température moyenne de l'influent brut est passée de 29 à 31 °C à la fois dans R1 et R2, puis a été réduite à 28 °C dans le filtre à sable et le bassin lamellé. Par conséquent, l'utilisation de réacteurs anaérobies peints en noir dans le climat ensoleillé du Sahel a entraîné une augmentation de la température de 2 °C durant toute l'année.

Des faibles valeurs d'oxygène dissous avec une faible variabilité ont été enregistrées dans les eaux usées brutes et les effluents de R1 et R2 ; ce qui démontre les bonnes conditions anaérobies de ces réacteurs. En revanche, des valeurs élevées d'oxygène dissous ont été observées à la fois dans le filtre à sable et dans le bassin lamellé pendant les deux périodes de suivi. Toutefois, cette situation peut s'expliquer par l'activité photosynthètique des algues et l'aération induite par la disposition des chicanes dans le bassin. Ces résultats sont conformes à ceux des études de Olukanni et Ducoste (2011) et Bolton et al. (2010). Dans le cas du filtre à sable, ces valeurs pourraint être liées à la ré-oxygénation des pores du sable entre les alimentations (en moyenne 5 heures), ce qui donnerait suffisamment de temps pour drainer le filtre.

Les concentrations moyennes en NH_3 -N dans les eaux usées brutes sont passées de 36 à 38 mg/l dans R1, puis ont diminué légèrement à 37 dans R2. Ces valeurs croissantes sont semblables à celles trouvées par Foresti et al. (2006) qui ont rapporté des valeurs de 30 mg/l dans l'influent brut et 50 mg/l de NH_3-N dans l'effluent d'un réacteur anaérobie. Khan et al. (2013) ont expliqué que cette augmentation de NH_3-N pourrait être due à l'hydrolyse de l'azote organique dans le processus anaérobie. En revanche, dans le bassin lamellé, les concentrations de NH_3 -N sont subitement passées de 37 mg/l dans l'influent à 5 mg/l dans l'effluent. Camargo-Valero (2008) a constaté que, selon les caractéristiques des bassins de lagunages et des conditions climatiques locales, les mécanismes et les voies par lesquelles l'azote sous ses diverses formes est éliminé pourrait être attribué à la volatilisation de l'ammoniac, à la sédimentation de l'azote organique par l'intermédiaire de l'absorption biologique, à sa rétention dans les boues au fond du bassin, à la nitrification – dénitrification et à l'assimilation du nitrate et de l'ammoniac par les algues. Cependant, des études menées ulterierement ont montré que seulement 2% de l'azote ammoniacal global pourrait être éliminé par volatilisation (Camargo-Valero et Mara, 2007a, 2010; Assunção et von Sperling 2012; Bastos et al. 2014).

De cette étude, il apparaît que les concentrations de nitrates dans les eaux usées brutes à la fois pour P1 et P2 respectivement de 3,5 et 4,7 mg/l ont été successivement réduites à 2,6 et 3 mg/l dans R1, à 1,7 et 2 mg/l dans R2, et 1,06 à 1,02 mg/l dans le bassin lamellé. Les causes possibles peuvent être des nitrates qui ont été transformés par d'autres organismes présents dans les unités de traitement sous d'autres formes d'azote (Metcalf et Eddy, 2003; Camargo-Valero, 2008; Babu, 2011). Contrairement au bassin lamellé, les concentrations de nitrates ont augmenté de manière significative de plus de 34 et 49 fois respectivement pendant les périodes 1 et 2 dans le filtre à sable. Ce fait pourrait être dû à l'alimentation intermittente où à une importante ré- oxygénation qu'a lieu dans le milieu poreux entre les deux alimentations du filtre à sable.

Similairement à NH_3-N, les concentrations d'ortho phosphate pour les deux périodes ont augmenté successivement de 9,9 et 10,5 mg/l de l'influent brut, à 12,2 et 12,9 mg/l dans l'effluent R1, et à 14,9 et 13,9 mg/l dans l'effluent R2. Cependant, dans les deux post-traitements aérobies (BL et FS), les concentrations en ortho phosphates sont réduites à 3,8 et 5,2 mg/l respectivement pendant P1. Les mécanismes d'élimination pourraient être dus à des activités microbiennes selon les conditions d'anaérobiose suivi d'aérobiose (Henze et al. 2008; Khan et al. 2013).

En résumé, les analyses statistiques ont montré que la diminution du temps de séjour hydraulique de 1.5 jours à 1 jour dans les réacteurs anaérobies a une influence sur les concentrations des effluents de ces réacteurs. Statistiquement, il y avait de différence significative entre les deux filières de traitement RA-BL et RA-FS et entre les deux périodes ($p > 0,05$), en matière de concentrations en MES, DBO_5, NH_3-N, NO_3-N, PO_4-P et E. coli sauf pour DCO.

Performances épuratoires de la station pilote

Après deux années de fonctionnement, les rendements épuratoires moyens obtenus montrent une bonne élimination de la pollution organique et des particules en suspension : 79%, 81% et 72% dans la filière RA-BL contre 84%, 88% et 88% dans la filière RA-FS respectivement en DCO, DBO_5 et MES. En termes d'abattement microbien, les deux options se révèlent plus efficace : 6 et 5 unités log d'élimination d'*Escherichia coli* respectivement pour RA-BL et RA-FS. La filière RA-BL élimine 84% de NH_3-N tandis que RA-SF ne peut abattre que 64%. En outre, ces rendements moyens d'abattement de la pollution sur la période de suivi étaient dans la gamme rapportée par d'autres auteurs avec des options de traitement dans des conditions climatiques similaires (Kilani et Ogunrombi, 1984; von Sperling et al. 2002, 2003; Shilton et Mara, 2005; Banda, 2007).

Les analyses statistiques ont révélé qu'il y avait de différence significative entre les deux filières de traitement RA-BL et RA-FS et entre les deux périodes ($p > 0,05$), en matière de rendements épuratoires pour la plupart des paramètres qui ont été analysés, à l'exception de la DCO. Les concentrations résiduelles en matières organiques et pathogènes de l'effluent traité répondent aux normes recommandées par l'Organisation Mondiale de la Santé (OMS, 2006) pour une réutilisation des effluents en agriculture non restrictive. Au vue, de cette bonne performance épuratoire, ces deux options de traitement peuvent être considérées comme des technologies alternatives de traitement des eaux usées à faible coût pour les populations à faible revenu dans les zones urbaines et péri-urbaines en Afrique Subsaharienne.

Performance hydraulique du bassin lamellé comparée au bassin sans lamelle

Les essais de traçage ont été effectuées sur le bassin lamellé (BL) et le bassin sans lamelle (BT) en utilisant comme traceur le sel de cuisine, afin de déterminer avec les courbes de restitution, le temps de séjour hydraulique réel, la vitesse d'écoulement des eaux usées, le coefficient de dispersion, l'efficacité volumétrique et le modèle hydraulique de chacun des bassins. En outre, cette étude a été réalisée pour confirmer l'effet des lamelles avec des bouchons en plastique fixés sur la performance hydraulique d'un bassin lamellé (trois lamelles verticales contrecourant) dans le contexte Sahélien.

Les temps de séjour hydraulique réel moyen étaient 4,1 et 3,2 jours respectivement pour BL et BT, contre un temps de séjour hydraulique théorique de 6,6 jours. Ceci a montré que l'introduction de trois lamelles verticales contre-courant dans un bassin recevant les effluents de deux réacteurs anaérobies pourrait augmenter le temps de séjour moyen d'environ 1 jour, c'est-à-dire une augmentation d'environ 22% du temps de séjour hydraulique réel. L'efficacité volumétrique du bassin lamellé était de 62 %, une zone inactive ‹‹morte›› de 38 % et un coefficient de dispersion de 0.53, contre une efficacité volumétrique du bassin témoin de 49 %, une zone inactive ‹‹morte›› de 51 % et un coefficient de dispersion de 0.66. Ces résultats sont en accord avec ceux rapportés par de Babu (2011) et Shilton et Harrison (2003) en terme de dispersion élevée et d'augmentation du temps de séjour.

Les études hydrodynamiques effectuées sur le bassin lamellé et le bassin sans lamelle de la station pilote ont montré que l'écoulement était du type de réacteur complètement mixte en série et dispersif. Ce qui a permis de déduire que les modèles dispersifs sont plus appropriés pour non seulement simuler le comportement du bassin lamellémais aussi prédire ces performances épuratoires. Par conséquent, ces résultats montrent qu'il existe un potentiel important de réduire l'emprise sur le terrain des ouvrages d'où le coût de la technologie.

Distribution et abattement *d'Escherichia coli* dans le bassin lamellé

Cette étude de la distribution et de l'abattement d'*Escherichia coli* dans le bassin lamellé (BL) révèle de manière générale, une charge très faible en coliformes fécaux à la surface du bassin et une charge relativement élevée au fond du bassin et ceci est observable dans tous les compartiments du bassin. Une décroissance successive de la charge bactérienne s'observe lorsque l'effluent passe d'un compartiment à un autre jusqu'à atteindre une valeur nulle en *E. coli* à la surface du dernier compartiment du bassin. En effet, il a été constaté que les concentrations de *E. coli* étaient plus faibles dans les couches supérieures de l'ensemble des quatre compartiments du BLavec un niveau indétectable (<1 UFC/100 ml) dans le dernier compartiment jusqu'à une profondeur de 0,60 m. Cette évolution de *E. coli* dans BL a confirmé les résultats de l'étude précédente sur le suivi de la performance épuratoire où *E. coli* n'a pas été détecté dans l'effluent du basin durant toute la période d'étude. En outre, ces résultats ont révélé l'avantage de recueillir à la surface les effluents d'un bassin (qualité de l'effluent).

Par ailleurs, il a été constaté qu'il y avait une différence significative dans le coefficient du taux d'abattement d'*E. coli* entre le bassin lamellé et celui sans lamelle. Cela impliquait que les lamelles avec les bouchons en plastique pourraient avoir un rôle important dans non seulement, l'amélioration de l'hydrodynamisme du bassin, mais aussi dans l'abattement d'*E. coli*. La sédimentation combinée avec les effets synergétiques d'autres facteurs environnementaux, physiques, chimiques et opérationnels (intensité du rayonnement solaire, la température, pH, oxygène dissout, profondeur du bassin, biomasse algale, limite en nutriments, hydrodynamique etc…) pourraient être responsable de l'abattement d'E. coli dans ce système (Kilani and Ogunrombi, 1984; Curtis et al. 1992; Davies-Colley et al. 1999; van der Steen et al. 2000; Oragui, 2003; von Sperling et al. 2003; von Sperling, 2005; Abis et Mara, 2006; Davies et al. 2009 ; Maïga et al. 2009; Nelson et al. 2009 ; Bolton et al. 2010; Buchanan et al. 2011; Ukpong, 2013; Ouali et al. 2012, 2014).

Biodiversité algale et zooplanctonique développée sur les bouchons en plastique fixés aux lamelles et dans la colonne d'eau du bassin lamellé

Les biofilms adhérés sur les lamelles, les murs intérieurs du bassin, les bouchons et dans la colonne d'eau ont été quantifiés en matière sèche. Cette adhésion et suspension varient considérablement selon les profils profondeurs (de la surface vers le fond du bassin) et les profils longitudinaux (d'entré vers la sortie du bassin). La densité du biofilm était plus élevée à la surface et décroit progressivement vers le fond du bassin et vers la sortie du bassin. Les densités moyennes étaient décroissantes selon les profondeurs respectivement de 370 g/m^2 à 0.1 g/m^2 sur les lamelles. En plus on observe que le biofilm est dense sur la Face B (contre-courant) des lamelles que celle de la face A (co-courant). Ces résultats sont en conformité avec les études menées par Babu (2011) sur l'effet des lamelles sur la structure de biofilm algal-bactérien dans un bassin lamellé. Le degré d'adhésion des biofilms dépend de plusieurs facteurs dont la composition du biofilm, la nature de support, les facteurs environnementaux, mais aussi le type des eaux usées (Characklis et al. 1990; Esterl et al. 2003; Babu, 2011; Paul, 2012).

En revanche en comparant la biomasse en suspension dans la colonne d'eau avec celle adhérée sur les lamelles, il était évident de constater que celle adhérée (1,5 kg de biomasse) était 36 fois plus importante que celle dispersée dans l'eau (0,04 kg de biomasse). Par conséquent, l'introduction de trois lamelles verticales contre-courant dans un bassin recevant les effluents de deux réacteurs anaérobies à haut rendement pourrait engendrer une augmentation d'environ 267 % la biomasse épuratrice dans un contexte sahélien. Ainsi, cette situation corrobore avec les résultats sur la bonne performance épuratoire du bassin lamellé démontrée plus haut.

Après le dépouillement des échantillons pour les zooplanctons, un total de 19 taxa ont été recensés. Ces organismes planctoniques appartiennent à 9 familles que sont : la famille des ***;*** *Daphnidae, Moinidae, Sididae, Cyclopidae, Diaptomidae* qui font partir de la classe des Crustacés. Puis les familles des *Brachionidaes*, *Testudinellidae*, *Asplanchnidae, Lecanidae,* qui sont des rotifères. De l'analyse quantitative et qualitative de ces échantillons, les Rotifères, les Copépodes, les Cladocères et des ostracodes ont été identifiés comme étant les grands groupes zooplanctoniques qui composent la faune aquatique. D'un point de vue richesse spécifique, les Rotifères (14 Taxa) dominent le peuplement zooplanctonique comme dans la plupart des eaux douce, suivi des Cladocères (4 Taxa), puis viennent les Copépodes (2 Taxa) avec une densité élevée des *Nauplii et des Copépodites.* Ces résultats sont similaires à ceux rapportés par (Ouéda et al. 2010) dans deux lacs de barrages ruraux et (Ouédraogo, 2013) dans des réservoirs urbains au Burkina Faso qui malgré les différentes pressions exercées sur les hydro systèmes, ces organismes sont présents avec des richesses spécifiques assez importantes. La dominance des rotifères s'expliquent par le fait qu'ils ont une grande capacité d'adaptation, ils s'adaptent mieux au milieu pollué (Hamaidi et al. 2008). Les rotifères sont des organismes microscopiques répandus dans les eaux douces et saumâtres. Ces organismes sont quantitativement dominants dans les communautés zoo-planctoniques des lacs et des parties calmes des rivières en raison de leur reproduction parthénogénétique et leur cycle de développement de courte durée. Beaucoup d'espèces de ce groupe du genre *brachionus et keratella* sont utilisées dans les fermes aquacoles pour l'alimentation des alevins (Ouéda et al. 2010).

Les analyses des principales composantes du bassin lamellé pilote et expérimental montre un écosystème viable où l'on retrouve une stratification décroissante de la biomasse algale, bactérienne, fongique et zooplanctonique. Les paramètres physico-chimiques tels que le pH ; la température et l'oxygène dissous conditionnant les réactions physico-chimiques et la survie des micro-organismes sont fortement corrélés à la biomasse. Les corrélations établies entre certains groupes de la flore bactérienne et les matières en suspension semblent correspondre au phénomène d'adsorption des bactéries par la matière en suspension décrit dans la littérature. La distribution de l'activité symbiotique (Algale-bactérienne) et parasitaire (phytoplancton-zooplancton) a montré que les lamelles ont eu un effet sur la qualité de l'eau et l'écologie du bassin.

Production et composition de biogaz dans les deux réacteurs anaérobies à haut rendement

La production moyenne journalière du volume de biogaz enregistrée par le premier réacteur anaérobie à haut rendement (R1) était de 107± 17 litres soit un volume surfacique de 9,7± 1,5 L/m^2 par jour, où 2,5 L/g de matières volatiles solides (MVS) éliminées. Quant au second réacteur anaérobie à haut rendement (R2) connecté en série à (R1), la production moyenne journalière du volume de biogaz de ce réacteur était de 105±14 litres soit un volume surfacique de 9,5± 1,4 L/m^2 par jour, où 1,8 L/g de matières volatiles solides (MVS) éliminées. La quantité de biogaz enregistrée dans R1 est supérieure à celle de R2, cela pourrait s'expliquer par le fait que R1 était placé en tête du traitement avec une forte charge organique comparé à R2. Bien que les taux de production de biogaz fussent plus élevés en R1 qu'en R2, les analyses statistiques ont montré qu'il n'y avait pas de différence significative entre ces deux réacteurs ($p > 0,05$). En outre, ces taux de production de biogaz restent faibles comparativement à ceux rapportés par Konaté et al. (2013) obtenu à partir d'un anaérobie dans des conditions climatiques similaires. Cette différence pourrait s'expliquer par les facteurs environnementaux, les conditions opérationnelles : telles que la charge organique, le temps de séjour etc...(El-Fadel et Massoud, 2001; Stadmark et Leonardson, 2005).

La composition moyenne du biogaz enregistrée durant toute la période de suivi des deux réacteurs anaérobies est donnée comme suit :

- 54% ± 10 méthane, 6% ± 1 du dioxyde de carbone, 8 % ± 2 N_2 et 32 % d'autres gaz (H_2, H_2S, H_2O, ...) pour R1 ;
- 44% ± 5 méthane, 12% ± 2 dioxyde de carbone 9 % ± 1 N_2 et 34 % d'autres gaz (H_2, H_2S, H_2O, ...) pour R2

La teneur en méthane était plus élevée dans R1, éventuellement en raison de la charge organique et température interne plus élevées que celles de R2. La teneur en H_2S du biogaz était très faible voir négligeable (1 ppm et de 0 ppm dans R1 et R2 respectivement). En effet, Konaté et al. (2013) a attribué ce fait à la rareté des sulfates dans les eaux usées domestique au Burkina Faso. D'autre part, la teneur des autres gaz n'était pas négligeable, puisque environ 32 % du biogaz était attribué à d'autres gaz, tels que H_2, H_2O. Cela pourrait s'expliquer par les processus de dénitrification et d'autres facteurs environnementaux et conditions opérationnelles. Cependant, la teneur en méthane reste dans la gamme des valeurs de référence des eaux usées d'origine domestique (Hodgson et Paspaliaris, 1996; Kotsyurbenko et al. 2004).

Taux d'accumulation de boues de deux réacteurs anaérobies à haut rendement et du bassin lamelle

Les taux d'accumulation de boues très faibles ont été enregistrés dans les deux réacteurs anaérobies à haut rendement et dans le bassin lamellé : 0,0006 ; 0,0002 et 0,0014 m^3 de boue per habitant et par an respectivement. Ces taux d'accumulation de boues restent très faibles comparés aux gammes de valeurs rapportées dans la littérature (Gomes de Souza, 1987; Mara et Pearson, 1998; Keffala et al. 2011; Picot et al. 2005) et même celles de conditions climatiques similaires (Nelson et al. 2004; Konaté et al. 2010, 2013).Cette faible production de boues pourrait être due à la forte biodégradabilité des eaux usées domestiques, combinée avec les conditions climatiques favorables (températures mésophiles constants dans le Sahel). Cette configuration de deux réacteurs anaérobies à haut rendement connectés en série, puis suivis par un bassin lamellé avec des bouchons en plastique fixés aux chicanes offre une excellente option de traitement des eaux usées domestiques qui minimise la production de boues, d'où pourrait réduire son coût de fonctionnement et d'entretien.

Conclusion

Cette étude a permis de concevoir, de mettre en œuvre, d'optimiser et de suivre les performances épuratoires de deux filières de traitement des eaux usées domestiques sous le climat sahélien de Ouagadougou au Burkina Faso. Les résultats présentent des rendements épuratoires très satisfaisants pour l'élimination des matières organiques, des matières en suspension, des pathogènes et des nutriments. En outre, cette étude a montré que l'introduction de trois lamelles verticales (avec les bouchons en plastique fixés aux lamelles) contrecourant dans un bassin (BL) recevant les effluents de deux réacteurs anaérobies pourrait avoir un rôle important dans non seulement l'augmentation du temps de séjour hydraulique, la diversité écologique, l'accroissement de la biomasse épuratrice (267%), mais aussi dans l'abattement des bactéries (*E. coli*).

En plus, les deux réacteurs anaérobies à haut rendement (R1 et R2) en tête du traitement de ces deux filières ont démontré une capacité importante de production de biogaz tant en quantité qu'en qualité sous climat Sahélien. Plus important encore, l'accumulation de boues très faible a été enregistrée respectivement dans R1, R2, et BL : 0,0006 ; 0,0002 et 0,0014 m^3 de boues par habitant par an, réduisant ainsi le coût de son extraction et de sa gestion.

Il ressort de cette étude que les concentrations résiduelles en matières organiques et pathogènes de l'effluent traité répondent aux normes recommandées par l'Organisation Mondiale de la Santé pour une réutilisation des effluents en agriculture non restrictive.

Au vue, de cette bonne performance épuratoire, ces deux options de traitement pourraient être considérées comme, des technologies alternatives de traitement des eaux usées à faible coût pour les populations à faible revenu dans les zones urbaines et péri-urbaines en Afrique Subsaharienne. Cela pourrait contribuer d'avantage aux efforts de réduction de la pauvreté et de la famille qui sévissent cette partie de l'Afrique.

Enfin, pour une vulgarisation à grande échelle de ces technologies, d'autres investigations supplémentaires pourraient s'intéresser à l'utilisation de ces effluents en aquaculture et en agriculture.

List of abbreviations

AR-BP	:	two-stage high-rate Anaerobic Reactors followed by a Baffled Pond
AR-SF	:	two-stage high-rate Anaerobic Reactors coupled with wet-dry Sand Filters
BP	:	Baffled Pond with attached-growth
BOD_5	:	5-day Biochemical Oxygen Demand
COD	:	Chemical Oxygen Demand
CP	:	Control Pond
DO	:	Dissolved Oxygen
IPCC	:	Intergovernmental Panel on Climate Change
HLR	:	Hydraulic loading rate
MPN	:	Most Probable Number
SF		wet-dry Sand Filters
ST	:	Septic Tank
TSS	:	Total Suspended Solids
UASB	:	Upflow Anaerobic Sludge Blanket
WSP	:	Waste Stabilization Ponds
WTTs	:	Wastewater Treatment Technologies

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Chapter 1

1. Introduction

1.1 Importance of wastewater treatment and reuse for sustainable sanitation in poor urban neighbourhoods

According to the World Commission on Sustainable Development, sustainability has been defined as ''development that meets the needs of the present generation without compromising the ability of future generations to meet their needs''. Therefore sustainable urban sanitation can be achieved when solutions are put in place to prevent waste generation threats to public health in the city and threats to the environment (including surface water, groundwater, air and soil) in a way that can be repeated time and time again indefinitely.

Wastewater management is one of the major issues faced by most developing countries and is particularly problematic among the low-income inhabitants of sub-Saharan African cities. Huge amounts of wastewater are generated and discharged haphazardly in the environment without any treatment and in most cases are used as water sources for peri-urban irrigation (Qadir *et al.,* 2010; Khan *et al*., 2013; Maiga *et al*., 2014; Saldías *et al*., 2015). Among others, Kayombo *et al*., (2005) found that the persistence of many water borne diseases (cholera, typhoid, dysentery, infectious hepatitis, parasitosis, and poliomyelitis) in these regions is due to the inadequate wastewater treatment systems. Therefore, sustainable wastewater management is required to improve public health and to produce microbiologically safe effluent for crop irrigation and fish farming.

Over recent decades, wastewater collection and treatment technologies in developed countries have been greatly improved and have become very important assets in mitigating the impact of domestic and industrial effluents on the environment. Developing countries, on the other hand, remains far behind. In West Africa for instance, only 31% of the population uses improved sanitation facilities and nearly one in four (1/4) uses no form of sanitation by practicing open defecation (MDGs report, 2008). Furthermore, according to Kulabako (2005), 90% of urban wastewater collection and treatment systems in developing countries are based on septic tanks and pit latrines. However, these systems present health risks in urban areas, if the water table is high and flooding is frequent, since the effluent from these systems is only partially treated before being discharged into the immediate environment. Furthermore, the wastewater collection and treatment facilities are often non-existent and the few that do exist are inadequate to thoroughly clean all the harmful substances and organisms, which eventually find their way to surface water and groundwater (Drechsel and Evans, 2010), and thereby to people's homes once again, where they transmit all sorts of water-borne diseases.

This situation is getting worse due to the rapid urbanization and industrialization of African cities, which are expected to double their population by 2030 (UN-Habitat, 2008). Also, in recent decades, because of the recurrent episodes of drought experienced by arid and semi-arid countries, the immigration of rural populations to cities has accelerated. Large numbers of Sahelian cultural and religious habits involve using water for sanitation, spilling of greywater into the streets (lack of sewerage), and open defecation in the back yards of slum areas. For instance, a survey in 2007 showed that 57% of the households in Burkina Faso were practicing open defecation (Koné, 2011). In most cases within this country, population density in cities is greater than 160 persons per hectare and most of its cities are impoverished and lack planned and adequate services. The consequences are enormous, ranging from the high cost of drinking water treatment, land degradation, and algal blooms causing eutrophication of rivers to threats to biodiversity, human health and environmental health in general. For instance, according to the World Bank (2008), about 3 million peoples die prematurely every year in the developing countries, due to water-borne diseases, and most of the victims are children under five years of age and women without adequate water supply and sanitation.

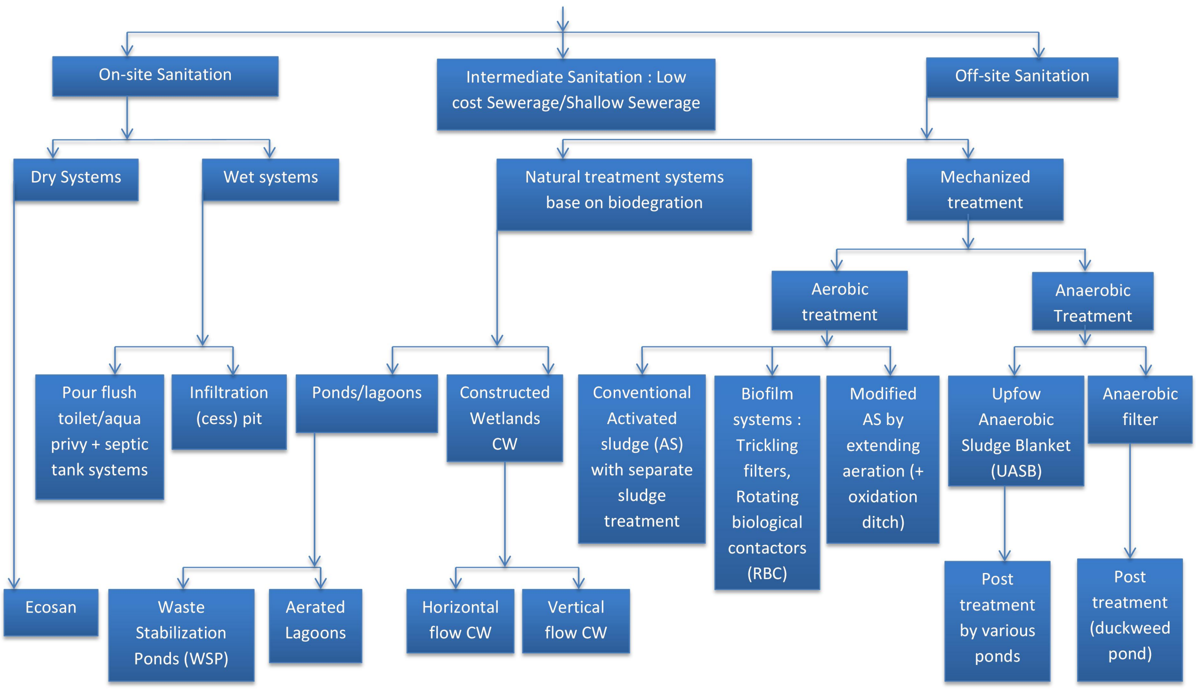
Moreover, Cisneros (2011) asserted that many low-income countries are located in arid or semi-arid regions. The climates of these areas are often characterized by long dry seasons and temperatures between 25 and 34°C. These regions also have short rainy seasons with great constancy (≥ 2500h/year) and energy (19.5-22.7 $MJ/m^2/day$) in the form of solar radiation.

Apart from the sanitation problems of these arid or semi-arid regions, they also face great water scarcity. Due to the scarcity of water resources, the reuse of highly concentrated untreated wastewater in peri-urban irrigation has become more common in recent years (Qadir *et al.,* 2010), thus making it key to first destroy the faecal pathogens that are present in the wastewater. Therefore, there is an increasing interest in these regions in the reuse of treated wastewater for urban irrigation, the reuse of treated excreta for fertilizer (Sangaré *et al.*, 2014), and the recovery of energy via the production of biogas (Mendoza et al., 2009). Treating municipal wastewater to reach World Health Organization (WHO) reuse standards at a low-cost remains a great challenge and there is an urgent need to optimize wastewater treatment technologies in low-income countries.

Hence, investing in sanitation and hygiene is not only about saving human lives and dignity; it is the foundation for investing in human development, especially in poor urban and peri-urban areas. However, one of the main bottlenecks encountered by the municipalities of sub-Saharan African countries is the limited information and awareness about more appropriate and sustainable technologies for managing sanitation problems in such a way that project costs are affordable while still protecting public health and water resources. From a sustainability perspective, Jeppsson and Hellström (2002) have highlighted that the most sustainable systems for wastewater treatment are those systems that are aimed at reusing the water for irrigation, the nutrients for fertilization and the organic matter for energy generation. Innovative and appropriate design parameters which account for local conditions in wastewater collection, treatment and disposal or reuse systems, can contribute to sustainable urban wastewater management particularly, in West African cities.

1.2 Wastewater treatment options

The number of available wastewater treatment technologies, and their combinations, is nearly unlimited. Therefore, each pollution problem calls for a specific, optimal solution involving a series of operations and processes, as organized in flow diagrams (Tilley et al., 2008). The basic choices in technology involve the following dichotomies: (i) dry or wet, (ii) centralized or decentralized (with sewers or not), (iii) mechanized or natural, (iv) biological or chemical-physical, and (v) aerobic or anaerobic. **Figure 1.1** presents an overview of these major options. Recent studies have shown that sustainable performance of a technology can only by achieved if the selection of the treatment technologies considers the environmental, economic and social factors associated with each geographical context (Mena-Ulecia and Hernández, 2015). Therefore, alternative sustainable concepts for developing countries should aim at removing the disadvantages of traditional concepts without losing the benefits.



(adapted from Van der Steen, 2008 lecture notes)

Figure 1.1: Overview of major options for wastewater treatment

1.3 Anaerobic treatment technologies

Anaerobic digestion refers to fermentation processes in which organic material is degraded by various anaerobic bacteria and biogas (composed of mainly methane and carbon dioxide) is produced. Anaerobic wastewater treatment option has been given more attention over the aerobic wastewater treatment since the era of energy crisis in the 1970s associated with the increased demand for industrial wastewater treatment (Henze *et al*., 2008). The development of the Upflow Anaerobic Sludge Blanket reactor (UASB) by Lettinga and co-workers (Lettinga *et al*., 1980) represents a breakthrough for anaerobic treatment because of its potential for net energy production.

Anaerobic treatment technologies can be divided into low-rate systems (such as septic tanks, anaerobic ponds or lined pits) or high-rate systems (UASB, Anaerobic filter, anaerobic contact process). High-rate systems have the advantage of supporting higher hydraulic loading rates and thus smaller tanks volumes, shorter retention times, reduced area requirements, and can be applied for both small and large scales, to treat domestic and industrial wastewater. However, the high rate systems are in general more complicated to construct, operate and maintain than low-rate systems.

Anaerobic treatment has been reported to be very effective in removing biodegradable organic compounds, leaving mineralized substances such as NH_4^+, PO_4^{3-} and S^{2-} in solution, but often is considered a pretreatment (Metcalf and Eddy, 2003; von Sperling and Charnicharo, 2005; Henze *et al*., 2008). There are reports indicating that not only the organic removal in anaerobic systems is deficient for full treatment, but the pathogen removal is also not sufficient (Lettinga *et al*., 1993; Metcalf and Eddy, 2003). Moreover, it is clear from numerous studies that the quality of anaerobic effluent rarely meets the discharge standards of most countries, despite several modifications (Lettinga *et al*., 1993; Khan *et al*., 2011a). For instance, UASB reactors have undergone several improvements: introduction of settlers at the top of gas-liquid-solid-separator, addition of anaerobic filters, two UASB reactors placed in series and even the incorporation of an external sludge digester (van Haandel and Lettinga, 1994; Lettinga, 2008; Lew *et al*., 2003; El Hamouri, 2004; von Sperling and Chernicharo, 2005; Khan *et al*., 2011a, b and c). Consequently, some additional or post-treatment is necessary in order to achieve the desired effluent quality and thus avoid the contamination of the receiving water bodies.

1.4 Post-treatment of anaerobic effluent

The inability of anaerobic processes alone to meet disposal standards of most countries has driven the development of subsequent post-treatment. A variety of post-treatment configurations based on various combinations with UASB or anaerobic reactors have been studied in some countries across the world (**Table 1.1)**. It is clear from numerous studies that combined biological anaerobic and aerobic treatment configurations are well-known for their low-cost, operational simplicity, efficiency, and reliable removal of nutrients (N and P) and pathogens (viruses, bacteria, protozoans and helminths) (Peña, 2002; von Sperling *et al.,* 2002; von Sperling and Chernicharo, 2005; Khan *et al.,* 2014).

A common process used at several wastewater treatment plants (some in laboratory or pilot scale) in warm countries, such as Brazil, Colombia, India, Egypt, Morocco and Uganda is apply a UASB followed by final polishing units (FPU) or polishing ponds (PP) (**Table 1.1**). It is obvious from this overview that the West African countries, with such favorable climatic conditions, are lagging behind in this respect. Apart from being efficient in removing pollutants, this combination anaerobic and aerobic processes offers technical, economical and operational advantages (von Sperling et al., 2002; Peña, 2002; El-Hamouri, 2004; von Sperling and Chernicharo, 2005; von Sperling et al., 2005; El-Shafai et al., 2007; Babu, 2011; Khan et al., 2014). Despite these great efforts, the final effluent is still generally devoid of dissolved oxygen (DO) and rich in nutrients. Moreover, polishing ponds operate at long hydraulic retention times (10 to 50 days), which would require more land (Mara, 1996; Khan, 2012). The need for extensive parcels of land is a big challenge and is critical in application of this technology, even in developing countries where land may not be so expensive.

Table 1.1: overview of various anaerobic effluent post treatment performances

Hybrid treatment systems	Effluent concentration (and % removal efficiency)							Reference & Country
	BOD	COD	TSS	NH_4^+-N	TN	TP	FC	
	mg/l	mg/l	mg/l	mg/l	mg/l	mg/l	MP /100mL	
CEPT+UASB+Zeolite	32 (85)	45 (91)	24 (88)	0.3 (99)	0.5 (99)	0.5 (94)	1E+5 (99)	(Aiyuk et al. 2004) Belgium
UASB+ Dissolved Air Flotation	-	17 (98)	4 (98.4)	-	-	0.6 (98)	-	(Penetra et al. 1999) Brazil
UASB+Coagulation flocculation	>20 (91)	>50 (87)	>30 (82)	-	-	-	4.3E+3 (99.9)	(Jaya Prakash et al. 2007) India
UASB+Slow Sand Filtration	12 (92.6)	27 (91)	20 (91)	-	-	-	1E+3 (99.995)	Tyagi et al. 2009) India
UASB+Polishing ponds	24 (92)	108 (79)	18 (96)	20 (50)	25 (55)	-	5.8E+2 (99.999)	(Khan et al. 2014) India
UASB+constructed wetlands	-	52 (82)	174 (65)	14 (70)	17.5 (70)	0.74 (89)	1E+3 (99.99)	Sousa et al. 2001) Brazil
UASB+Duckweed ponds	14 (96)	49 (93)	32 (91)	0.41 (98)	4.4 (85)	1.1 (78)	4E+3 (99.998)	(El-Shafai et al. 2007) Egypt
UASB+Down-flow hanging sponge	9 (96)	46 (91)	17 (93)	18 (28)	28 (40)	-	3.4E+4 (99.95)	(Machdar et al. 2000) Japan
UASB+ Sequencing Batch Reactor	5.8 (97)	26 (94)	5 (98)	0 (100)	12.6 (77)	1.2 (65)	7.5E+2	Moawad et al., 2009) Egypt (Khan et al., 2011a) India
UASB+ Rotating Biological Contactors	-	43	-	2.2 (92)	-	-	9.8E+2 (99.9)	Tawfik et al. 2005) Egypt
UASB+Trickling filter	17-57 (80-94)	60-120 (74-88)	<30 (73-89)	-	-	-	-	Chernicharo & Nascimento, 2001) Brazil
UASB+Overland flow system	48-62 (53-83)	98-119 (77-83)	17-57	14-18	-	-	2.4E+5 (99-99.9)	(Chernicharo et al. 2001) Brazil
UASB+ Activated Sludge Process	-	50 (85 -93)	13-18 (82)	-	-	-	-	(von Sperling et al. 2001) Brazil
UASB+Flash aeration system	22 (89)	57 (86)	47 (83)	-	-	-	5E+3 (99)	(Khan et al 2011b) India
UASB+ Baffled Pond	60	200	90	-	-	-	3.87E+6 (99.77)	(von Sperling et al. 2002) Brazil
Septic Tank+ Land infiltration	20 (91)	100	30 (86)		(75)	-	(99.999)	Mena-Ulecia & Hernández 2015) Chile
RACHAHR + MP	25 (93.6)	170 (78.8)	115 (65.2)	-	-	2.4 (70.7)	2.4E+3 (99.993)	(El Hamouri, 2004) Morocco
WSPs (AP+FP+MP)	220 (87)	(81)	110 (66)	29.1 (38.5)	-	12.5 (17.2)	5.4E+3 (99.966)	(Maiga et al. 2006) Burkina Faso
Anaerobic Tank + FP+ Baffled Ponds	-	-	28	19.1 (74.5)	-	-	-	(Babu, 2011) Uganda

UASB= Upflow anaerobic sludge blanket (reactor); AP= Anaerobic Pond; FP= Facultative pond; MP= Maturation; Pond CEPT= Chemically Enhanced Primary Treatment; WSPs= Waste Stabilization Ponds, RACHAHR= ''Réacteur Anaérobie et Chenal à Haut Rendement'' i.e. Anaerobic Reactor with a High rate algal pond

(Adapted from Khan, 2012)

1.5 Low-cost treatment options in Sub-Saharan Africa

The systems that have been developed in the industrialized world could not solve the existing sanitation problems, especially in low-income urban areas. Some, because of their high capital investments, maintenance costs and skilled manpower requirement have been a major barrier for their implementation by many countries in Africa (Veenstra and Alaerts, 1996; Agunwamba, 2001b; Bolton et al., 2010; Olukanni and Ducoste, 2011).

Consequently, Constructed Wetlands (CW), conventional Wastewater Stabilization Ponds (WSP) and Septic Tanks (ST) are the most common low-cost wastewater treatment technologies used in developing nations, especially in tropical regions (Mara, 2004; Babu, 2011; Mekonnen *et al*., 2015). The main reason could be due to cost effectiveness in construction and maintenance of these technologies.

Constructed wetlands (CW) are described as engineered systems that have been designed and constructed to mimic natural wetland systems. CW use mainly aquatic plants that have root systems which provide attachment sites for bacterial growth and activity. According to Yalcuk *et al.*, (2010), CW can serve as primary, secondary or tertiary water treatment systems. The major advantages of CW systems are their low operation costs, low energy requirements, resilience in the face of loading shocks, and effectiveness in reducing organic matter, odour and total suspended solids. In addition, CW have the potential for resource recovery, in the form of the harvested biomass, which could be applied as fodder for animals. Therefore, CW are an attractive alternative for African countries. Recently, Mekonnen *et al*., (2015) have reviewed the application and the performance of CW, at laboratory to full scales in African countries, including Tanzania, Egypt, Kenya, Nigeria, South Africa, Tunisia, Morocco, Uganda, Cameroon, Ethiopia, Benin, Burkina Faso and Cote d'Ivoire.

On the other hand, the major bottlenecks of CW include the need of a relatively large area for construction (2-10 m^2/inhabitant), their incomplete removal of pathogen, limited nitrification, and the possible breeding of mosquito if there is free surface water. Moreover, the efficiency of CW may be reduced over time due to clogging if too many suspended solids remain in the influent wastewater (Babu, 2011).

Another inexpensive alternative wastewater treatment option for African countries consists in Waste Stabilization Ponds (WSP). Mara (2004) has described, these as large, shallow basins enclosed by earthen embankments in which raw wastewater is treated by entirely natural symbiotic processes involving both algae and bacteria. The ponds can be used individually or in series. Mainly, three types of ponds are used: anaerobic, facultative, and maturation. Each of these ponds has different treatment and design characteristics. Apart from being a low-cost treatment technology, WSP are also reported to be a modern wastewater reclamation and resource recovery technology in tune with modern environmentally conscious societies (Pearson, 1996, Babu, 2011). WSP are effective in removing organic matter (Mara and Pearson, 1998; Metcalf & Eddy, 2003; Henze *et al*., 2008) and are highly efficient in pathogen removal (Curtis *et al*., 1992; Davies-Colley *et al*., 1999; Van der Steen *et al*., 1999; Zimmo *et al*., 2002; Maïga *et al*., 2009; Bolton *et al*., 2010). In addition, they are simple to operate and maintain, low in energy requirements and robust in structure (Mara, 2004; von Sperling and Chernicharo, 2005). Furthermore, another advantage of WSP is that the effluent may be used for crop irrigation. The algae in the effluent are very useful, since they act as a slow-release fertilizer and over time increase the organic content of the soil and thus its water-holding capacity (Mara, 2004).

However, WSP are perceived to have odour problems and require large areas of land (2-5 m^2/inhabitant) (Pattarkine *et al*., 2006). Despite this, there are currently many of these functioning in big modern cities like Melbourne, Australia; Amman, Jordan; and Nairobi, Kenya (Mara, 2004; Khan, 2012). Upgrading the anaerobic ponds to Upflow Anaerobic Sludge Blanket (UASB) reactors may be an appropriate alternative as suggested by many researchers (Peña, 2002; El Hamouri, 2004; Mara, 2004). On the other hand, UASB reactors have suffered from the need for skilled manpower, complex infrastructures, and larger budgets for operation and maintenance.

Last but not least, Septic Tanks (ST) are also known as a low-cost wastewater collection and treatment option in low to medium density urban areas in Africa. ST are described as small rectangular chambers (with 2 to 3 compartments), built below ground level, in which household or communal (up to 300 person' equivalent) wastewater is kept for days or years (Mara, 1996). Apart from being low-cost, septic tanks require small extensions of land, no electricity, minimal operation and maintenance, only locally available materials, and no special adaptations to control flies and odor if used correctly. However, these systems require frequent emptying (faecal sludge management) and cannot efficiently remove organic matter, suspended solids or pathogens in accordance with the effluent quality disposal standards for most countries. Therefore, a post-treatment after the ST is always necessary. One of the cheapest options could be a septic tank followed by sand filtration. But, on the other hand, it has also been reported by Tyagi *et al*. (2009) that rapid sand filtration is known to frequently clog.

In summary, the adaptability of these treatment options, their adaptability to the Sahelian context has yet to be demonstrated. Therefore, the selection of the best combination of anaerobic-aerobic systems to treat urban domestic sewage here is a challenging task involves finding a proper, reliable and efficient system that is easy to operate and maintain, technically feasible, locally applicable, and economically viable. A comprehensive investigation to understand the performance of low–cost wastewater treatment technologies in the Sahelian context is key to improving sanitation here.

1.6 Scope, aim and objectives of this research work

Aside from their health, environmental and economic benefits, solutions to wastewater collection and treatment should preferably be simple and ''low-tech'', to increase their affordability in poor urban communities. This commonly means that the technology should be less mechanized and have a lower degree of automatic process control, and that its construction, operation and maintenance should involve locally available personnel and materials, rather than imported mechanized components. It is also suggested that communal sanitation is indeed the only viable alternative for slums (Schouten and Mathenge, 2010), although this could stand further analysis.

The aim of this research was to develop pilot-scale domestic wastewater treatment systems that: (i) are acceptable and applicable to the local conditions of low-income neighbourhoods within sub-Saharan African cities, (ii) minimize area requirements, (iii) are simple and inexpensive in their operation and maintenance; (iv) require a low capital investment, and (v) include more compact treatment systems that combine efficient technologies (anaerobic-aerobic) with low energy consumption. Based on these criteria, and in the light of the above challenges, two options of domestic wastewater treatment technologies were designed and implemented at the International Institute for Water and Environmental Engineering (2iE) campus in Ouagadougou, Burkina Faso, in West Africa. These were monitored under different operational conditions and optimized for the local climate.

- The first option includes two-stage, high-rate Anaerobic Reactors with biogas recovery, followed by a Baffled Pond (AR-BP). This was inspired by conventional waste stabilization ponds (WSP), but the anaerobic pond was modified to form two anaerobic reactors, the facultative pond was not applied, and special plastic baffles with rough surfaces were introduced into the maturation pond. The baffles were made with sheets of plastic onto which caps of waste plastic bottles were affixed to increase the surface area (up to 60%) for the growth of biofilm. This configuration was expected to improve the hydraulic and biofilm patterns of the pond and, as a result, increase the removal efficiency for organics and pathogens.
- The second option consists of the same two-stage, high-rate Anaerobic Reactors, but in this case followed by Vertical-flow Wet and Dry Sand Filters (AR-SF). This process is similar to the conventional Septic Tank, but differs from it due to the collection of biogas and the higher quality of its effluent. As a result, this technology could be a better variation on the standard septic tank.

Figure 1.2 presents the schematic view of the two treatment options, used in this study. Taking into account geographical and climate factors, these hybrid treatment systems which have not been tested before in West Africa can be used as alternatives for low-cost domestic wastewater treatment.

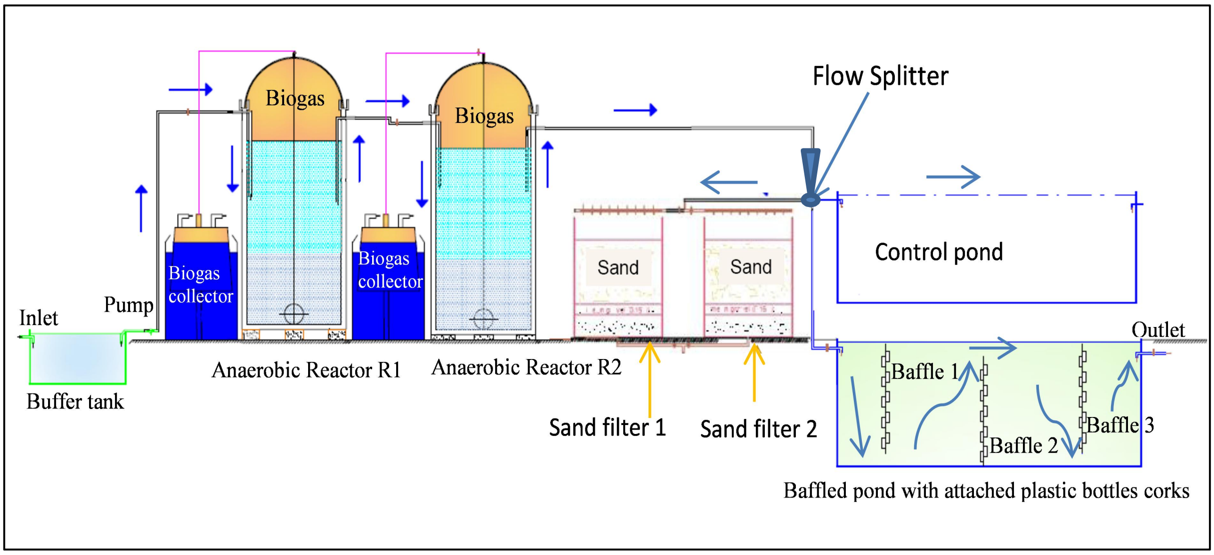


Figure 1. 2: Schematic view of the pilot plant

The overall aim of this research work was to optimize the performance of two-stage high-rate anaerobic reactors, followed by baffled pond with attached growth or wet-dry sand filters treating domestic wastewater for the urban poor, at pilot scale (approximately 50 population equivalent) in the warm, dry climate of Ouagadougou, Burkina Faso.

The specific objectives to be achieved in this research include:

- designing, implementing and testing of the performance of this technology at a pilot scale, in terms of the removal of organics, nutrients, and pathogens;
- investigating the hydraulic performance of the baffled pond with attached-growth, compared to that of the unbaffled pond;
- understanding the efficiency of removal of *Escherichia coli* in the baffled pond with attached-growth;
- evaluating the diversity and biomass of algae and zooplankton in the biofilm that develops on the plastic bottles caps affixed to the baffles and in the water column of the polishing pond;
- estimating the potential for biogas production, its composition, and the rate of sludge accumulation in the two-stage high-rate Anaerobic Reactors.

1.7 Structure of the thesis

This thesis comprises seven chapters, including this introductory section, conclusions, and future perspectives section. The following paragraphs provide a brief overview of each part.

The first section of this thesis that was presented is a general introduction to the importance of wastewater treatment and reuse for sustainable environmental management in low-income, urban neighbourhoods in Africa. It presented an overview of the major wastewater treatment options, with a particular emphasis on low-cost options, their limitations, and different combinations of anaerobic and aerobic processes. Additionally, it gives the objectives and justifications for this research, plus a brief description of the project.

The second chapter details the various aspects considered in the design and implementation of the pilot project. This chapter also presents the results of the performance evaluation of the two options, under two distinct theoretical hydraulic retention times in the two high-rate Anaerobic Reactors. The results are critically analysed and compared with the literature.

The third chapter presents a tracer test that was carried out on the baffled and unbaffled ponds, using common kitchen salt (sodium chloride). In addition, it focuses on the hydraulic characteristics and models that could be applied in predicting the performance of the ponds in terms of the removal of organic matter and pathogens.

The fourth chapter highlights the reasons why *Escherichia coli* were not found in the effluent of the baffled pond. Furthermore, the die-off rates of *Escherichia coli* in both baffled and unbaffled ponds were derived based on the hydraulic model obtained from the tracer test (Chapter 3). The importance of the baffles is also described in this.

The fifth chapter mainly focuses on the characteristics of the biofilm that formed in the baffled pond, in terms of algal biomass distribution, microbial diversity, and zooplankton species composition and distribution in the bulk water and attached media. Moreover, a statistical tool, Principal Component Analysis, was used to analyse the correlations among phytoplankton, zooplankton, bacteria, and suspended solids.

The sixth chapter presents and discusses the potential for biogas production, its quality, and the rate of sludge accumulation in the two anaerobic reactors. Finally, the thesis ends with some general conclusions, as well as some perspectives for the future of decentralized sanitation in Africa (chapter 7).

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Chapter 2

2. Design, implementation and performance evaluation of AR-BP and AR-SF

2.1 Design and implementation of the pilot plant

The design of the two options AR-BP and AR-SF was based on two concepts of low-cost wastewater treatment technologies as described in the introduction section. The two concepts are: (i) anaerobic systems based on the anaerobic ponds of WSPs, septic tanks, and Upflow Anaerobic Sludge Blanket (UASB) reactors; and (ii) aerobic systems including the facultative and maturation ponds of WSPs and the vertical-flow sand filtration.

2.1.1 Anaerobic systems

Anaerobic systems are mainly designed to remove organic matter and suspended solids via the sedimentation of the settleable fraction and its subsequent anaerobic digestion in the resulting sludge layer (Peña, 2002, Mara, 2004, Camargo-Valero, 2008). Furthermore, Camargo-Valero (2008) found that about 30 percent of influent soluble organic matter is transformed by anaerobic processes to biogas (CH_4, CO_2). A brief review of the design procedures for Anaerobic Ponds (AP), UASB reactors, and septic tanks is presented in the following paragraphs.

- **Anaerobic Pond**

The design procedure for anaerobic ponds is based on calculations of the influent volumetric organic load, as a function of temperature and wastewater strength. Hydraulically, either a plug flow or a completely mixed model is assumed (Peña et al., 2000). Typical design values and operation information for anaerobic ponds are showed in **Table 2.1** (Peña, 2002; Mara, 2004; Camargo-Valero, 2008). These literature values were used based on Burkina Faso climatic conditions (warm-dry) for a rational approach to design the anaerobic reactor. It was suggested by AWWA (1991) that before undertaking a final process design, to carry out extensive raw sewage BOD_5 sampling during at least two years, since high levels of this parameter often represent the main limiting factor in the design process within developing countries.

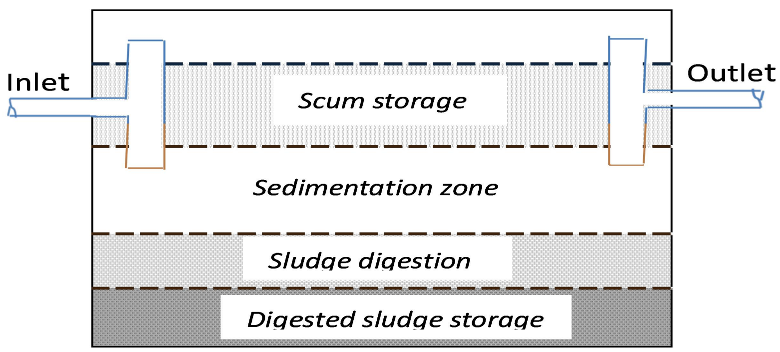
Table 2.1: Design and operation values for anaerobic ponds

Parameters	**Design value**		
Temperature (^{0}C)	Mean air temperature of the coldest month		
pH (-)	7.5		
Depth (m)	3 to 5		
Theoretical hydraulic retention time (d)	1 to 3		
Influent sulphate concentration (mg/l)	<500		
Design values of volumetric organic loading λ_v	λ_v (gBOD/m^{3}d)	BOD_5 (removal %)	Temperature (T) ^{0}C
	100	40	<10
	20T-100	2T+20	10 to 20
	10T+100	2T+20	20 *t*0 25
	350	70	>25
Anaerobic Pond volume V_{ap}(m^3)	$V_{ap}=\frac{BOD_{influent}(g/m^3)\times \text{influent flow rate } (m^3/d)}{\lambda_v}$ **(2.1)**		
Sludge accumulation rate SAR	0.04 m^3/person equivalent (PE)/year		
Interval of desludging n (years)	Once every 1 to 3 years when $n=\frac{V_{ap}}{3}\times\frac{1}{PE\times SAR}$ **(2.2)**		

Source: Peña (2002), Mara (2004), Camargo-Valero (2008), Van der Steen (2008)

- **Septic tanks**

The design procedure for Septic tanks that is presented here and described by Mara (1996) is based on Brazilian septic code. This reactor is considered to have four zones, each of which serves a different function: scum storage zone, sedimentation zone, sludge digestion zone, and digested sludge storage zone (**Figure 2.1**). **Table 2.2** outlines the basic equations mainly used to estimate the total septic tank design capacity. For better on-site effluent disposal, the septic tank should be divided into at least two compartments and usually two-thirds the total volume is assigned to the first compartment and one-third to the second (Mara 1996).



Source: Mara (1996)

Figure 2.1: The four functional zones of a Septic tank

By knowing the local conditions, such as the population served, the wastewater generated per capita per day, and the mean air temperature in the coldest month, the equations in **Table 2.2** could be systematically applied to determine a septic tank design capacity.

Table 2.2: Design equations for septic tank

Septic tank Zone	Design equation
Scum storage V_{sc} (m^3)	According to the Brazilian septic code, the volume of scum is about 30-40% of that of the sludge, thus the volume for scum storage may be calculated as follows: $V_{sc} = 0.4V_{sl}$ **(2.3)** Where: V_{sc} is the volume of scum in m^3 and V_{sl} is the volume of sludge accumulation in m^3
Sedimentation V_h (m^3)	The required time to allow settleable solids to sediment is given by Equation 2.2. It decreases with the number of people served: $t_h = 1.5 - 0.3 * \log(P * q)$ **(2.4)** Where: t_h= minimum mean hydraulic retention time for sedimentation, in days, (should not be less than 0.2 day) P=contributing population, q=wastewater flow per person, l/day. The volume for sedimentation (V_h in m^3) is given by: $V_h = 10^{-3} * P * q * t_h$ **(2.5)**
Sludge digestion V_d(m^3)	The requiredtime for anaerobic digestion to settle digested solids (t_d, days) is given by the Equation 2.4. It varies with temperature (T, in °C): $t_d = 1853T^{-1.25}$ **(2.6)** Various equations for t_d were derived in the literature by considering the process growth kinetics of a completely mixed anaerobic digester (Mara 1996). The volume of fresh sludge is assumed to be 1liter per person per day (l/cd). When it passes to the sludge storage zone and digests during t_d days, its volume reduces to 0.5 l/cd. Therefore, the volume of the sludge digestion zone (V_d, in m^3) may be calculated as: $V_d=0.5*10^{-3}*P*t_d$ **(2.7)**
Digested sludge storage V_{sl} (m^3)	The rate of accumulation of digested sludge (**r**, in m^3 per person per year) and the interval between successive desludging operations (**n**, years) are the main variables to evaluate the volume of the sludge storage zone. The following design values for (**r**) are used: For n<5:……………………r=0.06 m^3/person year And n>5:…………………..r=0.04 m^3/person year The sludge storage volume (V_{sl}, in m^3) is given by: $V_{sl}=r*P*n$ **(2.8)**
Total volume of the septic tank V (m^3)	The reactor capacity of the septic tank (V in m^3) is the sum of the volumes for scum storage, sedimentation, digestion and sludge storage(volume for the free board should be considered): $V=V_{sc}+V_h+V_d+V_{sl}$ **(2.9)** Since $V_{sc}=0.4V_{sl}$, then **2.9** becomes: $V=V_h+V_d+1.4V_{sl}$ **(2.10)**

Source: Mara (1996)

- **UASB reactor**

The design of UASB reactor combines the features of a high-rate bioreactor with those of an in-built secondary at the top. The major parameters considered when designing a UASB reactor are: organic load, hydraulic load, gas load, and solids retention time (SRT).

Prior to designing a UASB reactor, a thorough wastewater characterisation is necessary. When the allowable organic loading rate or the volumetric loading rate is known, the required UASB reactor volume can be easily calculated from the influent flow rate and its concentration (**Equation 2.11**) (Henze *et al*. 2008):

$$V_{UASB}=\frac{COD_{inf}*Q_{inf}}{\lambda_v} \quad \textbf{(2.11)}$$

Where: V_{UASB}= volume of the UASB reactor (m^3)
COD_{inf}= influent COD (kg/m^3)
Q_{inf}= influent flow rate (m^3/day)
λv= Volumetric organic loading rate (kgCOD/m^3/day

The λv depends on temperature, sludge quality, and wastewater composition (**Table 2.3**).

Table 2.3: Design guidelines for UASB with granular sludge

Operational Temperature (^{0}C)	Volumetric organic loading rate (kg COD/m^3/day)	
	Soluble COD	30% SS-COD
15	1.5-3	1.5-2
20	2-4	2-3
25	4-8	3-6
30	8-12	6-9
35	12-18	9-14
40	15-24	14-18

Source: Henze *et al*. (2008)

Once the size of the reactor is fixed, the upflow velocity can be determined from **Equation 2.12**:

$$U_{upw}=\frac{Q_{inf}}{A} \quad \textbf{(2.12)}$$

Where: U_{Upw}= average upflow velocity of the wastewater (m/h)
Q_{inf}= influent flow rate (m^3/h)
A= cross sectional area of the reactor (m^2)

Some typical values of U_{Upw} from the literature are applied depending on the wastewater characteristics: 3 m/h for soluble industrial wastewater; 1 to 1.5 m/h for partially soluble (pre-settled sewage); and 0.5 m/h for wastewater with a high content of suspended solids (Henze *et al.* 2008). In addition to the wastewater upflow velocity, the UASB reactor is also mixed by the turbulence brought about by the biogas production. The biogas upward velocity can be calculated using **Equation 2.13** below:

$$U_{biogas}=COD_{load}\frac{COD_{conv\text{-}meth}}{100}\frac{0.35}{F_{meth\text{-}biogas}}\frac{(T+273)}{273}U_{upw} \qquad \textbf{(2.13)}$$

Where: U_{biogas} = Biogas upward velocity (m/h)

COD_{load}= influent COD concentration (kg/m^3)

$COD_{conv\text{-}meth}$= % of the COD (in m^3/day) converted to methane

$F_{meth\text{-}biogas}$= methane fraction of biogas (generally between 0.6 and 0.9 of the wastewater)

T= average ambient temperature (^{0}C)

U_{Upw} = average upflow velocity of the wastewater (m/h)

Another parameter for optimal UASB design is sludge retention time (SRT; **Equation 2.14**).

$$SRT=\frac{\text{Total sludge present in the UASB reactor (kg)}}{\text{Sludge withdrawn per day from the reactor}(\frac{kg}{day})} \quad \text{(in days)} \qquad \textbf{(2.14)}$$

Generally, UASB reactors are applied where the temperature in the reactors will be above 20°C. At equilibrium condition, sludge withdrawn has to be equal to sludge produced daily (Henze *et al*. 2008).

2.1.2 Semi-aerobic and aerobic systems

- **Facultative pond**

A number of empirical and theoretical models exist for the design of facultative ponds. These include first-order plug flow reactors, first-order completely mixed reactors, first-order dispersed flow reactors **(Equations 2.19 to 2.21)**, and surface organic loading (**Equations 2.22, 2.23**) (**Table 2.4**). All of these provide reasonable designs, as long as the basis for the formula is understood and proper parameters are selected. This wide variation reflects the variety in design and the great importance of local climatic conditions (Thirumurthi, 1974).

Table 2.4: Design models for facultative ponds

Model	Characteristics
Marais and Shaw (1961) $C_e = \frac{C_i}{(1+K_t \times \theta)^n}$ **(2.15)** Where: **C_e**= effluent BOD_5 from the last pond n, (mg/l) **C_i** = influent BOD_5 (mg/l) **K_t** = reaction rate constant at temperature T, (d^{-1}) **θ** =theoretical hydraulic retention time (d) **n** = number of ponds in series	Model based on first-order kinetics in *n* ideal completely mixed reactors in series with equal retention time. This model assumes light penetration to the bottom ofpond. The value of $k_t$is temperature dependent as follows: $K_t=K_{35}(1.085)^{(35-T)}$ **(2.16)** Where: k_{35}= reaction rate constant at 35°C = 1.2 day^{-1} T = minimum operating water temperature (°C)
Reed et al. (1988) $C_e = C_i \times e^{-K_t \times \theta}$ **(2.17)** Where: **C_e**= effluent BOD_5 (mg/l) **C_i** = influent BOD_5 (mg/l) **K_t** = reaction rate constant at temperature T, (d^{-1}) **θ** =theoretical hydraulic retention time (d)	Model based on first-order kinetics in ideal plug flow reactors. The value of k_t is temperature dependent as follows: $K_t=K_{20}(1.09)^{(T-20)}$ **(2.18)** where k_{20} = reaction rate constant at 20°C day^{-1} T =minimum operating water temperature (°C) k_{20} depends on the BOD_5 surface loading rate,but if this is not known, a value of 0.1 d^{-1}may be used
Thirumurthi (1969), based on Wehner and Wilhelm's equation (1956) $C_e=C_i \times \frac{4ae^{\frac{1}{2\delta}}}{(1+a)^2 e^{\frac{a}{2\delta}} - e^{-\frac{a}{2\delta}}}$ **(2.19)** Where: $a=\sqrt{1+4K_t\theta\delta}$ **(2.20)** **C_e**= effluent BOD_5 (mg/l) **C_i** = influent BOD_5 (mg/l) **K_t** = reaction rate constant at temperature T, (d^{-1}) **θ** =theoretical hydraulic retention time (d) **δ** =dispersion number	This is a dispersed flow model based on firstorder kinetics. The value of δ is unknown at design stage and can be determined directly by tracer studies when the pond is in operating. The value of k is temperature dependent as follows: $K_t=K_{20}(1.09)^{(T-20)}$ **(2.21)** where k_{20} = reaction rate constant at 20°C, d^{-1} T = minimum operating water temperature, °C
McGarry and Pescod (1970) $\lambda_s = 60(1.099)^T$ **(2.22)** Where: **λ_s**= maximum BOD_5 loading before failure, (kg/ha d) **T**= Temperature (^{0}C)	Model based on the maximum surface BOD loading rate that can be applied to a facultative pond before it fails.
Mara (1987) $\lambda_s = 350(1.107 - 0.002T)^{T-25}$ **(2.23)** Where: **λ_s**= BOD_5 loading rate, (kg/ha d) **T**= Temperature (^{0}C)	This surface BOD loading rate model is based on McGarry – Pescod's model and incorporates a safety factor to give a global design equation for facultative ponds loading.

➢ **Maturation Pond**

Maturation ponds are mainly designed to reduce the amount of pathogenic organisms (faecal coliform bacteria and helminth eggs), but also BOD, suspended solids and nutrients (nitrogen and phosphorus). The size and number of these ponds working in series are normally determined by the required microbiological quality of the final effluent (Mara, 2004; Camargo-Valero, 2008). Marais (1974) proposed a model for *Escherichia coli* die-off based on first-order kinetics in an ideal completely mixed reactor. The faecal coliform (FC) removal is expressed as follows:

$$\frac{N_e}{N_{in}} = \frac{1}{[1+K_d\frac{HRT}{n}]^n} \qquad \textbf{(2.24)}$$

$$K_d=2.61(1.19)^{T-20} \qquad \textbf{(2.25)}$$

Where: N_{in}= number of faecal coliform in the influent (MPN/100mL)

N_e= number of faecal coliform in the effluent (MPN/100mL)

K_d=First order decay coefficient (day^{-1}).

HRT=hydraulic retention time (days)

n= number of maturation ponds in series

T= minimum operating water temperature (°C)

Marais' model has been criticised because it only includes time and temperature. According to Camargo-Valero (2008) the von Sperling's model can be recommended in preference to Marais's model, since it takes into consideration some additional parameters of pond geometry. The von Sperling (2005) model (**Equations 2.30 and 2.31**) was derived from an extensive evaluation of the coliform decay in facultative and maturation ponds, based on data from 186 different ponds in the world.

$$\frac{N_e}{N_{in}} = [\frac{4a}{(1+a)^2}]exp\left[\frac{1-a}{2\delta}\right] \qquad \textbf{(2.26)}$$

$$a=\sqrt{1+K_{d(T)}*\delta*HRT} \qquad \textbf{(2.27)}$$

$$\delta = [\frac{L}{B}]^{-1} \qquad \textbf{(2.28)}$$

$$K_{d(T)}=K_{d(20)}(1.07)^{T-20} \qquad \textbf{(2.29)}$$

$$K_{d(20)}=0.682H^{-1.286}HRT^{-0.103} \qquad \textbf{(2.30)}$$

Or simplified $K_{d(20)}=0.549H^{-1.456}$ **(2.31)**

Where: N_{in}= number of faecal coliform in the influent (MPN/100mL)

N_e= number of faecal coliform in the effluent (MPN/100mL)

δ= dispersion number

$K_{d(T)}$= first order decay coefficient at temperature T (day^{-1}).

$K_{d(20)}$= first order decay coefficient at temperature 20 °C (day^{-1}).

HRT=hydraulic retention time (days)

L, B, H= pond length, width and depth respectively (m)

T= Temperature (°C)

For modern, optimal and suitable design procedures, for both facultative and maturation ponds, most designers prefer the dispersed-flow model based on first-order kinetics (von Sperling, 2005). It is more flexible, may be set to adjust to different pond geometries, and the unknown dispersion number (δ) during the design stage can be determined based on the von Sperling (1999) Equation (**2.28**).

- **Baffled pond design**

Inclusion of baffles in pond systems has been an extensive area of interest for many researchers (Kilani and Ogunrombi, 1984; Muttamara and Puetpaiboon, 1997; Oke and Otun, 2001; von Sperling et al., 2002, 2003; Shilton and Harrison, 2003; Shilton and Mara, 2005; Banda, 2007; Babu, 2011; Olukanni and Ducoste, 2011; Sah et al., 2012; Olukanni, 2013; Cortés-Martínez et al., 2014). These studies have revealed that the addition of baffles or wind-induced mixing improves the hydraulic conditions of the ponds, enhances pond ecology and hence in most cases also improves the effluent quality. Moreover, these researchers have tested different lengths of baffles and have concluded that the use of channels or baffles to 70 percent of the length gives better results, both in the hydraulic system of the pond and in the wastewater treatment. Furthermore, some researchers have found that the introduction of artificial attached growth media in the pond water could enhance the performance of WSP in terms of the removal of organic, ammonia nitrogen and suspended solids (Shin and Polprasert, 1988).

In traditional baffle designs, a minimum of two baffles is recommended. However, according to Oke and Otun, (2001) and Shilton and Harrison, (2003) including a greater number of baffles significantly improves hydraulic efficiency of the ponds. Then, on the other hand, it is important to consider the costs of construction by conducting a cost effectiveness study. Thus, much investigation is currently being done on mathematical modeling to optimize the design of baffled ponds. For instance, Olukanni and Ducoste, (2011) used a computational fluid dynamics (CFD) model coupled with an optimization program to optimize the selection of the best WSP configuration, based on cost and treatment efficiency. These studies revealed that it was possible to minimize the baffled pond cost, meet or exceed a target effluent log reduction of faecal coliforms, and at the same time reduce the amount of construction material, while tolerating some degree of fluid mixing within a pond. Furthermore, recently Cortés-Martínez *et al.,* (2014) used the Matlab optimization Toolbox to show that short baffles could provide similar improvements as longer “traditional” baffle designs, and therefore potentially offering significant savings in construction costs. In addition, in the same study, it was found that more than four baffles gave only marginal improvements.

- **Vertical-flow sand filtration**

The American Water Works Association (AWWA) has merely described the filter as a bed of sand supported by a layer of gravel, all of which is confined within a box, with accessories to introduce and remove water. Filtration is one of the principal treatments applied in the treatment of potable water, however, now filtration is used as post-treatment in wastewater treatment plants. The first filtration process developed for the treatment of wastewater was the Slow Sand Filter, with typical filtration rates of 30 to 60 l/m^2/day, and later optimized to rapid sand filtration of 80 to 200 l/m^2/minute (Metcalf and Eddy, 2003).

According to AWWA (1991), a sand filter is simple in design, construction and operation. The key design parameters mainly include: Hydraulic loading rate (HLR), sand size, sand bed depth, water depth, and frequency of backwashing. The first step to consider in the design process is to size the bed (AWWA, 1991). The bed area and depth are basic dimensions that drive the rest of the design. The bed area is calculated by using **Equation 2.32**, based on a selected HLR. However, the depth of the sand bed is determined by the number of years of operation desired before resanding is needed and by any constraints on the filter box depth **(Equation 2.33**).

$$\mathrm{HLR} = \frac{\mathrm{Q}}{\mathrm{A}} \qquad \textbf{(2.32)}$$

$$\mathrm{n} = \frac{\mathrm{y_i} - \mathrm{y_f}}{\mathrm{R} * \mathrm{f}} \qquad \textbf{(2.33)}$$

Where: HLR= Hydraulic loading rate (m^3/m^2/h)
Q= flow rate of water (m^3/h)
A= required bed area (m^2)
n= years of operation before sand bed rebuilding is necessary (years)
y_i=initial sand bed depth (m)
y_f= final sand bed depth before rebuilding (m)
R=sand bed removal per scraping (m/scraping)
f= frequency of scraping (scrapings/years)

2.1.3 Approach to the development of AR-BP and AR-SF

The basic principles that have governed the design of two-stage high-rate anaerobic reactors followed by two polishing options (baffled pond with attached growth and wet-dry sand filters) are summarized in the following points:

- An optimal combination of low-cost, anaerobic and aerobic systems, which can achieve high-quality effluent and also offer resource recovery (water, nutrients and energy);
- The selected treatment units should be affordable for the local population and only include locally available construction materials;
- The anaerobic system was conceived based on the design concepts described in the above paragraphs **(anaerobic pond, UASB reactor, and septic tank design procedures)** and then the optimal option was adopted;
- The pilot plant was intended to be used by an urban poor community with about 50 inhabitants and each of them could contribute 40 L/capita/day (Maiga *et al.,* 2014);
- A design flow of 1 m^3/day was considered at the initial point, then it was increased gradually (to 1.5 m^3/day) to check for the optimum operating condition;
- An average temperature for the coldest month of 25 ^{0}C was adopted;
- An influent concentration of faecal coliforms of 10^6 MPN/100 mL and biochemical oxygen demand (BOD) of 250 mg/L were assumed (Maiga *et al*., 2006);

- Three vertical baffles with 70 percent pond depth length were adopted, since the literature has recommended a minimum of two and a maximum of four baffles (**Section 2.1.2**);
- A hydraulic loading rate of 0.02 m/h for a maximum sand bed area of 1 m^2 was adopted for the sand filter design.

2.2 Description and operation of the pilot plant

The application of the above design criteria resulted in the following characteristics of the pilot plant (**Table 2.5**). The anaerobic treatment unit that receives the fresh sewage includes two anaerobic reactors operated in series, and then the aerobic units finish the treatment via either a baffled pond or sand filters operated in parallel.

Table 2.5: Summary of the design characteristics of each treatment unit

Treatment unit	Wastewater flow rate (m^3/day)	Depth (m)	Surface area (m^2)	Water Volume (m^3)	Theoretical Hydraulic Retention Time (days)	Number of baffles (units)
Anaerobic Reactor (R1)	1 to 1.5	1	1.5	1.5	1.5 to 1	-
Anaerobic Reactor (R2)	1 to 1.5	1	1.5	1.5	1.5 to 1	-
Baffled Pond (BP)	0.5	1.1	3.2	3.5	7	3
Control Pond (CP)	0.5	1.1	3.2	3.5	7	0
Sand Filter (SF)*	0.5	0.8	0.5 to 1	0.5	Infiltration time 5 minutes	-

*: The two sand filters have the same characteristics but only one was in operation at one time

The pilot plant, illustrated in **Figures 1.2 & 2.2**, was designed and implemented at the International Institute for Water and Environmental Engineering (2iE) campus in Ouagadougou, Burkina Faso, in West Africa. This region is characterized by the Sudano Sahelian climate, which consists of two seasons: a dry season of 8 months from October to May, and a short rainy season of 4 months from June to September. Annual precipitation is between 600 and 900 mm. The coldest month of the year is January, with a mean temperature of 25^{o}C, and April is the warmest one with an average of 34^{o}C (min 16 ^{o}C and max 42^{o}C). The climate of the site was described in detail by Konate et al., (2010). The wastewater from the hostels and offices (excluding wastewater from laboratories) was collected through a pipe network to a buffer tank installed upstream of the pilot plant. A peristaltic pump was used to provide an intermittent flow 3 times a day (at 8:00 am, 1:00 pm and 5:00 pm) from the buffer tank to the system. The system comprised two anaerobic reactors in series followed by two parallel polishing treatment units: a baffled pond and sand filters. The first and second anaerobic reactors were designed for organic matter removal, while the baffled pond and the sand filters were designed for further pathogen removal. Gravity flow of wastewater within the system was adapted to minimise operation and maintenance costs as shown in **Figures 1.2.** and **2.2.**

1. Peristaltic pump
2. Raw wastewater sampling point
3. Anaerobic reactor 1 (R1)
4. Anaerobic reactor 1 (R2)
5. Flow splitter (crested weir)
6. Biogas collection points
7. Sand filters (SF1 & SF2)
8. Baffled pond (BP)
9. Control pond (CP)
10. Three vertical baffles with affixed caps of waste plastic bottles

Figure 2.2: Picture of the pilot plant

2.2.1 Two-stage High-rate Anaerobic Reactors

Two circular black plastic tanks of 1.4 m diameter (capacity of 2 m^3 each) were connected in series to serve as the high-rate anaerobic reactors (R1 and R2) and their theoretical hydraulic retention time was 1.5 days each. At the top of the water surface of the anaerobic reactors, 0.5 m^3 was left for the biogas collection and storage. The anaerobic reactors were simple tanks without internal structures, like covered anaerobic ponds. A mild slope (0.08 %) was provided between the two reactors, to allow gravity flow from the first reactor to the second. The flow from the second anaerobic reactor to both the baffled pond unit and one of the sand filters was regulated by a rectangular crested weir (**Figure 2.3 and Table 2.5**).

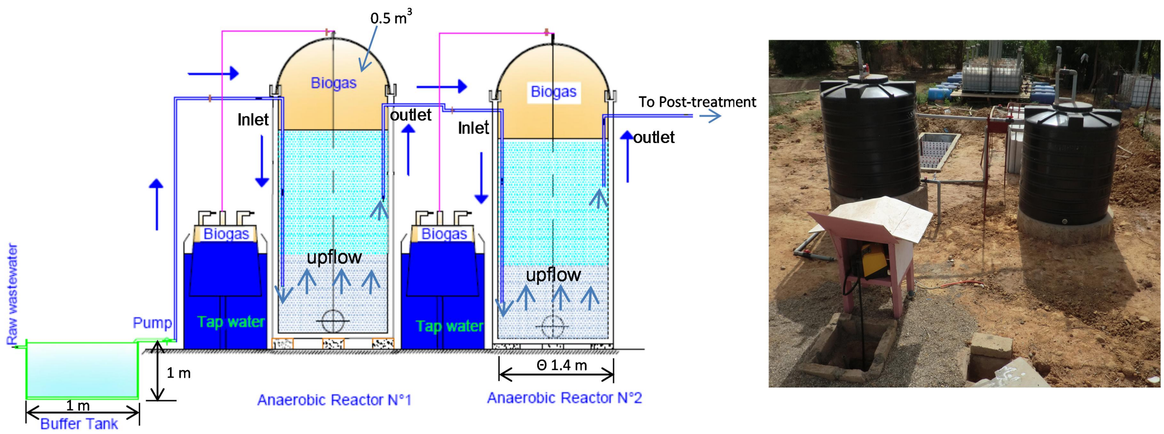


Figure 2.3: Schematic view and picture of the two anaerobic reactors connected in series

2.2.2 Baffled pond with attached growth

The baffled pond (BP) was rectangular: 3.2 m long x 1.0 m wide x 1.25 m deep (including 15 cm of freeboard, thus the wastewater was 1.1 m deep). It was expected that a depth of 1.1 m would be sufficient to maintain aerobic conditions, especially since the wastewater is forced by a baffle to come to the surface in the middle of the pond (Babu, 2011). The pond was made of hollow cement bricks and covered with a thin layer of cement mortar to reduce leakage that may contaminate the groundwater. Three vertical baffles with some specified intervals were installed, giving four compartments in the pond. The baffles were made with thin, flexible sheets of plastic on which caps from waste plastic bottles were affixed to increase the surface area (up to 60%) for the growth of biofilm. The baffles were placed perpendicular to the influent flow direction, completely across the pond, with two baffles extending downwards 70% of the pond depth while the one in the middle extended the same distance up from the bottom, creating alternate upward and downward flows. The wastewater was forced to flow under the first baffle, over the second one, and finally beneath the third baffle in order to flow out of the system. This configuration was simulated by Olukanni and Ducoste (2011) via a Computational Fluid Dynamics (CFD) model, which indicated a high faecal coliform log-unit removal, at a lower cost than the conventional horizontal flow. The theoretical mean hydraulic retention time for the baffled pond with attached growth was 7 days. The control pond (CP) has the same design and environmental conditions, except that baffles were not placed (**Figure 2.4 and Table 2.5**).

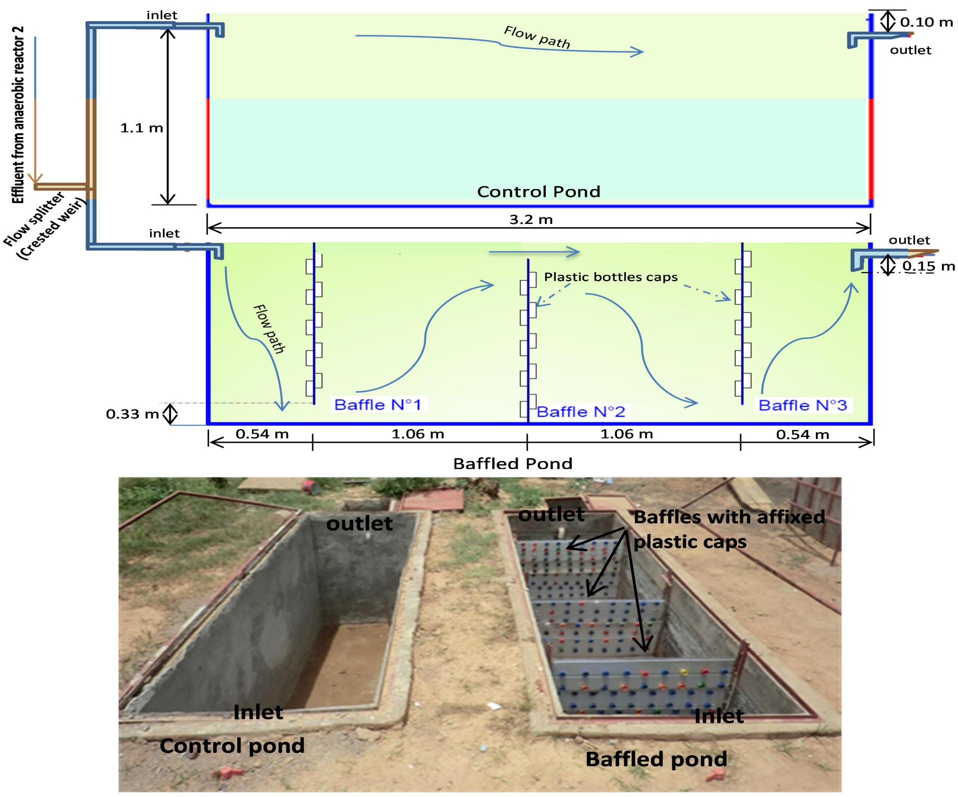


Figure 2.4: Schematic view and picture of the control pond and the baffled pond

2.2.3 Wet and sand filters

The sand filters (SF1 and SF2) were made up of three layers of local available filter media: one layer of coarse sand (0.50 m thick, with particles 0.05–2 mm in size), one layer of medium-size gravel (0.15 m thick, with particles 10–20 mm in size) and one layer of coarse gravel (0.15 m thick, with particles 20–40 mm in size), from the top to the bottom respectively. Perforated pipes (holes: 5 mm diameter, with a regular holes interval of 50 mm) were used to spread the influent over the top surface (1 m^2) of the sand filter. To avoid clogging, two infiltration units were considered. It was arranged in such a way that, when one was in operation, the other was taken out (**Figure 2.5 and Table 2.5**). By using the Stopwatch-and-Bucket Method, the average infiltration time was estimated at 5 minutes.

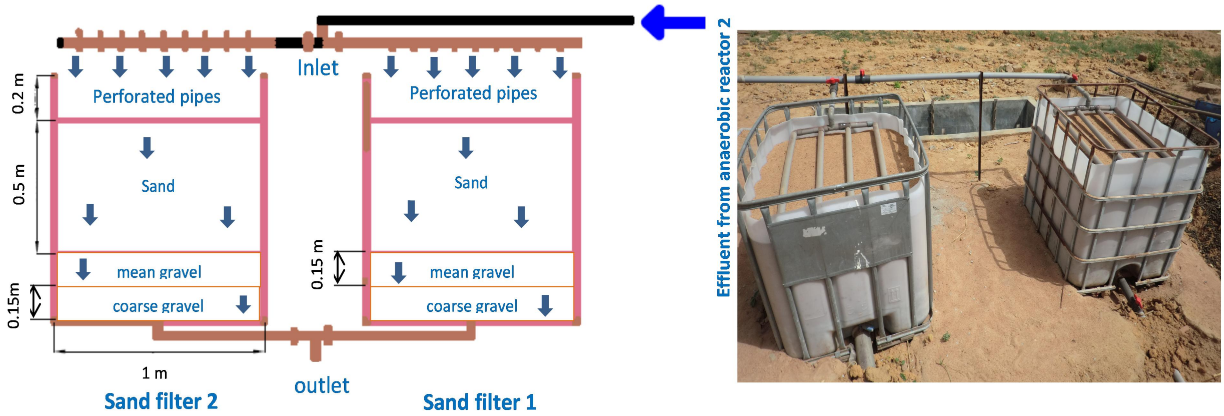


Figure 2.5: Schematic view and picture of the two sand filters

2.2.4 Pilot Plant start-up and operation

Before supplying the pilot plant with the domestic wastewater, clean water was used to wash and to test the water proofness of the system. On March 6, 2013, the first anaerobic reactor was filled with 1.5 m^3 of wastewater for acclimatization and growth of the necessary community of bacteria. Then one week later, the second anaerobic reactor was filled by pumping 1.5 m^3 of wastewater from R1. One week later, a constant flow of wastewater regulated by the peristaltic pump was supplied to the system. The pilot plant was operated under two conditions referred to as period 1 and 2. During period 1 (P1), from 8 May 2013 to 6 May 2014, an intermittent flow of 1 m^3 per day, pumped in three moments (at 8:00 am, 1:00 pm and 5:00 pm) was maintained. The theoretical hydraulic retention times of R1, R2, and BP during this period (P1) were 1.5, 1.5 and 7 days respectively, while the infiltration time of the sand filter was around 5 minutes. It should be noted that during P1, the CP had not yet been constructed. In period 2 (P2), from 13 May 2014 to 12 May 2015, the influent wastewater flow rate was increased to 1.5 m^3 a day in 3 times (at 8:00 am, 1:00 pm and 5:00 pm). Therefore, the theoretical hydraulic retention times of R1 and R2 were reduced to 1 day each, whereas that of BP was maintained at 7 days. On the other hand, the CP was put in operation, while the surface area of the unused sand filter was reduced by half (0.5 m^2). The major aim of increasing the influent flow rate, and reducing the sand bed area was to estimate optimal operating conditions of the pilot plant.

2.3 Performance evaluation of the two treatment options: AR-BP and AR-SF

The pilot plant included two options of domestic wastewater treatment. The first, two-stage high-rate anaerobic reactors followed by a Baffled Pond (AR-BP) and the second two-stage high-rate anaerobic reactors coupled with wet-dry sand filters (AR-SF). This section focuses on the evaluation of the performance of these two options in terms of the removal of organics, pathogens and nutrients under the two operating periods (P1 and P2), as described in **Section 2.2.4.**

2.3.1 Methodology

- **Sampling and analysis**

Grab samples of 500 ml were taken each week during the entire duration of the study, between 8:00 to 9:00 a.m., at the following points in the system:

- the influent wastewater (RW),
- the effluent of the first anaerobic reactor (R1),
- the effluent of the second anaerobic reactor (R2),
- the effluent of the baffled pond with attached growth (BP),
- the effluent of the wet-dry sand filter (SF), and
- the effluent of the control pond (CP).

Wastewater samples were collected in plastic bottles for organic and nutrient parameters and in glass bottles for indicator microorganisms. The collected samples were stored at 4 ºC and analysed within 3 hours for TSS, BOD_5, COD, nutrients and indicator microorganisms in the 2iE Laboratory, whereas pH, temperature, dissolved oxygen (DO) and electrical conductivity (EC) were measured immediately, *in situ*, according to the Standard Methods APHA (2012), as summarized in **Table 2.6**.

Table 2.6: Summary of methods and parameters analysed

Parameters		Reagents / apparatus
General parameters	pH, water temperature (T), dissolved oxygen (DO), electrical conductivity (EC), and redox potential	*In-situ* measurement with an integrated portable probe fitted with calibrated electrodes using the manufacturer'sinstructions (WTW 340i/SET).
	Total suspended solids (TSS)	Filtration of the samples through pre-weighed 0.45 µm GFC (Whatman® glass microfiber filters) filter.
Organic parameters	Total and filtered Chemical oxygen demand (COD)	Potassium dichromate method using a spectrophotometer at 620 nm light waves (Hack Lange, DR 5000)
	Total and filtered 5-day biochemical oxygen demand (BOD_5)	Using BOD meter-WTW OxiTop, after 5 days of incubation at 20 ^{o}C
Nutrients	Ammonia (NH_3)	Nessler solution method using a spectrophotometer at 425 nm light waves (Hack Lange, DR 5000)
	Nitrite (NO_2)	Nitriver method using a spectrophotometer at 585 nm light waves (Hack Lange, DR 5000)
	Nitrate (NO_3)	Nitraver method using a spectrophotometer at 500 nm light waves (Hack Lange, DR 5000)
	Orthophosphate (PO_4)	Phosver method using a spectrophotometer at 890 nm light waves (Hack Lange, DR 5000)
	Total phosphate (P)	Vanadomolybdate method using a spectrophotometer at 430 nm light waves (Hack Lange, DR 5000)
Indicator microorganisms	*Escherichia coli (E. coli)*and Faecal coliform	Spread plate method using Chromocult Coliform Agar (Merck KGaA 64271, Darmstadt, Germany) as culture medium.

- **Statistical analyses**

To appreciate the additional value of each treatment option, for each period, some statistical tests were conducted with STATISTICA 8.0 software (IBM). The t-test for independent (paired) samples for the treatment units in parallel at 5% significance level was used.

2.3.2 Results and Discussion

A total number of 565 samples were collected and analysed from the inlet of the pilot plant to the outlets of each treatment units, during each of the two periods.

- **Characteristics of raw and treated wastewater**

Tables 2.7 and 2.8 summarize the averages and the standard deviations of influent and effluent concentrations, together with the volumetric/surface mass loading rates for parameters that were analysed during Period 1 (P1) and Period 2 (P2). The allowable volumetric/surface mass loading rates are similar to the range of those reported in the literature (Metcalf & Eddy, 2003; von Sperling & Chernicharo, 2005; Van der Steen, 2008; Henze *et al*., 2008; Khan *et al*., 2013).

The p-values of the t-test comparing effluent concentrations and removal efficiencies in both BP and SF treatment options and for both periods are shown in **Table 2.9**.

The *E. coli* concentrations (around 10^7) in the raw wastewater (**Tables 2.7 and 2.8**) are similar to the range of those reported in the literature (Metcalf and Eddy, 2003; von Sperling and Chernicharo, 2005; Henze *et al*., 2008; Khan *et al*., 2013). *E. coli* was not detected in the effluent of any of the samples from the Baffled Pond during either period, whereas, about 1000 MPN/100ml of *E. coli* were found in Sand Filter effluents (**Tables 2.7 and 2.8**). In both Periods 1 and 2, a high variability of both *E. coli* and faecal coliforms was noticed in the raw wastewater and effluents of each treatment unit, except for the Baffled Pond, where no *E. coli* were detected (**Figure 2.6**). Possible explanations for this great variability of faecal indicators in raw wastewater could be: time of the year (hot or cold), type of sampling used (grab samples were collected at peak hours 8:00 am to 9: 00 am here), socioeconomic status of the populations contributing to the wastewater generation, low per capita water consumption, as discussed in more detail by Oliveira & von Sperling (2006) and Henze et al. (2008).

In order to understand the absence of *E. coli* in the Baffled Pond effluent, a Control Pond without baffles was constructed to provide a basis of comparison (**Chapter 4**).

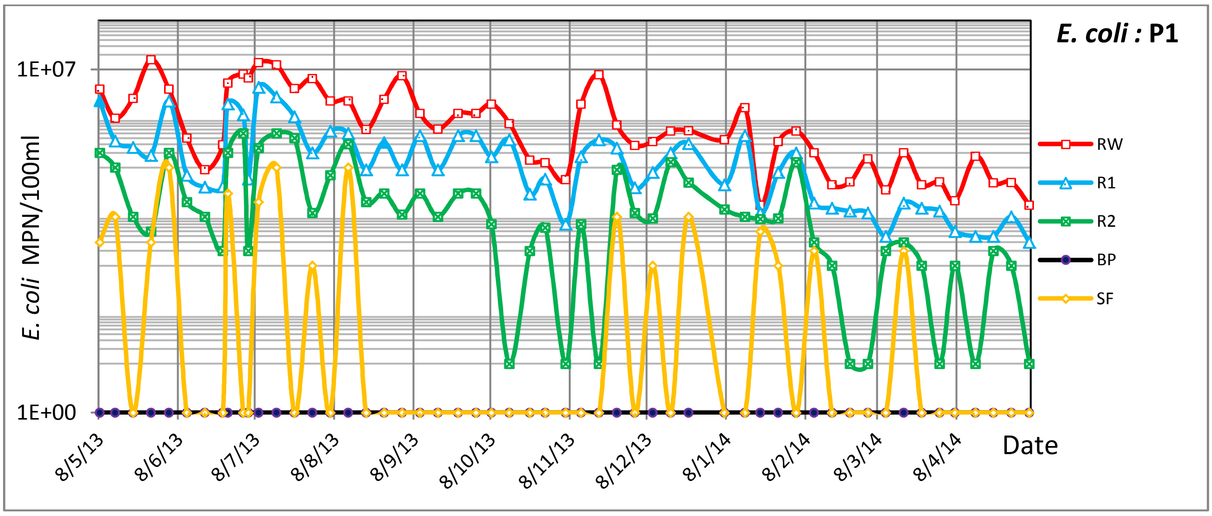


Figure 2.6: Variations of *E. coli* concentrations over time in raw and treated wastewater from each treatment units during Period 1.
Note that similar trends were observed during Period 2.

On the other hand, the mean concentrations in the raw wastewater for Periods 1 and 2 were estimated to be 424 and 425 mg/l for COD, 252 and 255 mg/l for BOD_5, and 148 and 134 mg/l for TSS, respectively (**Tables 2.7 and 2.8**). In general, a difference was noticed between the ranges usually reported in the literature (Metcalf and Eddy, 2003; von Sperling and Chernicharo, 2005; Henze *et al.,* 2008; Khan *et al*., 2013) and those effectively observed, taking into consideration COD, BOD_5, and TSS constituents in the domestic wastewater of developing countries, with a prevalence of influent concentrations lower than expected. This could be as a result of the high dilution factor of the 2iE campus wastewater, since no water-saving devices were in use.

Table 2.7: Summary of Period 1 influent and effluent concentrations, as well as volumetric/surface loading rates, at each treatment

	Raw wastewater (RW)		Anaerobic Reactor (R1)		Anaerobic Reactor (R2)		Baffled Pond (BP)		Sand filter (SF)		Number of Samples
	avg*	stdeva*	avg	stdeva*	avg	stdeva*	avg	stdeva*	avg	stdeva*	n
pH	**7.29**	0.28	**7.18**	0.21	**7.34**	0.20	**8.33**	0.58	**5.81**	0.95	60
Temperature (°C)	**30.26**	3.96	**31.68**	4.36	**31.3**	4.40	**28.10**	2.93	**29.43**	2.55	60
Electrical conductivity (µS/cm)	**761**	191	**785**	154	**773**	141	**539**	104	**685**	152	60
Dissolved oxygen (mg/L)	**0.67**	0.5	**0.48**	0.6	**0.57**	0.7	**8.74**	5.3	**7.33**	3.5	60
TSS (mg/L)	**148**	82	**50**	28	**16**	11	**20**	10	**5**	4	54
Total COD (mg/L)	**424**	201	**289**	153	**200**	110	**82**	35	**68**	41	54
Total BOD_5 (mg/L)	**252**	87	**177**	66	**120**	59	**45**	17	**26**	15	54
NH_3-N(mg/L)	**36.51**	12.11	**38.53**	13.16	**37.34**	13.52	**5.30**	2.79	**11.98**	6.66	52
NO_3-N (mg/L)	**3.78**	1.44	**2.61**	0.91	**1.70**	0.61	**1.06**	0.58	**34.51**	30.41	52
PO_4-P (mg/L)	**9.98**	5.25	**12.25**	8.44	**14.94**	9.91	**3.86**	2.13	**5.20**	3.29	52
E. Coli (n°/100 mL)	**2E+6**	4E+6	**4E+5**	8E+5	**6E+4**	1E+5	**ND**	0	**7E+3**	2E+4	60
Volumetric/surface loading rate			**R1 (g/m^3/day)**		**R2 (g/m^3/day)**		**BP (g/m^2/day)**		**SF (g/m^3/day)**		
TSS			**102.4**	54.5	**34.4**	18.5	**8.1**	4.3	**32.2**	17.3	
Total COD			**283**	134	**193**	102	**45**	24	**181**	96	
Total BOD_5			**168.1**	57.7	**118.3**	43.8	**27.7**	10.3	**110.9**	41.1	
NH_3-N			**24.3**	8.1	**25.7**	8.8	**6**	2.1	**24.1**	8.2	
NO_3-N			**2.5**	1	**1.7**	0.6	**0.4**	0.1	**1.6**	0.6	
PO_4-P			**6.7**	3.5	**8.2**	5.6	**1.9**	1.3	**7.7**	5.3	
E. coli			**2E+6**	2E+6	**3E+5**	6E+5	**7E+4**	1E+5	**3E+5**	5E+5	

*Arithmeticaverage and standard deviation +Not detected

Table 2.8: Summary of P2 influent and effluent concentrations, as well as volumetric/surface loading rates, at each treatment unit

	Raw wastewater (RW)		Anaerobic Reactor (R1)		Anaerobic Reactor (R2)		Baffled Pond (BP)		Sand filter (SF)		Number of Samples
	avg*	stdeva*	avg	stdeva*	avg	stdeva*	avg	stdeva*	avg	stdeva*	n
pH	**7.36**	0.25	**7.25**	0.20	**7.45**	0.26	**8.39**	0.50	**5.89**	0.98	53
Temperature (°C)	**29.5**	3.4	**31.0**	3.9	**30.5**	3.7	**27.9**	3	**29.1**	2.5	53
Electrical conductivity (µS/cm)	**879**	310	**900**	270	**907**	253	**617**	207	**795**	289	53
Dissolved oxygen (mg/L)	**0.63**	0.3	**0.72**	0.7	**0.56**	0.8	**12.12**	7.9	**8.05**	4.8	53
TSS (mg/L)	**134**	57	**47**	17	**14**	6	**15**	9	**5**	3	53
Total COD (mg/L)	**425**	175	**271**	117	**180**	82	**85**	33	**75**	32	53
Total BOD_5 (mg/L)	**255**	99	**165**	68	**99**	49	**40**	19	**26**	16	53
NH_3-N (mg/L)	**34.45**	13.6	**40.79**	16.95	**40.08**	17.11	**5.71**	3.53	**12.14**	6.97	53
NO_3-N (mg/L)	**4.78**	2.81	**3.12**	1.67	**2.02**	1.25	**1.02**	0.95	**49.35**	29.69	53
PO_4-P (mg/L)	**10.57**	4.17	**12.95**	4.78	**13.94**	5.22	**3.02**	2.25	**6.76**	7.29	53
E. Coli (n°/100 mL)	**2E+6**	3E+6	**4E+5**	8E+5	**7E+4**	1E+5	**ND**	0	**9E+3**	3E+4	53
Volumetric/surface loading rate			**R1 (g/m^3/day)**		**R2 (g/m^3/day)**		**BP (g/m^2/day)**		**SF (g/m^3/day)**		
TSS			**134.5**	57.3	**47.4**	16.7	**7.4**	2.6	**29.6**	10.4	
Total COD			**424.8**	175.4	**270.9**	116.9	**42.3**	18.3	**169.3**	73.1	
Total BOD_5			**254.6**	99.1	**164.7**	68.1	**25.7**	10.6	**103**	42.5	
NH_3-N			**34.5**	13.6	**40.8**	16.9	**6.4**	2.6	**25.5**	10.6	
NO_3-N			**4.8**	2.8	**3.1**	1.7	**0.5**	0.3	**2**	1	
PO_4-P			**10.6**	4.2	**13**	4.8	**2**	0.7	**8.1**	3	
E. coli			**2.3E+6**	3.2E+6	**4.5E+5**	8.4E+5	**7E+4**	1.3E+5	**2.8E+5**	5.2E+5	

*Arithmeticaverage and standard deviation   +Not detected

Table 2.9: P-values of the t-test comparing effluent concentrations and removal efficiencies in both Baffled Pond and Sand Filter treatment options and between Periods 1 and 2

Parameter	AR-BP against AR-SF				Period 1 against Period 2			
	Period 1		Period 2		Eff. Concentrations		R. efficiencies	
	Eff. Con.	R. effic.	Eff. Con.	R. effic.	AR-BP	AR-SF	AR-BP	AR-SF
TSS	0.0000	0.0000	0.0000	0.0000	0.0089	0.2855*	0.0911*	0.7241*
COD	0.0598*	0.0071	0.1307*	0.1011*	0.6286*	0.2852*	0.6954*	0.1212*
BOD_5	0.0000	0.0000	0.0000	0.0002	0.2324*	0.8474*	0.2118*	0.6821*
NH_3-N	0.0000	0.0000	0.0000	0.0000	0.5077*	0.9069*	0.2909*	0.2613*
NO_3-N	0.0000	0.0000	0.0000	0.0000	0.7829*	0.0129	0.0188	0.2408*
PO_4-P	0.0155	0.0087	0.0005	0.0000	0.0508*	0.1604*	0.0002	0.4733*
E. coli (log)	0.0219	0.0000	0.0131	0.0000	-	0.6991*	0.7578*	0.5268*

*$p \geq 0.05$: samples are not significantly different.

The weekly variability of COD, BOD_5, and TSS influent and effluent concentrations, of all treatment units during Period 1 are presented in **Figure 2.7**. Those of Period 2 were not presented here, since their distribution patterns are similar to those of Period 1. Great variability in BOD_5, COD and TSS were observed at each level of the treatment process and for both periods, which reflected a good response of the system to the high variability of raw sewage that it received. For instance, the coefficients of variation of total COD, BOD_5, and TSS in the raw wastewater during Period 1 were respectively 47%, 34%, and 55%. This has reflected a situation with high variability in the characteristics of the raw sewage produced in accordance with campus activities, marked by the mobility of staff and students. These variations are in line with the findings of Maiga *et al.* (2006) and Khan *et al*. (2013).

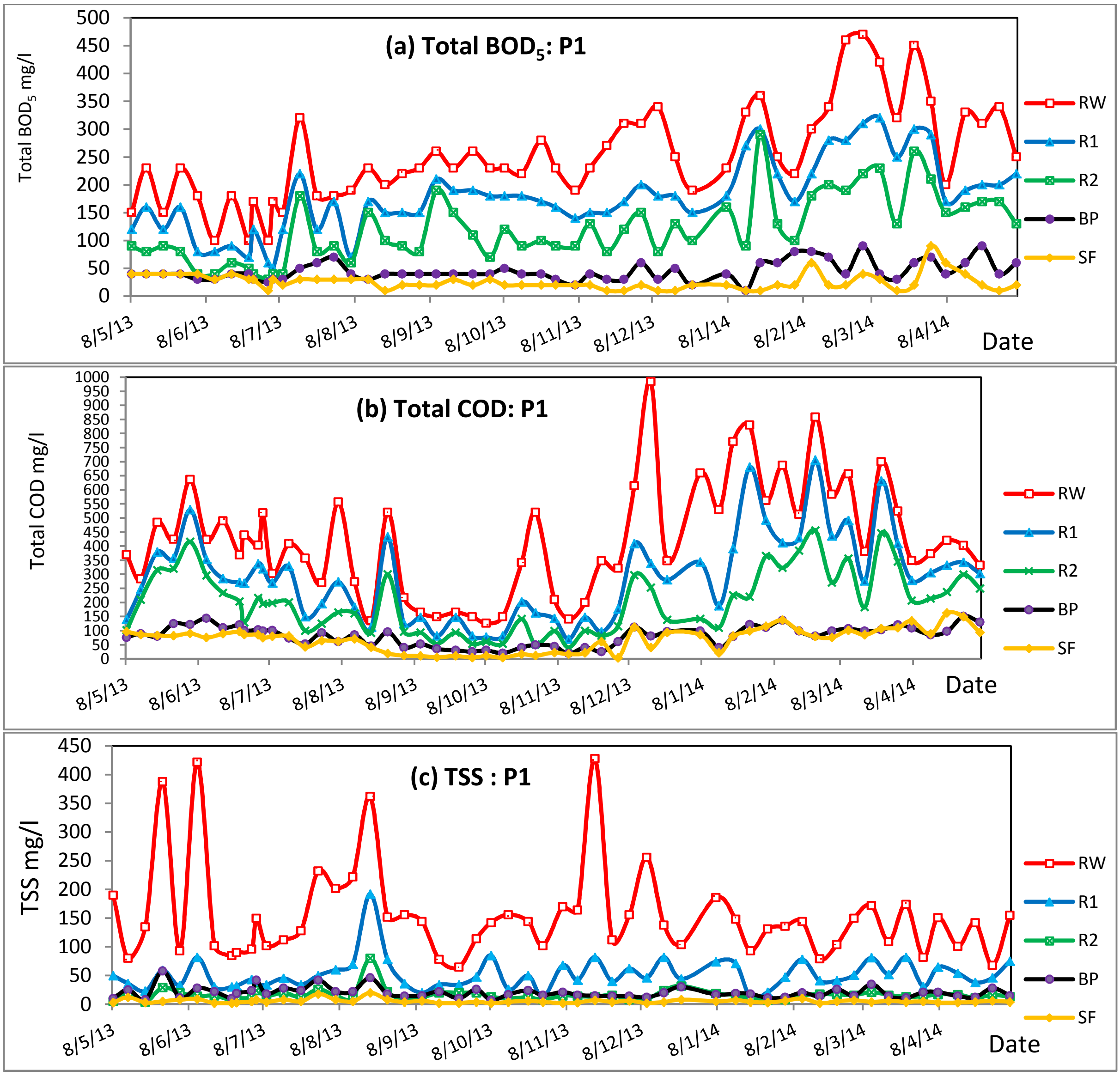


Figure 2.7: (a), (b) and (c). Variations in the concentrations of oxygen demand (BOD_5, COD) and suspended solids (TSS) over time in raw and treated wastewater from each treatment unit during Period 1.

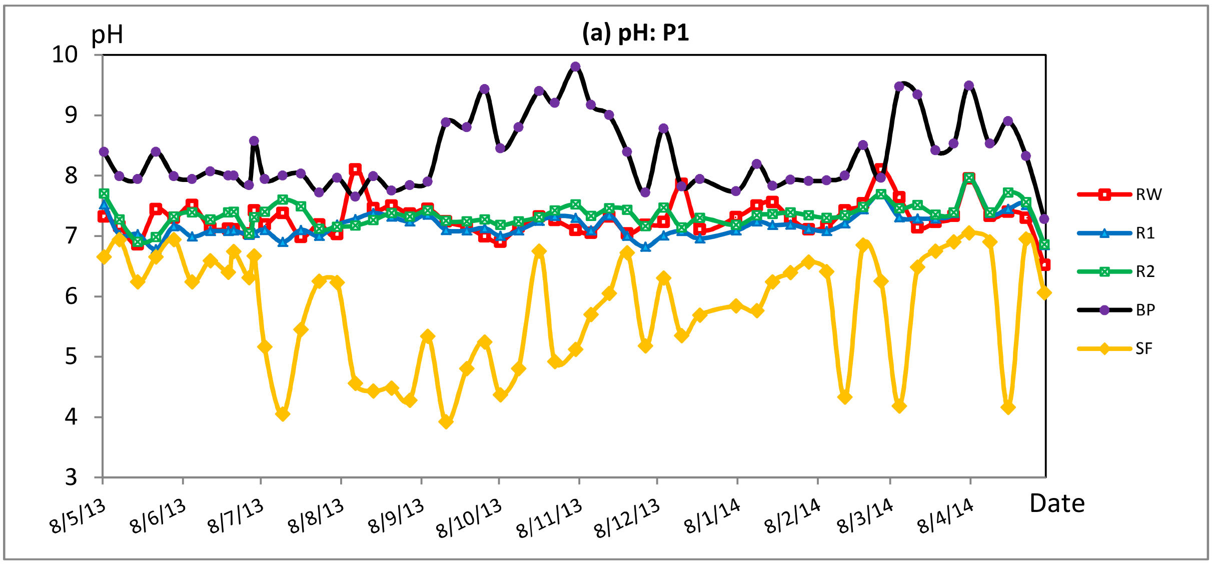
Note that similar trends were observed during Period 2

The pH values of the raw sewage and the effluents of the anaerobic reactors were in the neutral range, which indicates favourable conditions for bacterial growth and biological degradation of organic matter to produce biogas, as clearly stated in the literature (Peña, 2002; Foresti *et al.*, 2006). Besides, **Figure 2.8 (a)** depicts low variability of pH in the raw wastewater and the effluents of both of the anaerobic reactors (R1, R2) during Period 1. The same trends were also observed during Period 2 (not shown). Moreover, Mara (2004) reported that most bacteria prefer neutral or slightly alkaline conditions, around 6.5–8.5. Therefore, the anaerobic reactors may promote bacterial growth, which is beneficial to efficiently degrade the organic matter content in the wastewater. However, the pH remained high, between 8 and 9.8 for Periods 1 and 2 in the Baffled Pond (**Tables 2.7 and**

2.8), which was the result of the photosynthesis and stabilization system (Curtis *et al.* 1992; Kayombo *et al.* 2002). Contrary to the BP, low pH values from 3.9 to 6.7 for P1 and 3.7 to 7 for P2 were obtained in effluent of the Sand Filter. This may be due to the intermittent feeding and the release of H^+ that consumed the alkalinity of the medium and possibly reduces the pH during the high nitrification occurring in the system (Metcalf and Eddy, 2003).

On the other hand, the average temperature of the raw wastewater has increased from 29 to 31°C in both R1 and R2, and then decreased to 28 or 29 °C in the Sand Filter and the Baffled Pond (**Tables 2.7 and 2.8**) for both Periods 1 and 2. Therefore, the use of anaerobic reactors painted black in the sunny climate of the Sahel resulted in an increase in temperature of 2°C throughout the year. **Figure 2.8 (b)** shows similar trends and low variability for the weekly temperature was recorded in all treatment units. This situation may be explained by the sampling time which was between 8:00 a.m. to 9:00 a.m., since the air was still cool at that time.

Low values with low variability of dissolved oxygen were recorded in raw wastewater and effluents of R1 and R2 (**Figure 2.8b**), which shows the typical values expected in anaerobic conditions. However, high values were observed in both the Sand Filter and the Baffled Pond during both Periods 1 and 2. The main reasons of these high levels of DO in the baffled pond could be due to the photosynthesis of algae combined with alternate upward and downward flow induced by the baffles as previously reported by Olukanni & Ducoste (2011) and Bolton et al. (2010). In the case of the Sand Filter, it could be linked to the re-oxygenation of the sand pores between feedings (on average 5 hours), this would give enough time for the wastewater to fully drain.



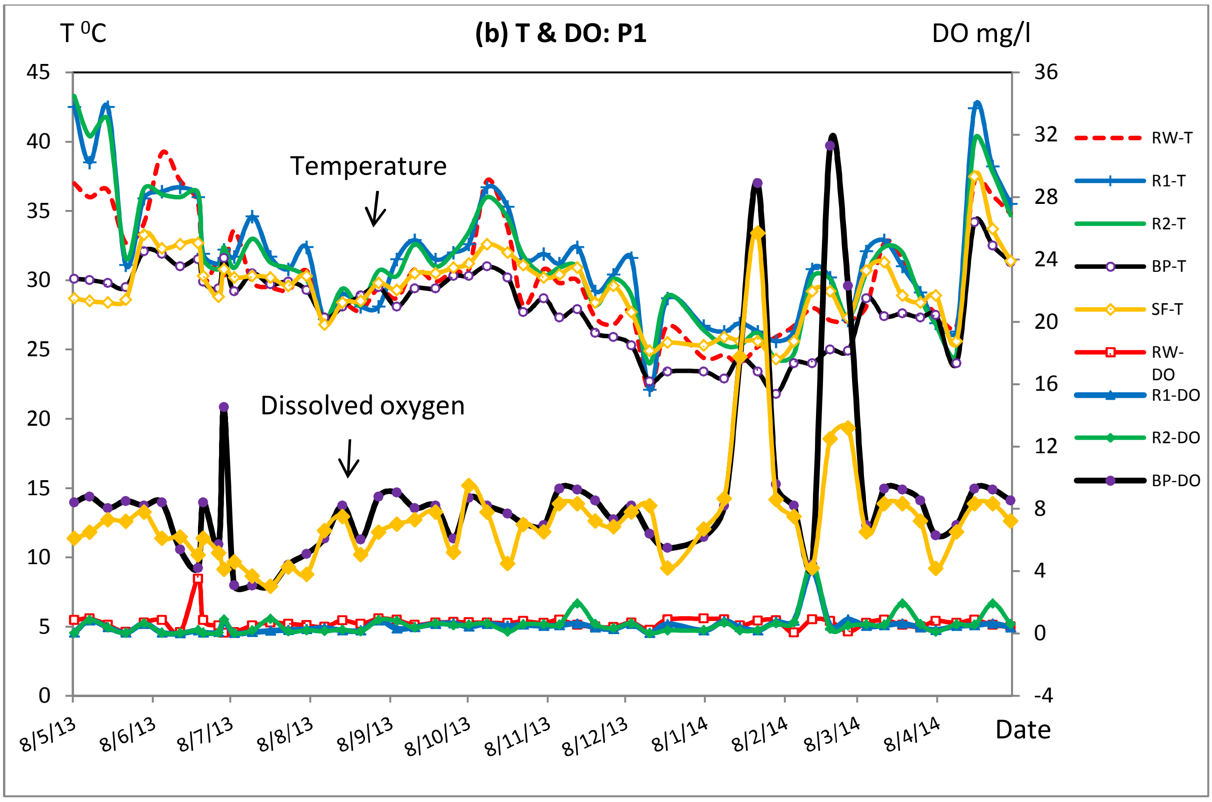


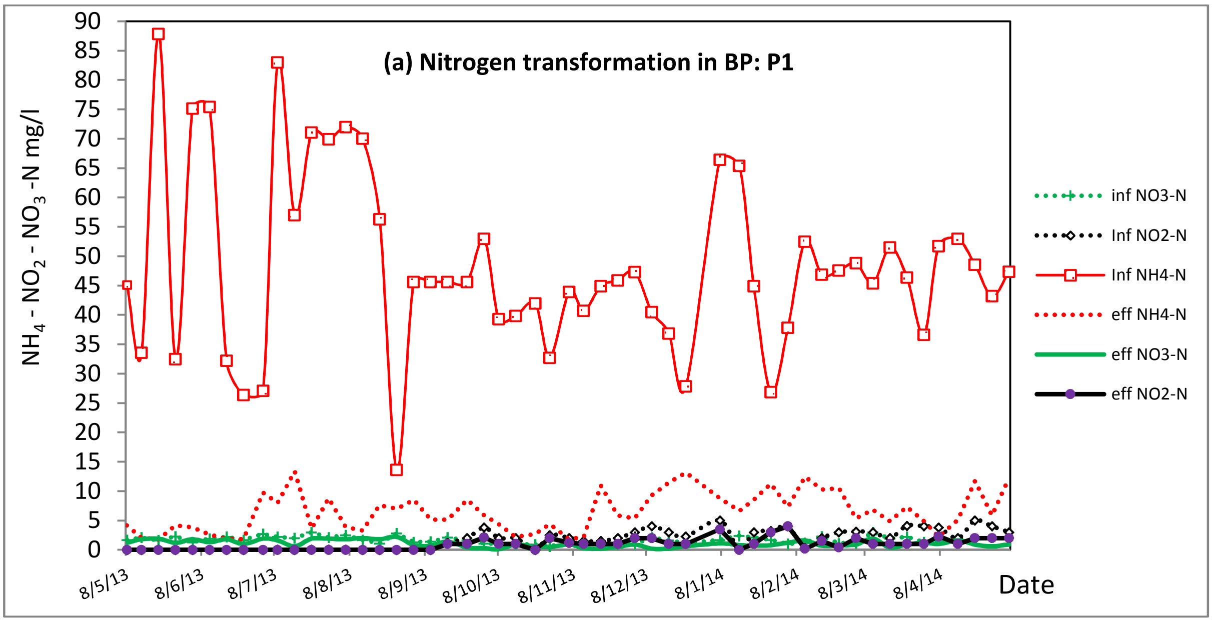
Figure 2.8: Variations in (a) temperature and (b) dissolved oxygen over time in raw and treated wastewater from each treatment unit during Period 1.
(Note that similar trends were observed during Period 2)

The concentration of NH_3-N in the influent wastewater increased from 36 to 38 mg/l in R1 and then slightly decreased to 37 in R2 for P1. In addition, during P2 the mean concentration of NH_3-N in the raw wastewater increased from 34 to 40 mg/l in R1 then remained similar in R2 (**Tables 2.7 and 2.8**). This result coincides with the results reported by Foresti *et al.*, (2006) who showed that in anaerobically treated wastewater, the NH_3-N concentration varied between 30 and 50 mg/l. Khan *et al.*, (2013) explained that the increase of NH_3-N was due to the hydrolysis of organic nitrogen in anaerobic process.

On the other hand, the concentration of NH_3-N in the Baffled Pond effluent dropped from 37 or 40 mg/l in the influent to 5 mg/l in the effluent for both Periods 1 and 2 respectively. Camargo-Valero (2008) found that depending on the characteristics of the ponds and local weather conditions, the mechanisms and pathways by which nitrogen in its various forms is removed from WSP can be attributed: to ammonia volatilisation, sedimentation of organic nitrogen via biological uptake, its retention in the ponds' bottom sludge; nitrification–denitrification and nitrate and ammonia assimilation by algae. However, it was later reported that only 2% of the overall ammonia nitrogen removed could be due to volatilization (Camargo-Valero and Mara, 2007a, 2010; Assunção and von Sperling 2012; Bastos *et al.*, 2014).

From this study, it appears that the nitrate concentrations (No_3-N) in the raw wastewater for both P1 and P2 respectively from 3.5 and 4.7 mg/l were successively reduced to 2.6 and 3 mg/l in R1, to 1.7 and 2 mg/l in R2, and to 1.06 and 1.02 in BP as shown in **Tables 2.7 and 2.8**. Possible causes may be that nitrates have been converted by other organisms present in those treatment units to dinitrogen gas or to other nitrogen forms (NH_3) through two other processes namely; assimilatory nitrate reduction and dissimilatory nitrate reduction (Metcalf and Eddy, 2003; Camargo-Valero, 2008; Babu, 2011). In dissimilatory nitrate reduction, nitrates are reduced to ammonia to yield energy rather than for biomass development. This implies that the ammonium produced would accumulate in the surrounding medium, which could be the case in the anaerobic reactors R1 and R2. However, Schumacher and Sekoulov (2002) have reported that assimilatory nitrate reduction to ammonia was carried out by attached algal biofilms on their cell surfaces with high pH (>10) and oxygen concentrations of 9 mg/l. This fact, could explain the nitrate reduction in the Baffled Pond (**Figure 2.9a**).

On the other hand, the nitrate concentrations rose significantly by more than 34 (from 1.70 to 34.51 mg/l) and 49 (from 2.02 to 49.35 mg/l) times respectively during Periods 1 and 2 in the Sand Filter, which achieved a high degree of nitrification. This fact could be due to the intermittent feeding, where an important re-oxygenation takes place within the porous media between the two supplies of wastewater in the Sand Filter. From the weekly variability curves (**Figure 2.9b**), it is clearly showed that when NH_4 and NO_2 concentrations decrease, the NO_3 concentrations increase, which is suited to the normal path of the nitrification process.



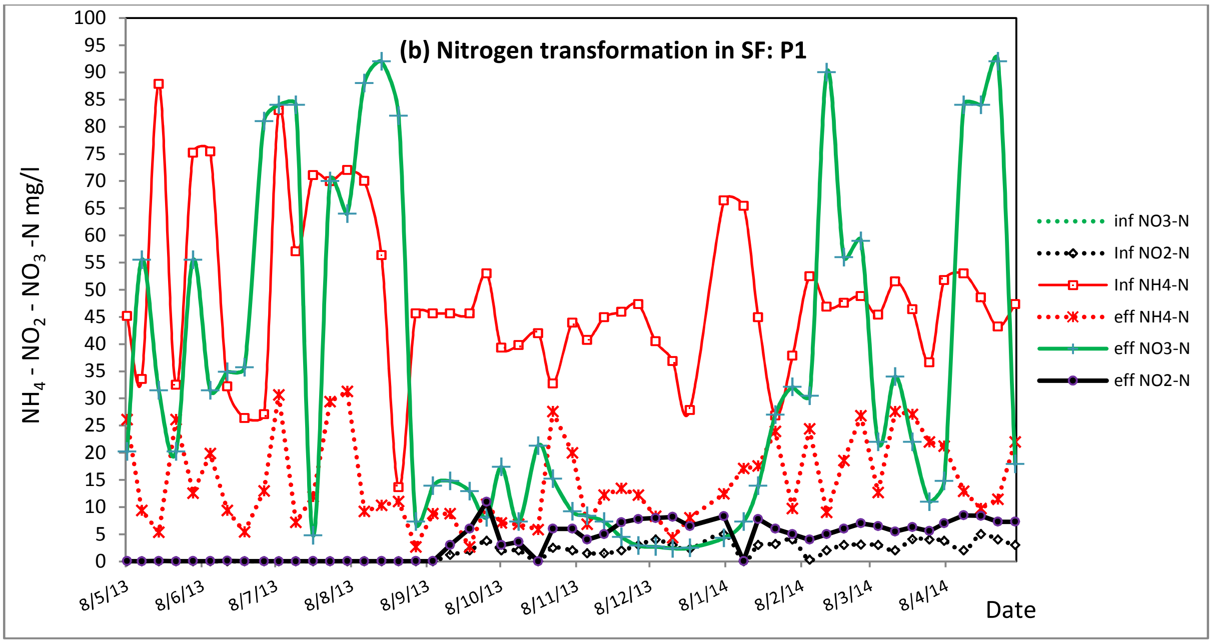


Figure 2.9: Variations of ammonium, nitrite and nitrate concentrations over time in influent and effluent wastewater from (a) the baffled pond and (b) the sand filter during Period 1. (Note that similar trends were observed during Period 2.)

In a similar way to the ammonia, the orthophosphate concentrations (PO_4-P) for both periods increased successively from 9.9 and 10.5 mg/l in raw wastewater, to 12.2 and 12.9 mg/l in R1, and to 14.9 and 13.9 mg/l in R2 (**Tables 2.7 and 2.8**) during the first stage of treatment. Nevertheless, in both aerobic treatments, BP and SF, the orthophosphate concentrations dropped to 3.8 and 5.2 mg/l respectively during P1. Henze *et al.*, (2008) and Khan *et al.*, (2013), when describing mechanisms for enhanced biological phosphorus removal, explain that under good anaerobic conditions, soluble phosphate increases due to the hydrolysis of phosphorous and phosphate. However, in the subsequent aerobic process (in the presence of oxygen or nitrate), soluble phosphate decreases under two main mechanisms: biomass assimilation and phosphate precipitation at high pH conditions as reported in the literature (Comeau *et al.*, 1987; Gerber *et al.,* 1987; von Sperling and Mascarenhas, 2005; Henze *et al.,* 2008), but it was not clear here whether such conditions were the causes of this fact in these treatment units. The weekly variability of orthophosphate influent and effluent concentrations, of all treatment units during Period 1, can be visualized in **Figure 2.10**. Those of period 2 were not presented here, since their distribution patterns are similar to period 1.

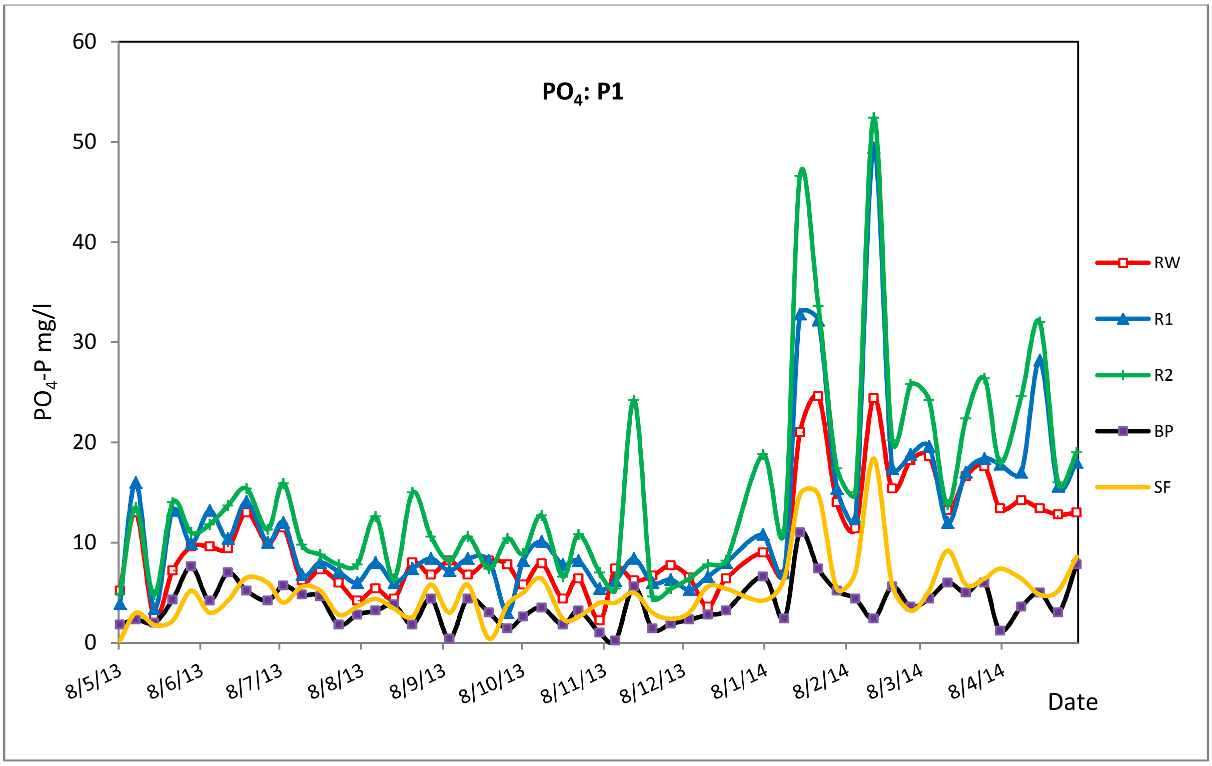


Figure 2.10: Variations of orthophosphate concentrations over time in raw and treated wastewater from each treatment unit during Period 1.
(Note that similar trends were observed during Period 2.)

In summary, the statistical analyses showed that the effluent concentration was influenced by the reduction in the theoretical hydraulic retention times between Period 1 and Period 2 (from 1.5 to 1 day) in Anaerobic Reactors 1 and 2. In fact, there were significant differences in TSS, BOD_5, NH_3, NO_3, PO_4 and *E. coli* concentrations (but not COD) in both AR-BP and AR-SF between Periods 1 and 2 (t-test, p= 0.05) (**Table 2.9**). Furthermore, the overall effluent quality in terms of bacteria, organics and nutrients of both treatment options, during Periods 1 and 2 was in compliance with the World Health Organization reuse guidelines (WHO, 2006) for restricted irrigation. Therefore, the treated wastewater could be acceptable for irrigation and fertilization (**Tables 2.7 and 2.8)**. Indeed, nitrogen and phosphorus are essential plant nutrients and, in general, have a positive effect on plant growth, unless applied in excess (Maiga *et al.* 2014).

- **Overall performance of the pilot plant**

The removal efficiencies in terms of *E. coli*, faecal coliform, TSS, BOD_5, COD, NH_3-N and PO_4-P of each treatment unit and options of the pilot plant during Periods 1 and 2 are reported in **Figure 2.12,** showing the central tendency and dispersion of these constituents. **Table 2.10** gives a comparison of the performance of the pilot plant to some technologies reported in the literature. In general, it is important to notice that better removal efficiencies were obtained within all options and treatment units for both periods.

The statistical analyses (**Table 2.9)** revealed that the removal efficiencies during Periods 1 and 2, for both treatment options (AR-BP and AR-SF) were significantly different for most constituents, with exception of COD. The differences in the results are probably due to the different inherent processes specific to each aerobic treatment. On the other hand, decreasing the theoretical hydraulic retention times of Anaerobic Reactors 1 and 2 from 1.5 days to 1 day, while maintaining those of aerobic treatments (BP and SF), did not significantly affect the performance of either the anaerobic reactors or the aerobic treatments. Indeed, there was no significant difference between TSS, COD, BOD_5, NH_3-N, NO_3-N, PO_4-P and *E. coli* removal rates when comparing the values obtained in AR-BP and AR-SF during Periods 1 and 2 (t-test, p= 0.05; **Table 2.9**).

R1 and R2 Performances

In the first stage of the treatment units (R1 and R2), more than 53% of both BOD_5 and COD, about 1 log unit of both *E. coli* and faecal coliform, and interestingly more than 88 % of TSS were removed (**Figure 2.12a-e)**. Pearson *et al.,* (1996), Peña, (2002), Mara, (2004) and von Sperling and Mascarenhas (2005) obtained similar results with anaerobic ponds and similar climatic conditions, influent wastewater, and surface loading rates. In addition, the present results are better than those of conventional waste stabilisation ponds (WSP), where the anaerobic pond, which is the initial treatment reactor, is designed to eliminate suspended solids and some of the soluble organic matter (Maiga *et al*., 2006).

However, the results revealed during Periods 1 and 2 an increase of both NH_3-N and PO_4-P in Anaerobic Reactors R1 and R2 (**Figure 2.12f,g)**, indicating the poor removal of nutrients in anaerobic processes (Khan *et al*., 2013; Bastos *et al*., 2014). On the other hand, about 30 to 32% (P1) and 33 to 35% (P2) of NO_3-N was removed in R1 and R2 respectively, which indicated that a slight denitrification process was taking place in these anaerobic reactors. The result could be related to the favourable temperature and biological activities in the reactors.

AR-BP Performance

The results obtained from this study have shown the benefits of combining anaerobic reactors and a baffled pond with attached-growth media (AR-BP) with respect to NH_3-N and pathogen removal. Removal of 84% of NH_3-N and up to 7 log units of *E. coli* (AR-BP) were achieved (**Figure 2.12a, f**). These results are similar to those found previously by Shin & Polprasert, (1988), Camargo-Valero and Mara, (2007a, 2010), Assunção and von Sperling, (2012) and Bastos *et al*., (2014).

It is interesting to observe that *E. coli* was not detected in any of the 113 samples of the effluent from the BP during the 2 periods of monitoring. A complete removal of *E. coli* from biological plants has not previously been reported in the literature. However, former studies have shown that the introduction of baffles in WSPs not only improved the hydraulics of the ponds, but also their treatment efficiencies (Muttamara and Puetpaiboon, 1997; Von

Sperling *et al.,* 2002, 2003; Shilton and Mara 2005; Olukanni and Ducoste, 2011). Certainly, this efficiency of the BP in terms of indicator microorganisms may be explained by the combination of the physical-chemical conditions and environmental factors: high temperature (>29°C), high pH (>9), a shallow pond (1.1 m), reasonable hydraulic retention time (7 days), and others relevant parameters not considered here such as ultraviolet light radiation, a significant amount of dissolved oxygen, algal biomass, nutrient limitation and competition, as described by Shin and Polprasert, (1988), Curtis *et al.,*(1992), Davies-Colley *et al.,*(1999), Maïga *et al.,* (2009) and Bolton *et al.,* (2010). Although, the effect of baffles with attached growth in a polishing pond in Sahelian conditions had not been investigated previously in detail, but, based on two years of monitoring, the introduction of these appropriate baffles seems to be one of the aspects that have greatly improved the efficiency of pathogen removal in this pilot plant.

These results of AR-BP confirm that this treatment option could be set as an alternative low-cost biological wastewater treatment for poor neighbourhoods in sub-Saharan African cities, since its performance was within reported ranges of other treatment facilities working in similar climatic conditions (Kilani and Ogunrombi, 1984; von Sperling *et al.,* 2002, 2003; Shilton and Mara, 2005; Banda, 2007; **Table 2.10**) and even better efficiencies in terms of some constituents were achieved. One reason could be attributed to good combination of the unit operations of the pilot plant and also the prevailing physical-chemical conditions and environmental factors resulting from the processes. Besides, this combination was simple to operate, little maintenance was required, no energy were needed (in fact, it provides energy in the form of biogas) and the investment cost was minimised by using locally available materials.

AR-SF Performance

This option of anaerobic reactors coupled with a wet-dry sand filter (AR-SF), has also given similar results and even better performance in terms of COD (84%), BOD_5 (89%) and TSS (96%) removal efficiencies, when compared to the AR-BP (**Figure 2.12c, d and e**). Moreover, despite the similarities in removal efficiencies, the statistical analyses (**Table 2.9**) showed that there is a significant difference in the performance of these two treatment options. Similar performance is reported by Li *et al*. (2012) where, pilot scale direct rapid sand filters were used to treat full-scale activated sludge effluent. The mean removals of TSS reported by the same authors ranged from 50 to 88%, and are inversely related to the loading rate and the average size of the grains of sand. It is important to notice that the good performance of this AR-SF was consistent with the hypothesis that a sand filter would be able to remove TSS, turbidity, and organics from wastewater (Nakhla and Farooq, 2003). The reason for this phenomenon is that the removal mechanism of TSS in the filtration process is the combination of transport and attachment (Li *et al*., 2012).

On the other hand, the average concentration of NO_3-N increased in the SF, which showed a high nitrification rate. These results coincide well with those of Panuvatvanicha *et al*., (2009), who suggest that an increase in sand depth could improve nitrification in filtration systems. One reason could be the oxygen accumulated in the biofilms surrounding sand surfaces and also in the pore spaces mainly in the intervals between feedings, therefore, resulting in fast consumption of available nitrogen by the nitrifiers.

Lesser performance in term of *E. coli*, (5 log-units) removal in AR-SF was achieved compared to the AR-BP (**Figure 2.12a**). Nonetheless, the mean faecal coliform removal rate was higher than 0.6–1.5 log units reported by Li *et al.*, (2012). The discrepancies in the results are probably due to the different process parameters and environmental conditions, such as low pH (5.6). Furthermore, the SF was operated during the study with no clogging effect, since the infiltration time remained constant on average during Periods 1 and 2 (**Figure 2.11**). This is in contradiction with the findings of Tyagi *et al.*, (2009). This good performance could be due to the high reduction of TSS occurring in the anaerobic reactors that precede the sand filter and also the available surface area that could be optimized.

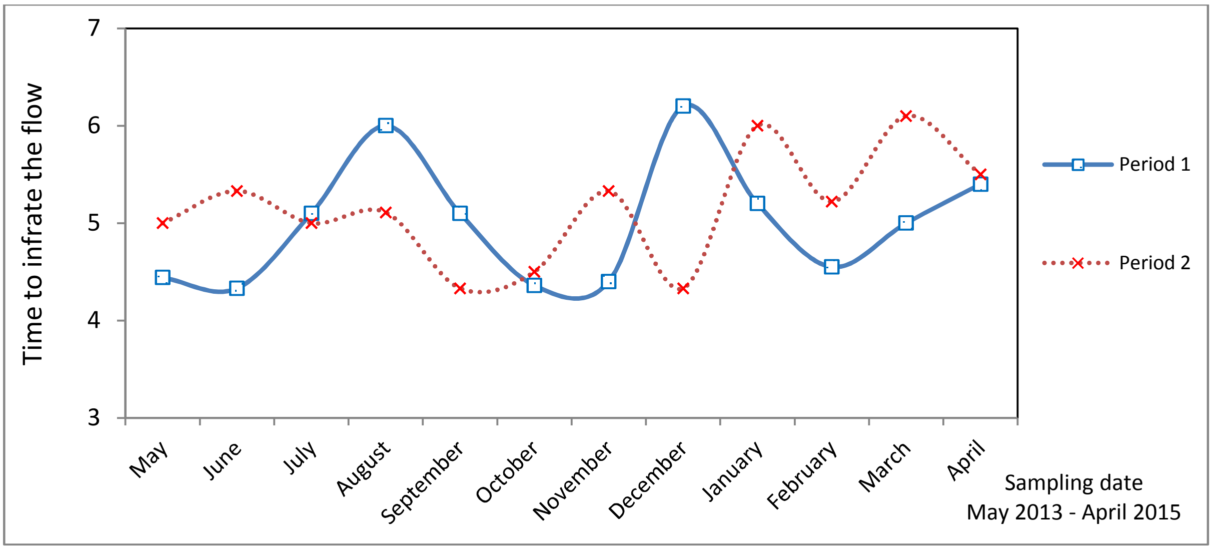


Figure 2.11: The time required for an equal amount of wastewater to pass through the Sand Filter during Periods 1 and 2

The argument stated in the AR-BP option could also be applied to the AR-SF option, in terms of performance compared to those reported in the literature (**Table 2.10**). This could be suggested as an alternative low-cost wastewater treatment technology for the urban poor of West Africa.

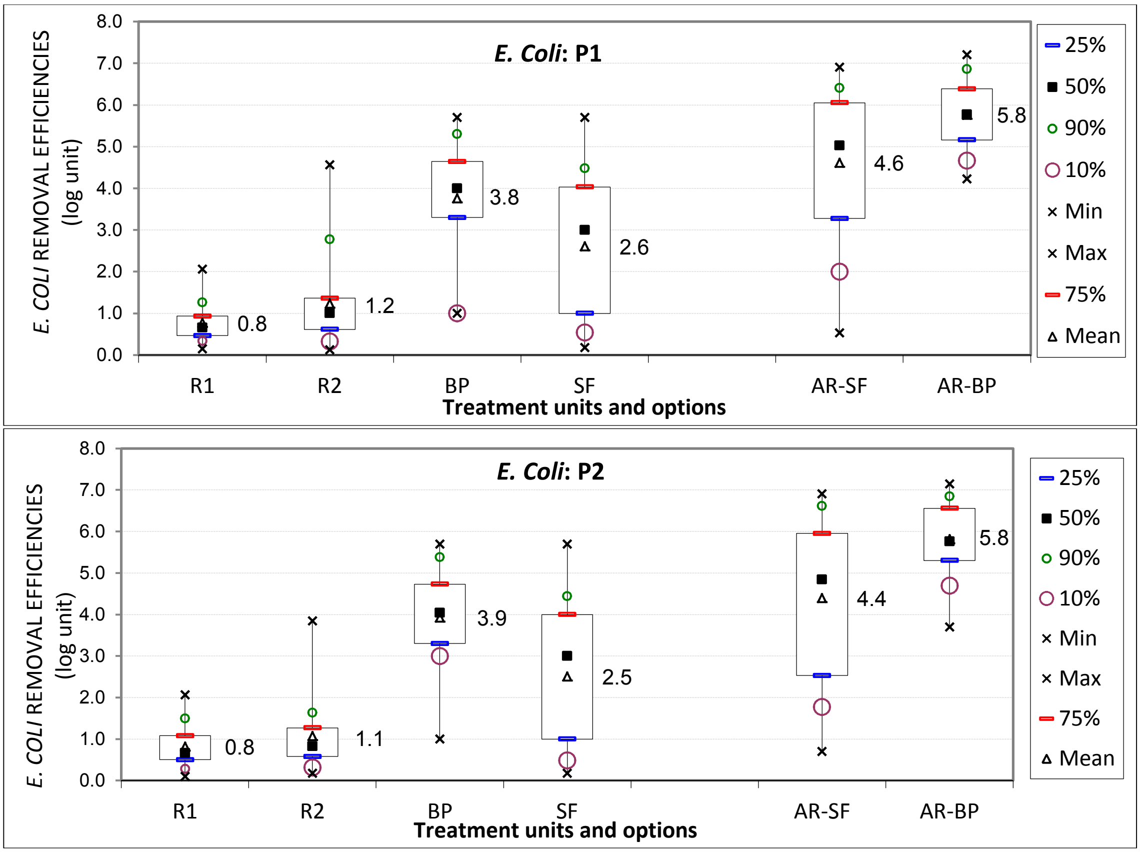


Figure 2.12a: *E. coli* removal efficiency of each treatment unit and options for both P1 & P2

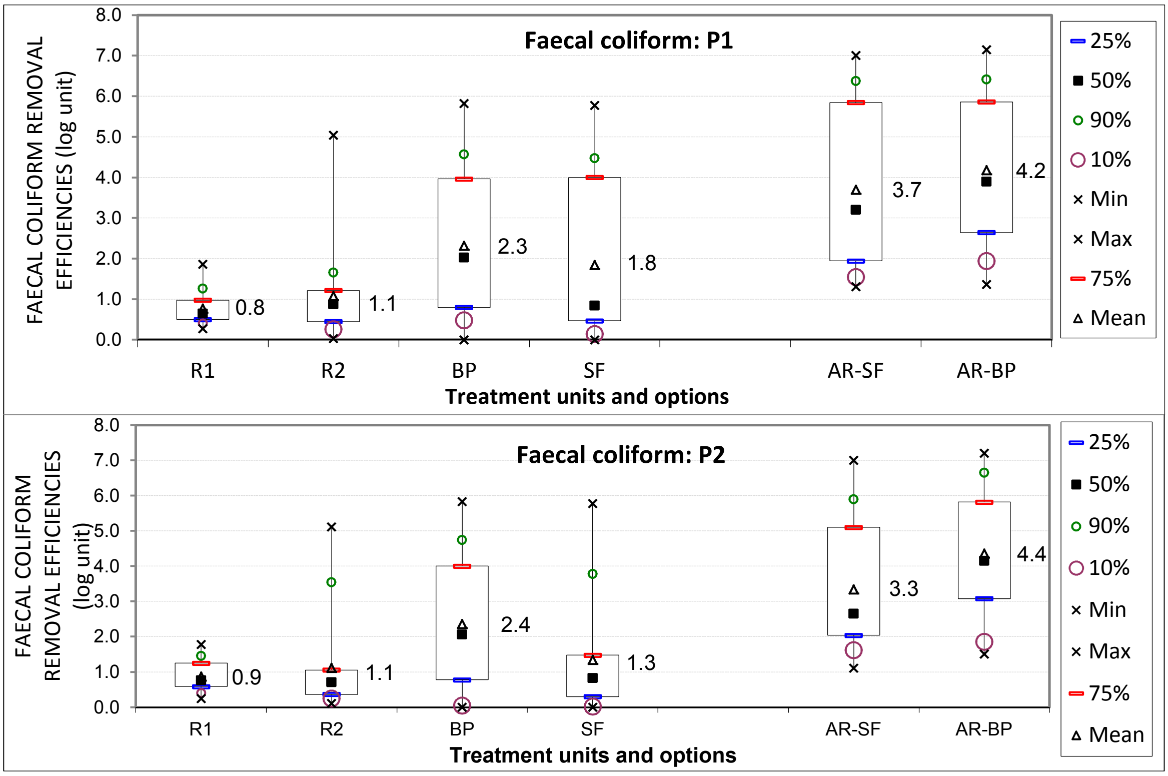


Figure 2.12b: Faecal coliform removal efficiency of each treatment unit and combination, for Periods 1 and 2

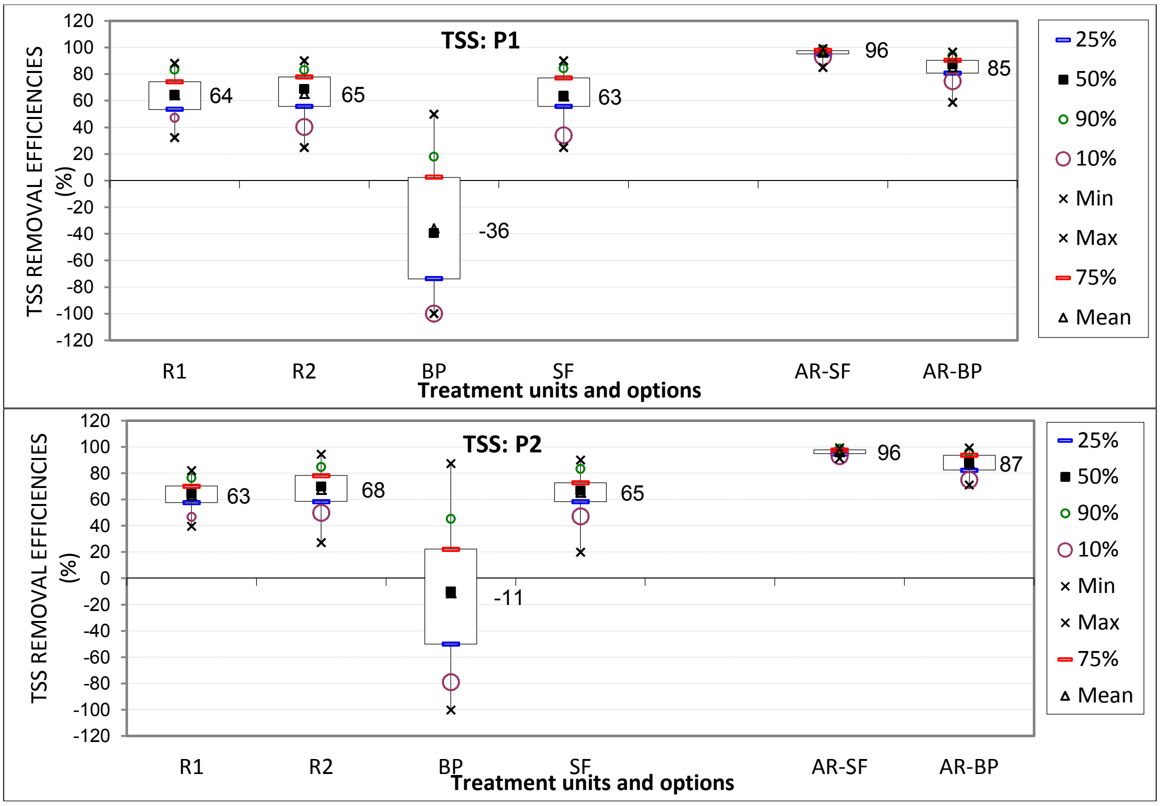


Figure 2.12c: Total Suspended Solids (TSS) removal efficiency of each treatment unit and combination, for Periods 1 and 2

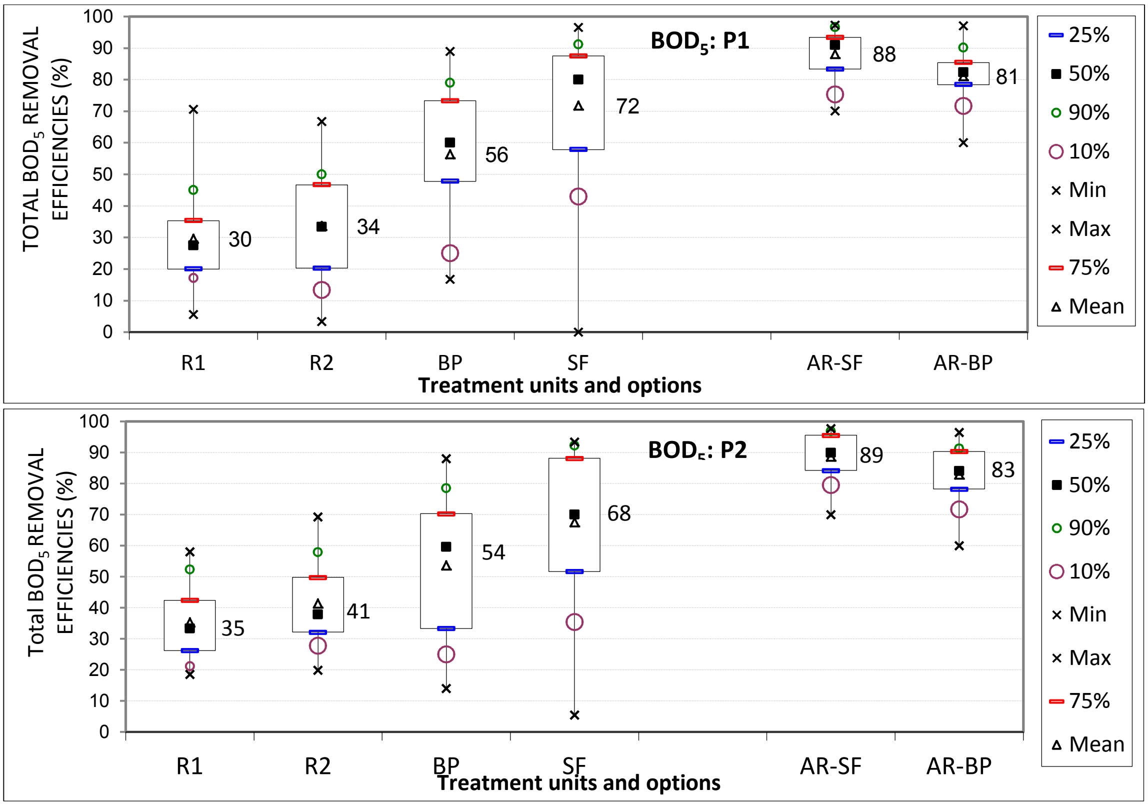


Figure 2.12d: Removal efficiency for Total Biochemical Oxygen Demand (BOD_5) of each treatment unit and combination, for Periods 1 and 2.

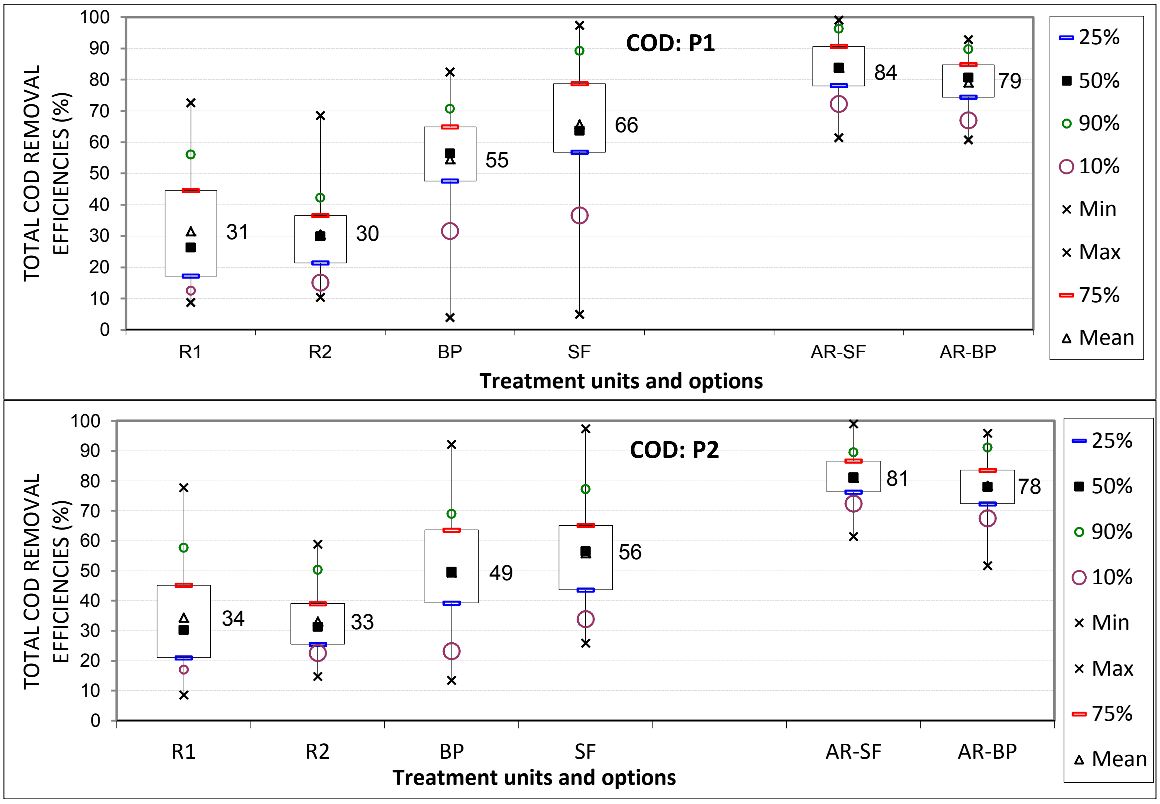


Figure 2.12e: Removal efficiency for Total Chemical Oxygen Demand (COD) of each treatment unit and combination, for Periods 1 and 2.

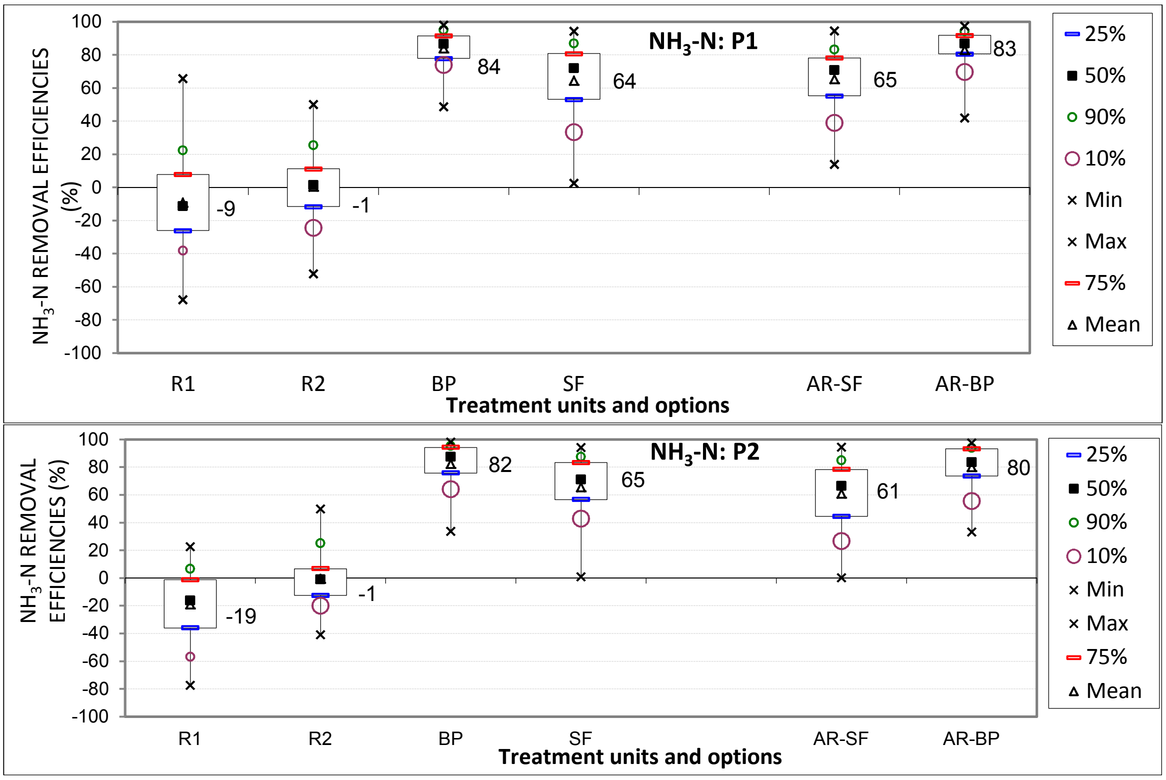


Figure 2.12f: Removal efficiency for Ammonia (NH_3-N) of each treatment unit and combination, for Periods 1 and 2

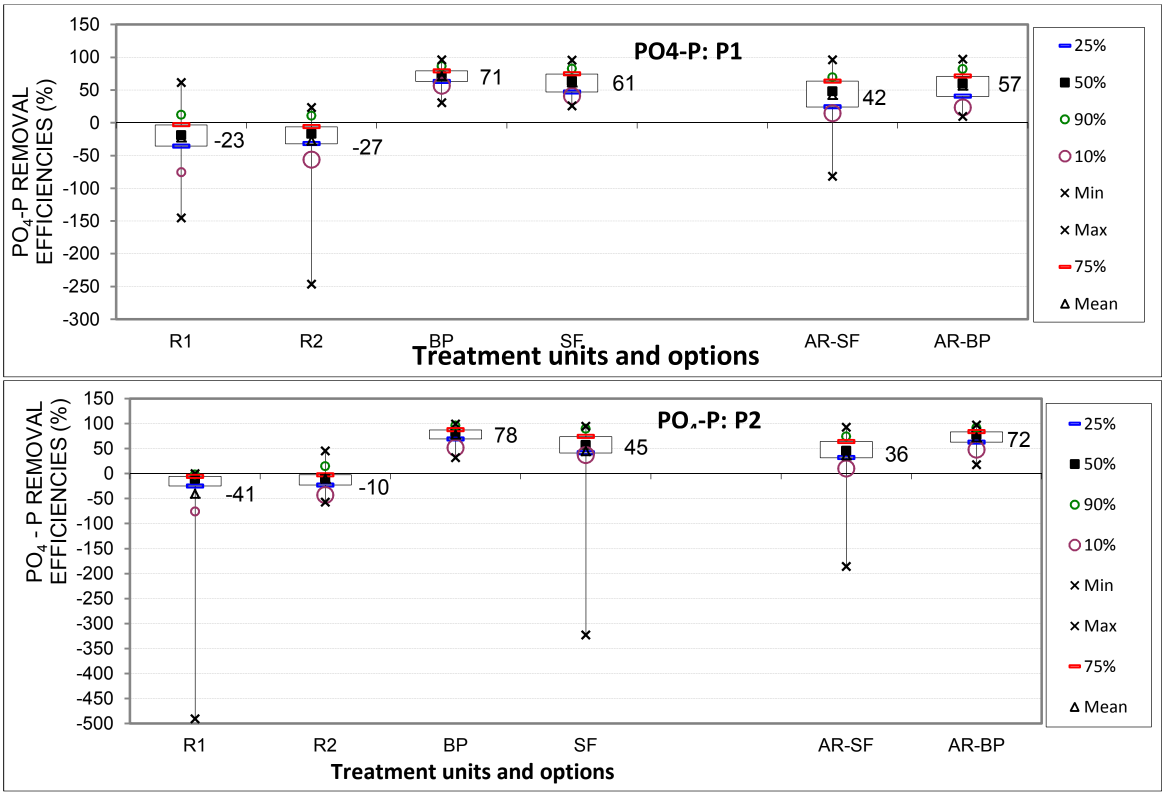


Figure 2.12g: Removal efficiency for Orthophosphate (PO_4-P) of each treatment unit and combination, for Periods 1 and 2.

Table 2.10: Removal efficiencies of *E. coli*, Total Suspended Solids (TSS), Biochemical Oxygen Demand (BOD_5) and Chemical Oxygen Demand (COD) for the two treatment combinationsand similar processes reported in the literature.

	E. coli	TSS	BOD_5	COD
	Log unit	%	%	%
R1+R2+BP*	4 – 7	71 - 94	60 - 97	61 - 93
R1+R2+SF@	1 – 7	91 - 99	70 - 97	61 - 99
WSPs (AP+FP+MP)[a] (Maiga *et al*. 2006)	1 - 6	66	87	81
UASB+SSF[b](Khan *et al*. 2013)	1 - 3	91	92.6	91
UASB + Constructed Wetlands (Khan *et al*. 2013)	1 - 2	65	-	82
Chlorination/ozonation[d] (Van der Steen 2008)	2 - 6	NA	NA	NA
UASB+POST[e] (Oliveira & von Sperling 2011)	1 - 6	82	88	77

* R1+R2+BP=Anaerobic reactor N°1 + Anaerobic reactor N°2 + Baffled Pond

@ R1+R2+SF=Anaerobic reactor N°1 + Anaerobic reactor N°2 + Sand filter

[a]Waste Stabilisation Ponds (WSP) =Anaerobic Pond + Facultative Pond + Maturation Pond

[b] Slow Sand filter

[e]UASB + POST= Upflow Anaerobic Sludge Blanket includes as post-treatment: aerated filter; anaerobic filter; trickling filter; flotation unit; facultative pond or maturation pond.

2.4 Conclusions

The design, implementation and evaluation of the two-stage high-rate anaerobic reactors followed by a baffled pond with attached growth or a wet-dry sand filter for domestic wastewater treatment in a sub-Saharan Africa warm and dry climate of Ouagadougou were conducted and optimized. Based on the results obtained in this study, the two combinations have revealed efficient and attractive alternative to treat wastewater in West Africa, under a Sahelian climate. In this study, a great variability was noticed in the effluent concentrations and in the removal efficiencies, considering all analysed constituents and all treatment units. It was found out that high pathogen removal efficiencies were achieved in both treatment options. Moreover, the anaerobic reactors coupled with the sand filter option presented high nitrification rate, while the Baffled Pond with attached-growth revealed a better efficiency in ammonia nitrogen and *E. coli* removals. Furthermore, no *E. coli* were ever detected in effluent of the Baffled Pond, nor did clogging occur in the Sand Filter, during the entire study. Based on the outcome of this research, it was concluded that, both treatment options could be applied as alternative low-cost wastewater treatment technologies for African cities and it is recommendable to reuse the effluent for restricted peri-urban irrigation.

However, because of the fact that the AR-BP has raised more unanswered questions, and by taking into account that the reuse of AR-SF effluents may be detrimental to plants (high pH, NO_3), if great care is not taken. It is also worth noting that a large amount of wastewater may be lost during the feedings of the sand filter due to evaporation, hence the following chapters will mainly focus on the AR-BP option. Therefore, in order to gain as much as possible an insight into this combination (AR-BP), the following chapters will stress: the hydraulic performance of the baffled pond, the complete elimination of *E. coli* in the baffled pond, the diversity and biomass of algae and zooplankton in the biofilm that develops on the plastic bottle caps affixed on the baffles, and the potentials of biogas recovery from the Anaerobic Reactors 1 and 2.

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Chapter 3

3. Hydraulic performance of the baffled pond and its control

In order to predict a wastewater pond's behavior, one would ideally have a complete picture of how the fluid passes through it. Since this is impractical, another approach is to find out how long individual particles remain in the pond. This can be determined easily and directly by the widely-used tracer test (Levenspiel, 1999). The objective of this chapter was to investigate the effect of baffles with increased surface area on the hydraulic characteristics of a baffled pond under Sahelian conditions. To achieve this, a tracer test was carried out on baffled and unbaffled ponds using common kitchen salt (sodium chloride). Furthermore, the analyses of the hydraulic characteristics and the models that could be applied in predicting the performance of the ponds in terms of the removal of organic matter and pathogens are considered.

3.1 Hydraulics and wastewater treatment

3.1.1 The need for hydraulic analysis

Wastewater treatment performances are influenced by many factors, such as the geometry of the treatment reactor, the location of inlet and outlet of the reactor, the loading regime, the climatic and the environmental conditions (Safieddine, 2007; Badrot-Nico et al., 2009; Abbassi et al; 2010). Another important parameter that has proved to affect the overall treatment efficiency is the hydraulic behavior of the wastewater treatment reactor (Nameche and Vasel, 1998; Shilton et al., 2000; Metcalf & Eddy 2003; Short et al; 2010; Babu, 2011). Moreover, numerous studies have found that the removal of pollutants within a wastewater treatment reactor (e.g., a wastewater pond) occurs via a diverse range of interactions between the sediments, litter, substrate, microorganisms, and the wastewater as it moves through the system. Therefore, the dynamics of water movement through the reactor has a significant influence on the efficiency and extent of these interactions. Moreover, many of the important biogeochemical reactions rely on contact time between wastewater constituents and microorganisms and the associated substrate, whereas wastewater velocity can be an important determining factor for other pollutant removal processes, such as mass transfer (Metcalf & Eddy, 2003; Małoszewski *et al*., 2006; Chang et al., 2011). Consequently, any short-circuiting or dead zones that occur within a reactor will have an effect on contact time, flow velocities and, therefore, treatment efficiency.

The introduction of baffles (with various configurations or wind-induced mixing) in ponds has been shown to improve their performance in terms of hydraulic and treatment efficiency, compared to unbaffled ponds (Kilani and Ogunrombi, 1984; Muttamara and Puetpaiboon, 1997; von Sperling et al., 2002, 2003; Shilton and Harrison, 2003; Shilton and Mara, 2005; Banda, 2007; Babu, 2011; Olukanni and Ducoste 2011; Sah et al., 2012; Olukanni, 2013; Cortés-Martínez et al., 2014). However, many of the treatment plants that have been built do not perform hydraulically as designed, because of a lack of appreciation of the hydraulics of these reactors. Hydraulic analysis is a gateway to better modelling of wastewater treatment systems (von Sperling et al., 2002, 2003; Metcalf & Eddy 2003; Badrot-Nico et al., 2009). Thus, understanding the mixing and the transport mechanisms in wastewater basins is vital in improving treatment performance and design (Nameche and Vasel, 1998; Shilton et al., 2000; Abbassi et al; 2010; Short et al; 2010; Babu, 2011; Chang et al., 2011). Very few studies have been conducted specifically in the warm and dry climatic conditions of West Africa, to investigate at a pilot scale the combined impact of both baffles and attached-growth on the hydraulic performance of waste stabilisation ponds (WSP). In this regard, this study was to investigate the effect of baffles with increased surface area on the hydraulic characteristics of WSP under Sahelian conditions.

The principal technique by which researchers and engineers have gained information about the direction and velocity of water movement or energy carried by water is through the use of inert tracers. In addition, this is also an indirect way of identifying other hydraulic parameters, such as dispersivity, actual (measured) mean hydraulic retention time, dead volume; short circuiting index, dispersion number, and even porosity and hydraulic conductivity in pack medium through further analyses (Babu, 2011; Chang et al., 2011).

3.1.2 Tracer selection

Over centuries, a number of tracers have been used to determine the hydraulic performance of reactors. According to Metcalf & Eddy, (2003) and Chang *et al*., (2011), an ideal tracer should be representative, should not affect the flow, (i.e., it follows the same path as the water) and is easily detected. In addition, it must be conservative, inexpensive to analyse, and have low toxicity, high solubility, and low background concentration in the natural environment. It is clear from numerous studies that the most popular choices for tracers are isotopes, ions, and dyes. For instance, the most common tracers include stable nitrogen isotopes ^{15}N, stable oxygen isotope $^{18}O/^{16}O$ ratio, Congo Red, fluorosilicic acid (H_2SiF_6), hexafluoride gas, lithium chloride (LiCl), potassium permanganate, rhodamine WT, and sodium chloride (Metcalf and Eddy, 2003; Lin et al., 2003; Dierberg and DeBusk, 2005; Kadlec et al., 2005 ; Małoszewski et al., 2006; Ronkanen and Kløve 2007, 2008; Camargo-Valero, 2008; Babu, 2011; Chang et al., 2011). Although isotope technology has high accuracy, it has been reported to be expensive (Chang et al., 2011). On the other hand, ionic compounds, especially bromide and lithium chloride, have been widely used due to their ease of analysis, as well as their common application in studying natural systems (Metcalf and Eddy, 2003). Other type of tracers like, rhodamine WT was found to be sensitive to light and temperature and, in addition, requires a fluorometer, which is expensive. Sodium chloride is the cheapest and easiest option, despite its tendency to form density currents, but this can be avoided by the correct mixing of the tracer (Metcalf and Eddy, 2003).

In tracer studies, a tracer is injected in the reactor (pond) influent and its concentration in the effluent is recorded in a series of grab samples collected at specific time intervals

(Metcalf and Eddy, 2003). Two methods are frequently used to introduce the tracer into the reactor: the Step Method and the Pulse Method, with the choice depending on the influent and effluent configurations. The Step Method involves a continuous addition of the tracer until the effluent concentration is equal to the influent concentration. In the Pulse Method, the tracer is added for a short period of time, usually less than the theoretical retention time. Tracer response curves are usually used to analyse the hydraulic characteristics of reactors, because of the complexity of the hydraulic flow pattern of the reactors. More details about the tracer response curves analysis are available in the books of Levenspiel, (1999) and Metcalf & Eddy, (2003).

3.1.3 Patterns of flow in ponds

The various pond systems that exist differ from each other in geometry, hydraulic flows, important biochemical processes, and hence their performance in pollutant removal. To develop pollutant removal models, the concepts of local pollutant reduction are combined with wastewater treatment hydraulic considerations and biochemical processes (von Sperling et al., 2002, 2003; Sah et al., 2012). Conceptual modelling of the hydrodynamic regime is widely used to predict the mean hydraulic retention time and the performance of wastewater ponds (Badrot-Nico et al., 2009; Olukanni and Ducoste 2011; Sah et al., 2012; Olukanni, 2013; Cortés-Martínez et al., 2014).

Generally, three different regimes are considered when designing or modelling the hydraulic regimes of ponds: Plug Flow (PF), Completely Stirred Tank Reactor (CSTR), and Dispersed Flow (DF). More details about these models are presented in **Table 2.4** of **Chapter 2**. The two idealised and extreme models of CSTR and PF are frequently assumed, but according to von Sperling (2002), the CSTR conceptual model is valid only for length to width ratios not significantly greater than unity. When the ratio of the length to the width exceeds unity, von Sperling (2002, 2005) recommends considering a DF regime. Moreover, the ideal flow regimes (PF and CSTR) are hard to achieve in practice, since there are many factors leading to non-ideal flow in ponds reactors: short circuiting, temperature differences (density current: cold or hot), wind-driven circulation patterns, mixing, pond geometry; axil dispersion (advection and dispersion). The acceptability of the Completely Stirred or Plug Flow models is based on simplicity rather than what actually takes place in the ponds.

Non-ideal flow regimes are currently more and more used for better representation of real pond hydraulics. Agunwamba et al., (1992); Levenspiel, (1999); Małoszewski et al., (2006) and Kadlec & Wallace (2009) describe reactors with non-ideal flow using the Dispersion and Mixers-in-Series Models. The Dispersion Model is regarded as a process between PF and CSTR, which are deemed as two extremes of the hydraulic patterns in unit operation. Depending on the intensity of intermixing, the prediction of this model ranges from PF at one extreme and CSTR at the other (The number of reactors in series (N) tends to infinity and N equal to unity respectively in **Equation 3.7**; Kadlec & Wallace, 2009). The Mixers-in-

Series Model considers the system to be divided into a series of mixed reactor tanks. The dispersion number (d) and the number of reactors in series (N) are the parameters used in describing the Dispersion Model and the Mixers-in-Series Model respectively. Prior to a tracer test, the behavior of the ponds was not known, so, both models were tested to see which fit the experimental tracer data best.

The parameters for the Dispersion and the Mixers-in-Series Models (d and N) can be calculated from the tracer response curve using mean retention time (t_m) (**Equation 3.1**) and variance (σ) (**Equation 3.2**). A detention time distribution (DTD) is commonly used to represent the time (t_i) that various fractions of water spend in the pond (Metcalf & Eddy, 2003; Levenspiel, 1999).

The mean retention time (t_m) is approximated if the concentration versus time tracer response curve (C) is defined by discrete time measurements as shown in **Equation 3.1** below

$$t_m = \frac{\sum t_i C_i \Delta t_i}{\sum C_i \Delta t} \quad (3.1)$$

Where t_m= mean detention time based on discrete time step measurements (days)
t_i= time at measurement (days)
C_i= tracer concentration measurement (g/m^3)
Δt_i= time increment (days)

$$\sigma_c^2 = \frac{\sum t_i^2 C_i \Delta t_i}{\sum C_i \Delta t} - t_m^2 \quad (3.2)$$

Where σ_c^2= variance based on discrete time measurements

- **Dispersion model**

The following equations (2.3, 2.4 and 2.5) are used to determine the dispersion number (d) as described by Metcalf & Eddy, (2003) and Levenspiel, (1999).

$$d = \frac{D}{vL} \quad (3.3)$$

Where d = dispersion number, (no units)
D = coefficient of dispersion (m^2/s)
v = fluid velocity (m/s)
L = reactor length (m)
Metcalf & Eddy, (2003) has described the relationships among variance of normalized tracer responseσ_n^2, variance σ_c^2 derived from tracer response curve (C), mean detention time (t_m), and dispersion number (d) for a pulse tracer input as follows:

$$\sigma_n^2 = \frac{\sigma_c^2}{t_m^2} = 2\frac{D}{vL} \quad (3.4)$$

$$2d=\frac{\sigma_c^{\,2}}{t_m^{\,2}} \quad (3.5)$$

Where σn^2 = variance of normalized tracer response C curve

σc^2 = variance derived from curve C

t_m= mean retention time

According to Metcalf and Eddy, (2003), the calculated dispersion values are used to explain the degree of axial dispersion in wastewater treatment as presented in **Table 3.1**.

Table 3.1: Description of axial dispersion values

d	Degree of Axial Dispersion
0	ideal plug flow
<0.05	low dispersion
0.05-0.25	moderate dispersion
>0.25	high dispersion
approaches infinity (∞)	system is considered completely mixed

Source: (Metcalf and Eddy, 2003)

Moreover, other mathematical transport (**Equation 3.6**) has been reported by Małoszewski et al., (2006) to best describe the hydraulic flow patterns in ponds. This includes the Multi-Flow Dispersion Model (MFDM) and Single-Flow Dispersion Model (DM). The MFDM depicts tracer transport along several individual flow paths (characterized by different water velocities and dispersivities), while the DM assumes convective-dispersive transport along a single flow path.

$$\alpha_{L_i} V_i \frac{\partial^2 C_i}{\partial x^2} + V_i \frac{\partial C_i}{\partial x} = \frac{\partial C_i}{\partial t} \quad (3.6)$$

Where: Ci (t) is the concentration of tracer in the effluent from the i^{th} flow-path, and, α_{Li} and v_i are the longitudinal dispersivity and the mean water velocity for the i^{th} flow path, respectively, x is the length of the flow-path and t is the time after injection.

The solution of **Equation 3.6** for an instantaneous injection was more elaborated by Małoszewski et al., (2006).

- **Mixers-in-Series Model**

The Mixers-in-Series Model is reported by Kadlec and Wallace, (2009) to be a gamma distribution of detention times given in **Equation 3.7** below:

$$g(t) = \frac{N}{t_m \Gamma(N)} * (\frac{N_t}{t_m})^{N-1} * exp(-\frac{N_t}{t_m}) \quad (3.7)$$

Where: Γ(N)=gamma function of N , = (N-1)!, factorial, if N is an integer, d^{-1}

N=number of tanks (shape parameter), unitless

t = detention time, (days)

t_m = mean detention time, (days)

When N= 1, the gamma distribution becomes the exponential distribution. Both the gamma distribution and the gamma function are readily available as computer spreadsheet tools (GAMMADIST and GAMMALN in Excel™). **Equation 3.7** may easily be fitted to tracer data by selecting N and t to minimize error by using SOLVER in Excel™.

The number of complete mixed reactors in series (N) can also be calculated using the mean retention time (**Equation 3.1**) and variance (**Equation 3.2**) derived from tracer response curve as in the **Equation 3.8** below:

$$N=\frac{{t_m}^2}{{\sigma_n}^2} \qquad \textbf{(3.8)}$$

Where N = number of mixers in series.

- **Other hydraulic characteristics**

The dead volume, an index of short circuiting and the volumetric efficiency of the pond reactor, can be determined using **Equations 3.10, 3.11 and 3.12** respectively (Levenspiel, 1999)

After determination of the mean theoretical detention time (t_{HRT}) with Equation 2.9, the fraction of dead volume can be calculated as:

$$t_{HRT}=\frac{V_r}{Q} \qquad \textbf{(3.9)}$$

$$Dead\ volume=\left(1-\frac{t_m}{t_{HRT}}\right)*100\% \qquad \textbf{(3.10)}$$

Where: t_{HRT} = theoretical hydraulic retention time (days), V_r= volume of the reactor (m^3), Q= flow rate (m^3/day) and t_m = mean detention time (days)

The index of short-circuiting (α_s) indicates how fast the influent reaches the effluent point. It is expressed as values that range from 0 to 1. When α_s approaches 1, the extent of short circuiting can be considered large.

$$\alpha_s=\frac{t_m-t_p}{t_m} \qquad \textbf{(3.11)}$$

Where: α_s= index of short-circuiting and t_p=time taken to reach the maximum tracer concentration.

The volumetric efficiency can be calculated by dividing the mean hydraulic retention time (t_m) by the mean theoretical detention time (t_{HRT}).

$$Volumetric\ efficiency=\frac{t_m}{t_{HRT}} \qquad \textbf{(3.12)}$$

3.2 Methodology

3.2.1 Pilot plant description

The Baffled Pond with attached-growth and the control pond used in this study were described in **Chapter 2** (**Figure 2.4**). The operational conditions were also as described in **Chapter 2**. After evaluating the performance of the two options (AR-BP and AR-SF) of the pilot plant, and because of the fact that the AR-BP option has exhibited higher pathogen removal with zero level detection of *E. coli* (**Chapter 2**), there was a need to study the effect of baffles on the hydraulic characteristics of the baffled pond. This is addressed in this chapter by comparing the tracer results of the Baffled Pond with those of the Control Pond.

3.2.2 Tracer experimental procedures

Due to the advantages of ease of detection and operation, low cost, low toxicity, high solubility, and inexpensive analysis, sodium chloride (common kitchen salt) was selected as the tracing substance to determine the hydraulic flow patterns and the actual HRT of both the Baffled Pond (BP) and the Control Pond (CP) in this pilot project.

First of all, the kitchen salt crystal (sodium chloride) was purchased at the local market in Ouagadougou and ground into a powder. The determination of the quantity to be used in the ponds was based on experience in Brazil (von Sperling Marcos, UFMG Belo Horizonto, Brazil). Since the background concentration of salt in domestic sewage is high, larger amount was required to observe peak conductivity (µs/cm) in the ponds. For example, a pond of 100 m^3 needs an addition of 320 kg of NaCl, which means a peak concentration of 3200 mg NaCl /l could be achieved. In addition, to the quantity of salt required for the tracer test, the correlation between salinity and conductivity should also be determined, since it is only the conductivity that would be measured *in situ*. Therefore, a laboratory test was conducted to determine the correlation between salinity and conductivity. The results show a good linear correlation of R^2=0.998 (**Figure 3.1**).

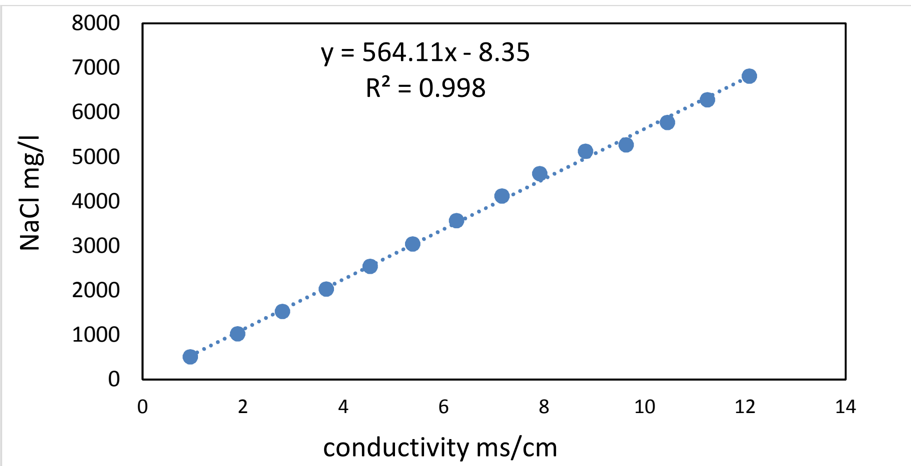


Figure 3.1: Results of a laboratory test (in 2iE) of the relationship between electrical conductivity and sodium chloride concentration

Twenty-four kilograms of kitchen salt powder (NaCl) was injected after dissolving it in 45 l of tap water, which was the volume necessary to completely dissolve it. The tracer solution was injected via the flow distribution box (**Figure 3.2**) by using a plastic bucket with faucet located at the bottom to regulate the outflow of the tracer solution. Chang *et al*. (2011) recommend injecting the tracer during less than 2% of the theoretical hydraulic retention time (HRT), which in this case was 7 days for both the BP and the CP, so it was decided to inject the tracer during 1 hour. During the injection, the dissolved salt solution was gently stirred to keep a homogeneous solution.

However, before injecting the NaCl, the electrical conductivity of both ponds was measured at three different depths: 15, 60, and 90 cm in each compartment of the BP and the corresponding positions in the CP. Then, the tracer test was performed simultaneously for the two ponds and was run for 12 days (from 25 May to 5 June, 2015). The measurements of the electrical conductivity and the wastewater temperature were taken manually and simultaneously every 5 minutes, during the pumping hours, three times a day at 8:00 am, 1:00 pm and 5:00 pm, at the outlets of both ponds (**Figure 3.2**). This task was continued, until the baseline conductivity values (initial values) were once again reached. Moreover, to appreciate the dispersion of the dissolved salt, the electrical conductivity of both ponds was measured at three different depths and at four positions along the longitudinal length (as done before the injection of the tracer), one day after this injection and the last day of the experiment. The flow rates through the BP and the CP were controlled by the crested weir and stayed practically constant during tracer test (Q= 0.5 m^3/d for each pond).

Figure 3.2: Injection of the kitchen salt (NaCl) tracer and measurements of the electrical conductivity at the outlets of the baffled and Control Ponds

The computational procedures for calculating the tracer actual (measured) mean hydraulic retention time, dead volume, short-circuiting index, dispersion number and number of reactors in series were carried out based on the Excel Spreadsheet model provided by professor von Sperling Marcos (UFMG , Belo Horizonto, Brazil), which was adapted from

Levenspiel, (1999); Metcalf and Eddy, (2003); and Kadlec and Wallace, (2009) books. In addition, the gamma distribution model (**Equation 3.7**) was used to evaluate the degree that the model fits with the experimental data. At last, the NaCl distribution in both ponds was calculated by the Surfer Software Package.

3.3 Results and Discussion

3.3.1 Evaluation of the tracer Test

The results of the tracer test with salt (sodium chloride, NaCl) were corrected for the background content of salt, based on measurements taken before the injection of the tracer. The data received from actual tracer studies from the field were plotted and found to be skewed with long tails (**Figures 3.3 and 3.4**). Various parameters were used to represent the hydraulic characteristics in the Baffled and Control ponds (**Table 3.2**). The hydraulic characteristics were calculated based on Levenspiel (1999); Metcalf and Eddy (2003); and Kadlec and Wallace, (2009) and the concentration and time data that were obtained (**section 3.1.3**). Tracer distribution patterns were drawn, at depths of 15, 60 and 90 cm, to depict the flow patterns in both the Baffled and Control ponds immediately before the injection of the tracer, one day after the injection and at the end of the experiment (**Figure 3.5**). The light green color of **Figure 3.5** shows lesser concentration of the tracer, while the blue to the dark blue colors indicates higher concentrations and the more concentrated contours lines represents uniform distribution of the tracer.

The flow patterns of the Baffled and Control Ponds were different from one another, implying that the baffles arrangement affected the flow pattern (**Figure 3.3**). This was also observed by Babu (2011) who explained this phenomenon as being due to the vertical baffles placed perpendicular to the flow direction seemed to perform best in reducing short-circuiting. However, the tracer movement towards the outlet seems to be fast at the beginning for both ponds. The tracer took about 3 hours 20 minutes to reach the first peak in the BP, while 4 hours was necessary to attain the first peak in the CP (**Figures 3.3 and 3.4**). One might have expected tracer progress towards the outlet to be slower in the BP, but this was not the case (**Figures 3.3 and 3.4**). This early arrival of tracer at the outlet of BP can be reasoned partially by remembering that the velocity distribution in the Baffled Pond was parabolic (Safieddine, 2007), and besides, the baffles sheets did not fit not tightly against the walls of the pond, so the wastewater could also flow around them. Moreover, similar observations were also reported by Safieddine, (2007) and Chang et al., (2011), who explained that this could happen when the location of the dead regions were towards the outlet of the pond. **Figure 3.5** presents a typical situation concerning the positions of the dead regions. This confirms the results previously reported by these researchers.

Another important issue from the flow distribution visualization techniques, which cannot be obtained by residence time distribution analyses alone, was the exact behavior of the tracer at certain points in time in the pond and the locations of probable dead zones (**Figure 3.5**).

For instance, after one day, most of the tracer was still in the first compartment and bottom layers of the Baffled Pond, whereas in the Control Pond it was seen to mainly distribute towards the outlet and top layers (**Figure 3.5**). A similar trend has also been reported by Olukanni and Ducoste (2011) Computational Fluid Dynamic (**CFD**) model. Furthermore, at the end the experiment, the Baffled Pond was able to achieved the initial condition, while this was not the case in the Control Pond, since more tracer concentrations was retained in the middle and bottom layers of the left side corners of the Control Pond, which could represent the dead zones (**Figure 3.5**). This agrees with the findings reported by some researchers who found that an increase in the number of baffles reduced the amount of dead water within the system (Kilani and Ogunrombi, 1984; Lloyd *et al*., 2003; Aldana *et al*., 2005). This confirms the advantage of installing baffles to improve hydraulic performance.

On the other hand, visual examination of the tracer response curves for both the Baffled and Control Ponds (**Figure 3.3**) shows that the curves were characterized by several peaks (maxima). Under quasi steady state hydraulic conditions during the experiment, this effect suggests that the tracer is transported along different flow-paths (Małoszewski et al., 2006). This agrees with the findings reported by Małoszewski et al. (2006) and Camargo-Valero, (2008) who found that the peaks might result from sampling and analytical uncertainties or from small unsteadiness of flow. In addition, critical analysis of the normalized tracer response curves strongly suggests that the hydraulic regime in both ponds was close to complete mixing and even better the surface area under the BP curve represents half of that the CP (**Figure 3.4**). These findings are in compliance with the results reported by Camargo-Valero, (2008) who reported that such tracer response curve depict a rapid mixing in the pond, involving a process with a very high reaction rate.

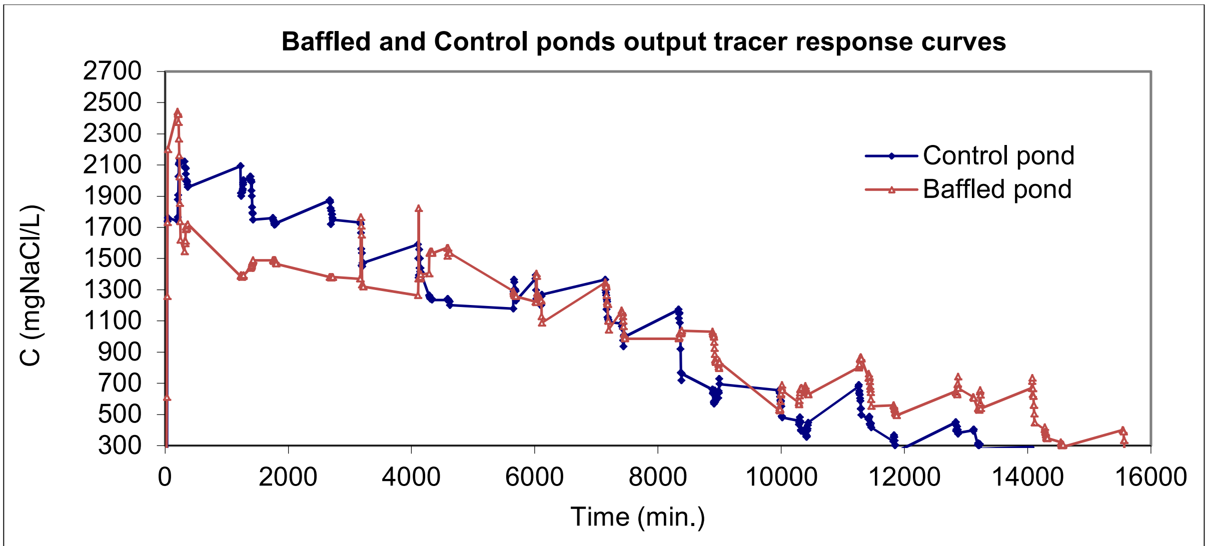


Figure 3.3: Tracer response curves showing the relationship between salt concentration and transit time in the Baffled and Control Ponds

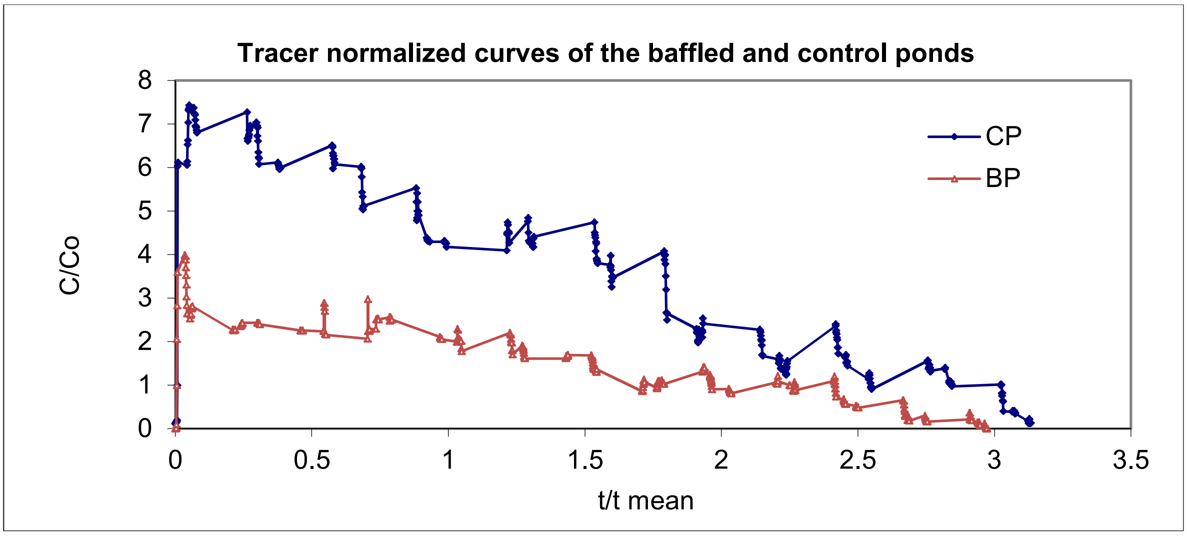


Figure 3.4: Normalised tracer salt concentration-transit time curves for the Baffled and Control Ponds

The actual mean hydraulic retention times (t_m) for both the BP and the CP (4.1 and 3.2 days respectively) were shorter than their theoretical hydraulic retention time (t_{HRT}) which was 6.6 days; **Table 3.2**). This has also been observed in previous tracer studies in ponds (Małoszewski et al., 2006; Camargo-Valero, 2008, Chang et al., 2011). The reason why the actual mean retention time was lower than the theoretical hydraulic retention time could be attributed to retardation of wastewaters in zones of stagnant flow (dead zones) and to the occurrence of preferential or bypassing flows (Małoszewski et al., 2006). However, when comparing the two ponds, it was seen that the actual HRT for the BP was higher than that of The CP. Such a discrepancy may be due to the longer distance and travel time created by the baffles (Shilton and Harrison, 2003), which increased the mean retention time by approximately 1 day (22 % increament). A similar conclusion was reached by Babu, (2011) who found that by installing 15 baffles perpendicular to the influent flow directionin maturation ponds, the HRT was increased by about 2 days.

Table 3.2: Parameters used to describe the hydraulic characteristics of the Baffled and Control Ponds

Parameters	**Baffled Pond (BP)**	**Control Pond (CP)**
Theoretical HRT (t_{HRT}, days)	6.6	6.6
Actual HRT (t_m, days)	4.1	3.2
Number of mix reactors in series (N)	1	2
Dispersion number (d)	0.53	0.66
Volumetric efficiency (%)	62	49
Dead volume (%)	38	51
Recovery of tracer (%)	231	229

The data processed showed that the BP and the CP were described by 1 and 2 mixed reactors in series respectively (**Table 3.2**). Furthermore, they were found to have high dispersion numbers (0.5 and 0.6 for the BP and the CP respectively). According to the Mixers-in-Series Model, the BP and the CP can be respectively described by 1 and 2 mixed reactors in series with high dispersion (Levenspiel, 1999; Metcalf and Eddy, 2003; Kadlec and Wallace, 2009). This finding was opposite to that of Kone et al. (2002), who reported plug flow with dispersion in a facultative pond operated under the same climatic conditions. Such disparity of flow regime could be due the type of tracer used to perform the test.

The percentage of dead zones in the BP and the CP was found to be 38% and 51% respectively. This shows that the volume of the BP pond was more efficiently used than the volume of the CP pond, in which more than half of the total volume was found to dead zones. Once again, it can be concluded that the addition of baffles with plastic bottle caps affixed to it can increase the surface area for biofilm growth, and they can also have a positive effect on pond hydraulics (Kilani and Ogunrombi, 1984; Lloyd *et al*., 2003; Shilton and Harrison, 2003; Aldana *et al*., 2005; Babu, 2011; Olukanni and Ducoste, 2011).

The percentage recovery of NaCl was greater than 200%, which means that the total concentrations of NaCl measured at the end of the experiment were greater than the total initial NaCl input (**Table 3.2**). A similar observation was also reported by Babu, (2011), who explained that this could be accounted for by the analytical grade of the tracer. Another possible explanation may be related to the high background content of salt in the domestic wastewater before and during the experiments. In light of this high tracer recovery, it is evident that loss of water from the ponds through seepage into the ground is negligible.

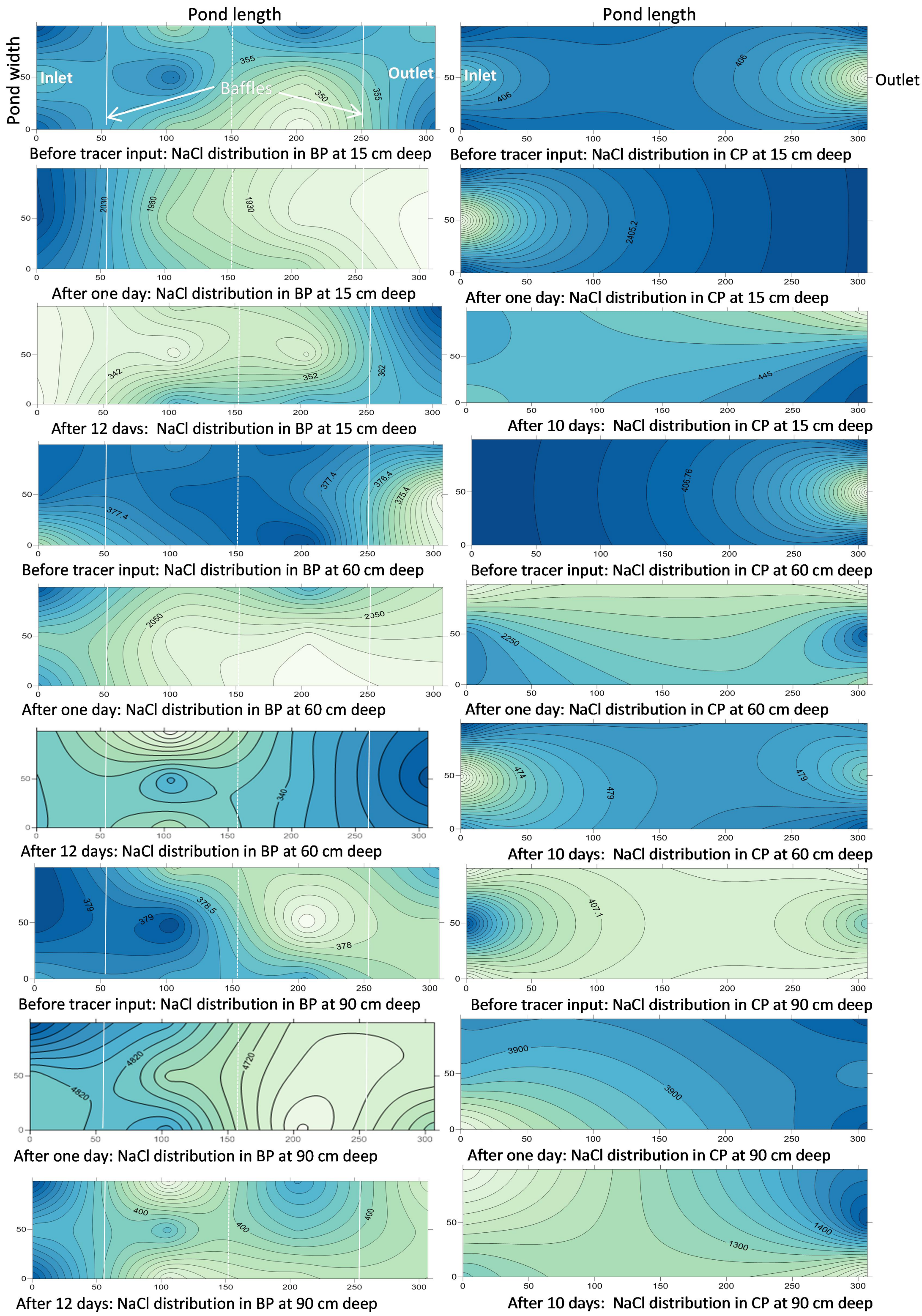


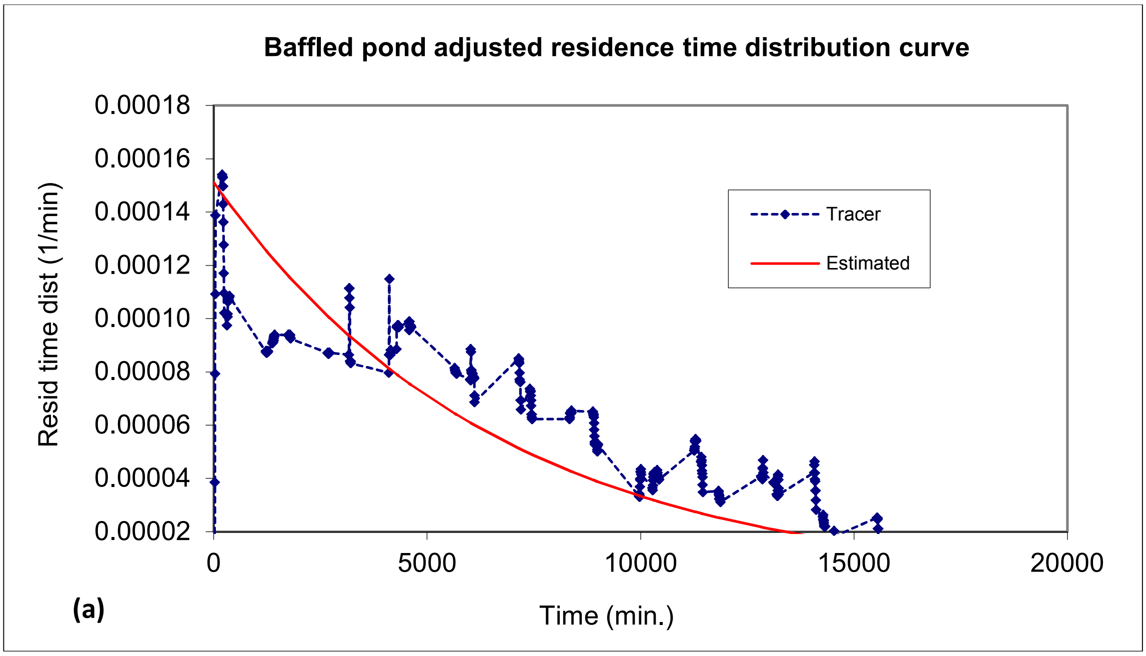
Figure 3.5: Top view of tracer concentrations at 15, 60 and 90 cm deep in the Baffled Pond (left) and the Control Pond (right); before the tracer was added, one day after injection and at the end of the experiment.

Note: Light green indicates less tracer and dark blue indicates more.

3.3.2 Model fitting

The flow regimes in the Baffled and Control Ponds can be characterized by selecting the curve that most closely fits the experimental tracer response curves in the family of theoretical curves, such as the gamma distribution model, the multi-flow dispersion model, the single-flow dispersion model (**Equations 3.6 and 3.7**) and many others mathematical transport models. Hence, in this study the Excel Spreadsheet model provided by professor Marcos von Sperling Marcos (UFMG, Belo Horizonto, Brazil), was used to evaluate the index of the model fitting process of the experimental data with the gamma distribution model **(Figure 3.6**). The tracer data of the Baffled Pond best fit the estimated data from the gamma distribution much better than those of the Control Pond (**Figures 3.6a and b**). However, in both cases, the real flow pattern deviates slightly from ideal behavior depicted by the gamma distribution model. For the Control Pond, a better fit in the tail rather than the peak area was achieved (**Figure 3.6b**), whereas that of the Baffled Pond approximately follows the model but deviates somewhat towards the tail (**Figure 3.6a**). The model fitting data from this exercise showed for N=1 (number of mixed reactors in series) and d=0.53 (dispersion number), a correlation coefficient of 0.451 was achieved in the Baffled pond, while in the control pond, for N=2 and d=0.66, a correlation coefficient of 0.423 was obtained. This fairly weak correlation could likely be attributed to the sampling measurement periods, since the tracers concentrations were only recorded during pumping hours, which corresponded to the outflow times. Similar observations were also reported by Chang et al., (2011), who found that sampling period had an effect on the tracer response curve.

However, despite this weak correlation, one can assert that the Dispersion and Mixed reactors-in-Series Models are basically suitable for cases in which the real flow pattern deviates only slightly from ideal behavior (von Sperling et al., 2002, 2003, 2005; Małoszewski et al., 2006; Safieddine, 2007, Olukanni and Ducoste, 2011; Olukanni, 2013; Cortés-Martínez et al., 2014). Furthermore, because the Dispersion Model is more flexible and is able to represent all types of reactors and besides, the first-order decay (K) coefficients derived from that model are assumed to best represent the reality of reaction kinetics (von Sperling et al., 2002, 2003, 2005). Thus, the first-order organic and fecal coliform removal rates for each pond could therefore be calculated using the classical equations of Wehner and Wilhelm (1956) given by **equations 2.19, 2.20** and **2.21** (**Table 2.4 in Chapter 2**), since the actual hydraulic retention times and the dispersion numbers are known.



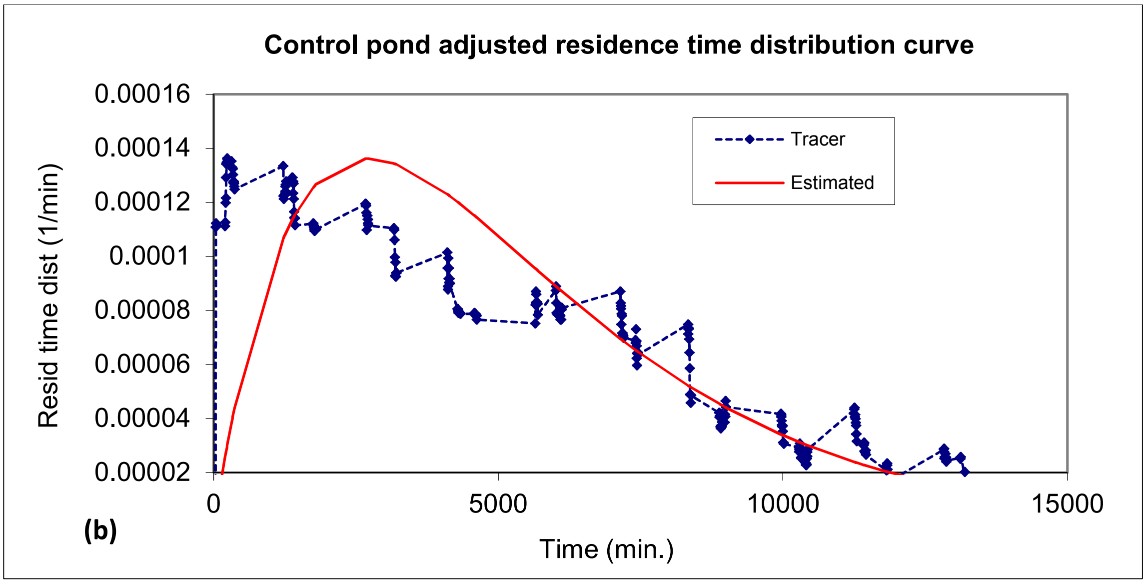


Figure 3.6: Comparison between the actual tracer residence time distributions function and the Mixed –Reactors-in-Series Model (Gamma distribution model)

3.4 Conclusions

The tracer study was carried out to confirm the effect of baffles with a roughened surface on the hydraulic patterns by comparing the results of the tracer response curves of a baffled pond and its control under Sahelian conditions. The actual mean hydraulic retention times for the BP and the CP were 4.1 and 3.2 days respectively. This indicates that by introducing three verticals baffles in a pond receiving an anaerobically treated effluent could increase the mean retention time by approximately 1 day (ie. 22 % increased). Therefore, these findings show that there is significant potential for size reduction and cost optimization to be achieved by the incorporation of properly designed baffles in ponds in tropical climates. Moreover, it was found that the volume of the Baffled Pond was more efficiently used for wastewater treatment than the unbaffled Control Pond, since more half of the volume of the latter was calculated to be an inactive, 'dead' zone. The actual mean retention times based on the tracer study calculations for each pond was lower than the theoretical mean retention times. A possible explanation could be attributed to the slowing of wastewaters in zones of stagnant flow and to the occurrence of preferential or bypassing flows. On the other hand, the tracer experiments showed that the outcome of the fluid flow pattern of these ponds can be fairly approximated as resulting of one and two mixed reactors in series with high dispersion for the Baffled and Control Ponds respectively. Consequently, both Dispersed and Mixed-Reactors-in-Series Models are more appropriate for predicting the performance of the Baffled Pond. Because of the high tracer recovery from both ponds, one can conclude that the loss of water through seepage is negligible and that sodium chloride can be considered an economical and conservative tracer.

3.5 References

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Chapter 4

4. *E. coli* distribution and its removal in a Sahelian Baffled Pond

The performance evaluation (**Chapter 2**) revealed zero detection of *Escherichia coli* in the effluent of the Baffled Pond with attached growth during the two years of operation with two different hydraulic flow rates. The aim of this chapter was to investigate and understand this lack of detection of *Escherichia coli*. Furthermore, the die-off rates of *Escherichia coli* in both the Baffled and Control Ponds were derived based on the Dispersed-flow Model obtained from the tracer test (**Chapter 3**). The importance of these baffles with roughened surfaces is also discussed in greater depth here.

4.1 Removal mechanisms of *E. coli* in WSP

New concepts have to be developed to control the growing problems of water pollution in developing countries (Parker, 1988). For this reason, many researchers have strived to improve existing wastewater treatment systems. Wastes Stabilization Ponds (WSPs), for example, have undergone many innovations. Some studies discussed and demonstrated the role of baffles in WSPs and have concluded that they improve the hydraulics and increase the efficiency of organic and pathogens removal (Kilani and Ogunrombi, 1984; Muttamara and Puetpaiboon, 1997; von Sperling et al., 2003; Ouali et al., 2012). A number of previous studies have also looked at the optimization aspects of WSPs with respect to many factors such as cost and efficiency of treatment plant construction and operation (Oke and Otun 2001; Olukanni and Ducoste, 2011; Cortés-Martínez et al., 2014). In a study conducted by Shin and Polprasert (1987, 1988) have shown that the introduction of artificial media for biofilms to attach to within the pond water enhance the performance of WSPs in terms of the removal of organics, ammonia and suspended solids.

Furthermore, WSPs have been an important research area over the past two decades to understand the driving processes of disinfection in these systems. Indeed, *E. coli* and faecal coliform bacteria have been widely used as biological indicators for monitoring wastewater treatment systems. Previous studies have shown that the main factors driving disinfection (*E. coli* decay) in ponds are: solar radiation (especially UVB and UVA) (von Sperling, 2005; Davies et al., 2009 ; Nelson et al., 2009 ; Bolton et al., 2010; Ouali et al., 2012, 2014), pH, dissolved oxygen, temperature, ammonia content (Curtis et al., 1992; Davies-Colley et al., 1999; van der Steen et al., 2000; Oragui, 2003; Abis and Mara, 2006; Maïga et al., 2009; Buchanan et al., 2011; Ukpong, 2013), algal biomass (van der Steen et al., 2000), nutrient limitations, activity of bacteriophage viruses, water depth (Maïga et al., 2009), hydrodynamics of the ponds (Kilani and Ogunrombi, 1984; von Sperling et al., 2003).

Several studies on the effects of individual parameters from this list on microbial indicator species in aquatic systems have been conducted using laboratory or pilot plant conditions, but converged on the conclusion that a synergistic effect of physical, chemical and environmental factors were responsible for the inactivation of pathogens in these systems

(Curtis et al., 1992; Davies-Colley et al., 1999; Oragui, 2003; Maïga et al., 2009; Nelson et al., 2009; Bolton et al., 2010; Ouali et al., 2014).

For instance, Oragui, (2003) found that, in a two-meter-deep facultative pond, pathogen concentrations decreased largely in the top meter of the pond. He argued that this decline in the top layer was due to the combined effects of high pH and toxicity of ammonia and sulphide. Earlier studies carried by Curtis et al. (1992) on the exposure of faecal indicator microorganisms to high pH values in the absence of light have shown that pH alone was not toxic, unless extreme values were used, which would be hardly reached in pond systems. Similar experiments by the same authors showed that high concentrations of dissolved oxygen alone did not affect these microorganisms. It was concluded that a combination of high dissolved oxygen, solar radiation, high pH, and algal growth had a lethal effect on the microorganisms. Furthermore, Maiga et al., (2009) found a strong correlation between the inactivation of coliform bacteria and pond depth, with shallower ponds being more effective (t test, α=0.05). They also found that the damage caused to enteric pathogens was more pronounced in the presence of sunlight than in the dark.

On the other hand, various studies focused on the development of empirical models, which according to Bastos et al., (2011) followed a first-order kinetic, regardless of the mechanism of bacterial inactivation. However, besides this progress, no one has reported complete inactivation of *E. coli* in conventional or improved WSPs.

The current study analyses an alternative treatment system combining two high-rate anaerobic reactors in series, followed by a baffled pond with attached-growth on recycled plastic bottle caps, which was developed and tested under Sahelian climate at the 2iE Research Site in Ouagadougou, Burkina Faso (West Africa). The system was operated at a pilot scale under two different flow rates (1 and 1.5 m^3/d) and an average influent *E. coli* count of 5.6 x 10^7/100 ml (**Chapter 2**). This pilot plant was monitored for two years and no E*. coli* were ever detected in the effluent.

The objective of this study was to investigate and understand the non-detection of *E. coli* at the outlet of this improved WSP. Some of the most significant physical, chemical, and environmental factors which could be responsible for the inactivation of *E. coli* in the baffled pond were investigated, as reported by previous studies.

4.2 Methodology

4.2.1 Experimental setup

The Baffled Pond with attached-growth and the Control Pond used in this study are described in **Chapter 2** (**Figure 2.4**), together with their operational conditions. After evaluating the performance of the two options (AR-BP and AR-SF) of the pilot plant, and because of the fact that the AR-BP option exhibited higher pathogen removal with zero detection of *E. coli* (**Chapter 2**), there was an interest to understand the mechanisms that led to such important results. This was addressed by carrying out an intensive investigation on the distribution of *E. coli* in both the Baffled and Control Ponds, in order to give a better description and improved understanding of the non-detection of *E-coli* in the effluent of this Baffled Pond.

4.2.2 Sampling and analysis

A "shaft syringe" developed by Konaté et al., (2010) was modified and was used for sampling. The sampler was made of a 3 m plastic rod on which 60 mL syringes were fixed. The syringes were 15 cm equidistant from one to another and a string was fixed to each syringe to facilitate the wastewater sampling. Grab samples of 500 ml were taken weekly between 8:00 to 9:00 am, at three different depths (15, 60 and 105 cm) and in each compartment (A, B, C and D) of the pond during five months from March to July 2014 (**Figure 4.1)**. The collected samples were stored at 4ºC and analysed within 3 hours for *E. coli*, chlorophyll-A and total chemical oxygen demand (COD), according to APHA (2012).

The spread plate method was used to determine the presence and abundance of *E. coli* as an indicator microorganism in the samples and Chromocult Agar (Merck KGaA 64271, Darmstadt, Germany) was used as the culture medium. Chlorophyll-A was measured using the spectrophotometric method (APHA, 2012). It consisted of a sequential procedure of filtration, centrifugation and extraction of the chlorophyll by using an organic solvent (90% acetone). The extracted chlorophyll was then analysed by a spectrophotometer Hach Lange DR 5000 with 750 nm and 665 nm light absorption wavelengths. Dissolved COD was analysed by using potassium dichromate as a strong oxidizing agent and was measured with a spectrophometer at a wavelength of 620 nm (Hach Lange, DR 5000).

The pH, water temperature (T) and dissolved oxygen (DO) were measured *in situ* with an integrated portable pH-T-DO meter (WTW 340i/SET) fixed to a cylindrical iron rod carefully graduated. This device was previously calibrated using standard buffer solutions before any usage. The measurements were carried out during the pumping hours, three times a day at 8:00 am, 1:00 pm and 5:00 pm at seven different depths (15, 30, 45, 60, 75, 90 and 105 cm from top to bottom) and in each compartment (A, B, C and D) of the pond. The sampling positions are depicted on **Figure 4.1**.

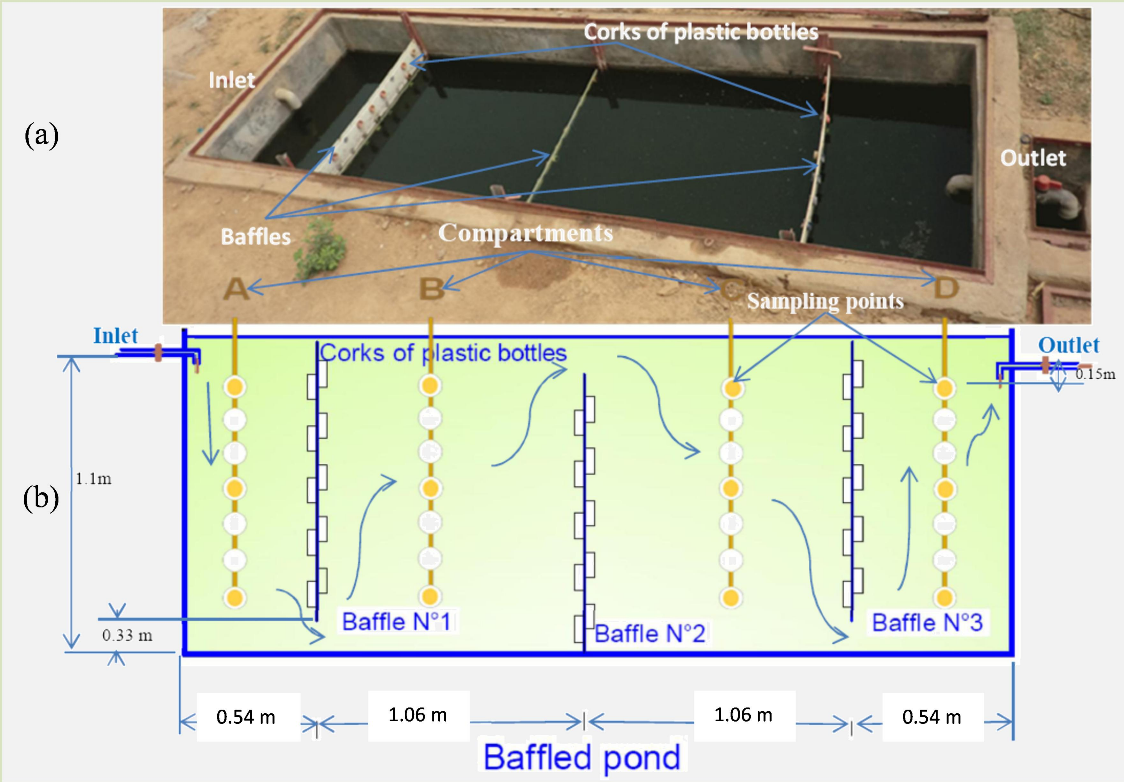

Figure 4.1: (**a**) Photograph of the Baffled Pond (**b**) Side view of the Baffled Pond with attached plastic bottle caps and the sampling points

(Note: The three orange sampling points in each compartment were for*E-coli*; Chlorophyll-A and COD, while all the seven sampling points in each compartment were for pH, temperature and dissolved oxygen.)

Tracer experiments (**Chapter 3)** indicated that it was reasonable to consider both ponds to be Mixed-Reactors-in-Series or Dispersion Model. However, because the Dispersed-flow Model is more flexible and is able to represent all types of reactors and besides, the first-order decay (K) coefficients derived from that model are assumed to best represent reality and the true reaction kinetics (Wehner and Wilhelm, 1956; von Sperling, 2002, 2003; von Sperling et al., 2003). The first order decay coefficients for each pond were therefore derived using the classical equations (Wehner and Wilhelm, 1956; **Equations 2.19, 2.20 and 2.21; Table 2.4 in chapter 2**).

4.3 Results and Discussion

A total of 240 samples were collected from different depths and in the four compartments of the Baffled Pond and tested for *E. coli*, Chlorophyll-A and COD.

4.3.1 Baffled Pond investigation

- ***E. coli* distribution pattern**

Arithmetic mean and standard deviation of *E. coli* from different depths and in the four compartments are listed in **Table 4.1.** Influent values ranged from 10^3 to 10^4 per 100 ml at the inlet and the final concentration of *E. coli* at the outlet was consistently <1 per 100 ml.

This pattern clearly indicates that the gradual decrease in *E. coli* concentration follows the theoretical water course in the pond. *E. coli* concentrations were consistently lower in the top 0.60 m of the water column, and higher in the lower layers of all four compartments (**Figure 4.2**, **Table 4.1**). *E. coli* concentrations diminished from the bottom toward the top of the pond and also from the inlet toward the outlet **(Figure 4.2**).

It is important to point out that from a depth of 0.60 m upward in Compartment D, no *E. coli* were detected throughout the entire experiment (**Figure 4.2**). Since the effluent of the Baffled Pond was collected from the top layer, at a depth of 0.15 m, no *E. coli* were detected in the effluent. Based on the relatively high concentrations of *E. coli* detected at the bottom, it seems that the environmental conditions of the upper layer were more favourable for the destruction of *E. coli*. Some of the *E. coli* removal in the Baffled Pond may have occurred through sedimentation of these bacteria through association with particulate matter, such as dead algae. Most of the settling occurred in the first compartments of the pond (**Figure 4.2**). This removal via sedimentation is clearly related to the hydraulic retention time (Stott, 2003; Konaté et al., 2010). This finding contrasts with the results of van der Steen et al., (2000) and Sarikaya et al., (1987b), who reported that after treatment in an anaerobic pond or anaerobic reactor, *E. coli* sedimentation was of minor importance.

Table 4.1: Summary of *E. coli* concentration per 100 ml at different depths and in all compartments of the Baffled Pond

Sampling depth (m)		compartment				Number of samples per depth
		A	B	C	D	
0.15	Avg*	**1.0E+03**	**8.8E+02**	**3.8E+02**	<1	20
	δ*	1.7E+03	1.7E+03	7.4E+02	NA	
0.60	Avg	**1.3E+03**	**1.1E+03**	**8.4E+02**	<1	20
	δ	1.8E+03	2.8E+03	1.4E+03	NA	
1.05	Avg	**2.2E+03**	**2.0E+03**	**1.1E+03**	**5.8E+02**	20
	δ	5.2E+03	3.7E+03	1.8E+03	7.2E+02	

Avg: Arithmetic average; **δ:** standard deviation; <1 per 100ml; **NA:** Not applicable

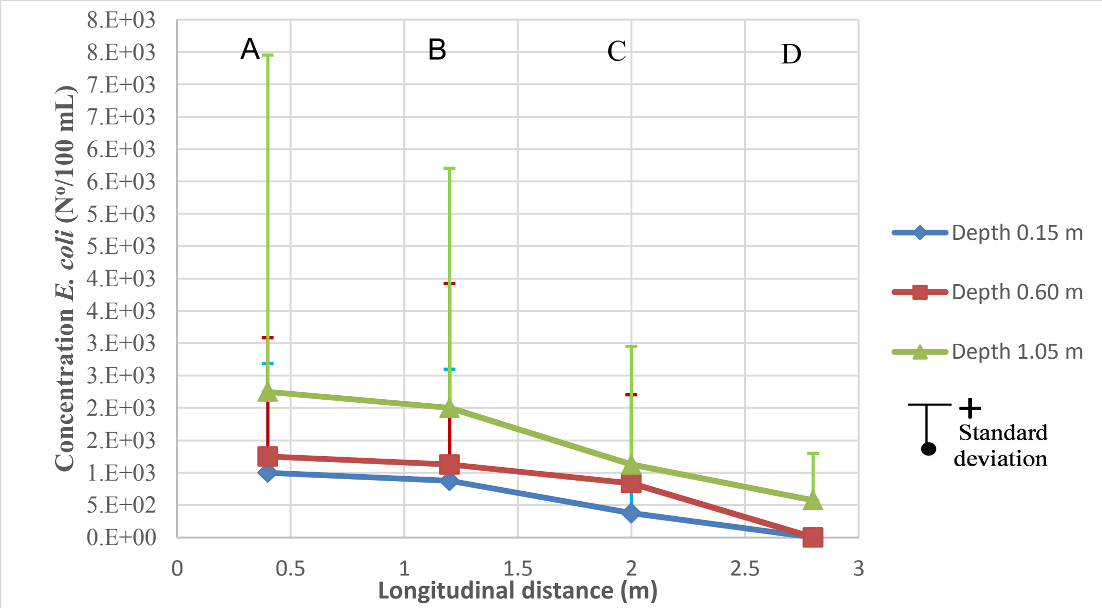

Figure 4.2: Arithmetic average distribution pattern of *E. coli* over depth and longitudinal distance in the four compartments of the Baffled Pond (20 samples for each depth)

A comparative study was conducted by monitoring the Baffled and unbaffled Control Ponds for six months working, under the same environmental conditions. **Table 4.2** presents a summary of the hydraulic characteristics, *E. coli* removal efficiencies and decay rates in both the BP and the CP. In 31 tests, the residual concentrations of *E. coli* remained higher than 1000 per 100 ml at the outlet of the CP. *E. coli* decay showed an average removal of around 4.5 log-units in the BP and 1.1 log-units in the CP. Lower values 2.2 and 2.3 log-units compare to the BP and higher values with respect to the CP were reported by Van der Steen et al., (2000). The statistical analyses revealed that the performance of the BP and the CP in terms of *E. coli* removal were significantly different (t-test, α=0.05). When comparing the hydraulic characteristics, it was shown that the actual HRT of the BP (4.1 days) was higher than that of the CP (3.2 days). This was believed to be due to the longer travel time created by the baffles in the pond. High dispersion numbers were determined (0.5 and 0.6 in BP and CP respectively). The dead volumes in the BP and the CP were found to be 38% and 51% respectively, while their effective volumetric efficiencies were 62% and 49% respectively (**Table 4.2**), as discussed earlier in **Chapter 3**. Therefore, the volume of the BP was used more efficiently for wastewater treatment than in the CP. According to both the Mixers-in - Series and the Dispersion Models, the BP and the CP behaved like Mixed-Reactors-in-Series (Levenspiel, 1999; Metcalf and Eddy, 2003). It can be concluded that the baffles with attached media, which increase the hydraulic retention time in the BP, may have played an important role in the removal of *E. coli*.

Other parameters that may have contributed to the elimination of *E. coli* are discussed later. The inactivation of *E. coli* in WSPs is due to very complex interactions of physical, chemical and biological processes (van der Steen et al., 2000). For instance, the photo-oxidation process was reported by Curtis et al., (1992) to be affected by the pH and the DO of the wastewater, which could also be the case in the present study, where simultaneous high pH and DO were achieved. The effects of factors influencing inactivation rate (*k*) of *E. coli* in dark and light conditions were reported by previous researchers. It appeared to be high (0.796 h^{-1}) in the light and low (0.045 h^{-1}) in the dark (Maiga et al., 2009). The *k* value of 1.1 day^{-1} found in the CP is in agreement with the findings of von Sperling (2002, 2003), von Sperling et al. (2003) and Van der Steen et al., (2000). The *k* in the BP was 9.1 day^{-1}, eight times the *k* value in the CP. This increase in *E. coli* decay efficiency may be explained by the presence of the baffles in the BP. Therefore, the shallower pond depth (1.1 m), combined with the hot climatic conditions in this country, are ideal conditions for *E. coli* decay (Maiga et al., 2009; Bolton et al., 2010; Konaté et al., 2010; Ouali et al., 2014).

Table 4.2: Summary of hydraulic characteristics, *E. coli* removal efficiencies and decay rates in the Baffled Pond and the Control Pond

Parameter	Influent wastewater	Baffled Pond	Control Pond	Number of samples
Flow (m^3 day^{-1})	0.5	0.5	0.5	31
Actual hydraulic retention time (days)	-	4.1	3.2	1
Dispersion number	-	0.5	0.6	1
Volumetric efficiency (%)	-	62	49	1
Dead volume (%)	-	38	51	1
Flow type	-	One mixed reactor	Two mixed reactors in series	1
E. coli concentration (N°/100 mL)	1E+5± 2E+5	< 1	4E+03±7E+03	31
E. coli log-unit removal (%)	-	4.5±0.8	1.1±0.7	31
k_d first-order *E. coli* decay coefficient (day^{-1})	-	9.1±3.2	1.1±1.2	31

- ***Chlorophyll-A Distribution Pattern***

Figure 4.3 shows the profile of Chlorophyll-A concentrations in the four compartments and at different depths. In the upper layers of all the compartments, above a depth of 0.60 m, higher values of Chlorophyll A (≥ 1000 µg/l) were observed and lower values (≤ 200 µg/l) were found deeper in the pond (**Figure 4.3**). The low concentration of Chlorophyll-A in the bottom layers may be due to the scarce amount of sunlight that reaches there (Maïga et al., 2009). The algal growth (with a high concentration of Chlorophyll-A) in the upper layers of the Baffled Pond has two opposing effects on *E. coli* (Van der Steen et al., 2000). In the first place, algal and other particles absorb solar radiation in the upper layers of the pond and, as a result, protect *E. coli* from destruction. Secondly, pH and DO will increase due to algal photosynthesis, therefore stimulating the photo-oxidation process that kills *E. coli* (Curtis et al., 1992). The algal distribution profile for this study was similar to that predicted by the 3D model of Sah et al., (2011), which was developed for a secondary facultative pond.

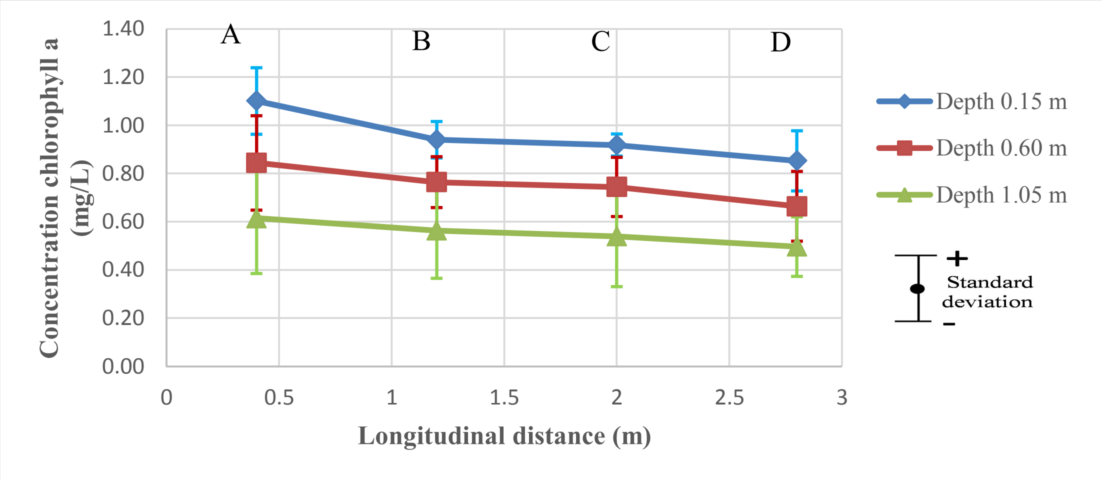

Figure 4.3: Arithmetic average distribution pattern of Chlorophyll-A over depth and longitudinal distance in the four compartments of the Baffled Pond (20 samples for each depth)

- ***COD distribution pattern***

Figure 4.4 shows the profile of dissolved COD concentrations in the four compartments and at different depths. One of the conditions for the survival of *E. coli* is the availability of carbon sources to sustain its cellular metabolism (Van der Steen et al., 2000). Therefore, the availability of carbon sources for the *E. coli* was measured via the dissolved COD concentration. The Baffled Pond had lower COD concentrations in the upper layers (a mean of 120 mg/L) and high values (250 mg/L) in the lower layers for all compartments, especially in the intermediate ones (B and C) (**Figure 4.4**). The results of this experiment indicate similar distribution patterns for *E. coli* and COD. This interdependency between *E. coli* and COD is in line with the findings of van der Steen et al. (2000), where very low *E. coli* decay is expected in the bottom layer, since enough carbon and nutrients are available.

On the other hand, the fact that higher concentrations of COD and *E. coli* were observed in the lower layers and higher algal values in the upper layers leads one to the conclusion that this polishing Baffled Pond was operating as a facultative pond. In addition, these results (high concentrations of COD and faecal coliform bacteria in the lower layers) are in agreement with those of El Halouani et al., (1993), Metcalf and Eddy (2003), Henze et al., (2008), and Buchanan et al., (2011), who argued that the die-off rate of coliform bacteria depends on the amount and type of organic matter in the wastewater and its temperature. If the water contains significant concentrations of organic matter and is at an elevated temperature, the bacteria may increase in number which was the case of this present study for the bottom layers. A similar phenomenon was been observed by Gerba (2008) in eutrophic tropical waters, which received organically enriched waters after heavy rainfall.

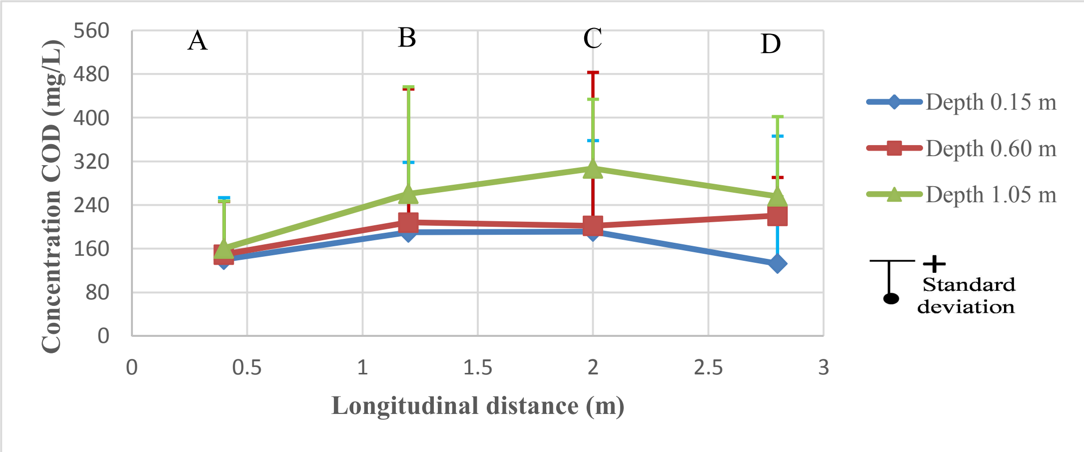

Figure 4.4: Arithmetic average distribution pattern of COD over depth and longitudinal distance in the four compartments of the Baffled Pond (20 samples for each depth)

4.3.2 pH, DO and Temperature variations and distributions in the BP

- ***pH, DO and Temperature diurnal variations***

Dissolved oxygen (DO), pH, and temperature were recorded at seven different depths of each compartment of the Baffled Pond and three times a day during the pumping hours (**Table 4.3**). Early in the mornings at 8:00 am, the average pH recorded throughout the experimental period was 8.7 in all compartments of the entire depth of the Baffled Pond. In the afternoons and evenings, at 1:00 pm and 5:00 pm, the pH had increased to 9.5 and 9.6 respectively, in the upper layers of all compartments up to a depth of 0.60 m, then decreased in the bottom layers of all compartments to 8.9, 8.8 respectively (**Table 4.3**). The results are consistent with the explanation by Bolton et al., (2010) that diurnal changes in pH occur mainly in WSPs due to algal photosynthesis.

Indeed, similar trends were also obtained with respect to DO. In the upper layers of all the compartments, up to a depth of 0.60 m, in the morning, the average DO concentration was 3.7 mg/L and then decreased to 2.9 mg/L in the lower layers of all compartments. However, there was an increase of DO in the afternoons and evenings and a decrease from top to bottom in all compartments: 9.4 to 4.8 mg/L and 12.5 to 4.4 mg/L respectively (**Table 4.3**). The main reasons of these high levels of DO in the Baffled Pond could be due to the photosynthesis of algae combined with alternate upward and downward flow induced by the baffles (Olukanni and Ducoste, 2011; Bolton et al., 2010).

Temperature, pH and DO showed similar trends. Lower values were recorded in the mornings and higher values in the afternoons and evenings from top to bottom in all compartments: 30.1 ◦C to 29.8 ◦C, 33.6 ◦C to 30.3 ◦C and 33.3 ◦C to 30.4 ◦C respectively (**Table 4.3**). This result is comparable to that observed by Abis and Mara (2006) and Ukpong (2013), whose work was also done in a hot climate.

From this study, it appears that pH, DO and temperature varies:

- during the day time at 8:00 am, 1:00 pm and 5:00 pm,
- from the upper layers to the lower layers and
- from one compartment to another.

These findings agree with those of Maïga et al., (2009), van der Steen et al., (2000), Sarikaya et al., (1987b), and von Sperling and Mascarenhas (2005), who reported that a shallow depth helped to create a predominantly aerobic environment and solar radiation, can easily reach the entire depth of the pond. Besides, the diurnal variations of pH, DO and temperature were in agreement with the seasonal fluctuation recorded during the experimental period and this is linked to the intensity of solar radiation (Kayombo et al., 2002).

Table 4.3: Summary of average values of pH, DO and temperature in the four compartments and at different depths of the Baffled Pond during the morning, noon and afternoon

Compartment	Variables	Time of sampling	Sampling depth (cm)						
			15[a]	30	45	60	75	90	105
			Avg± Std*						
A	pH	8:00am	**8.6** ± 0.5	**8.5** ± 0.7	**8.6** ± 0.7	**8.6** ± 0.7	**8.6** ± 0.7	**8.6**± 0.7	**8.6**± 0.7
		1:00pm	**9.3** ± 0.8	**9.4** ± 0.8	**9.3** ± 0.7	**9.0** ± 0.6	**8.9** ± 0.6	**8.8**± 0.6	**8.7**± 0.6
		5:00pm	**9.5** ± 0.7	**9.4** ± 1	**9.3** ± 0.9	**9.0** ± 0.8	**8.7** ± 0.8	**8.7**± 0.7	**8.6**± 0.7
	DO mg/L	8:00am	**4**± 1.1	**3.4**± 1.1	**3.2**± 1	**3**± 0.9	**2.8**± 0.7	**2.7**± 0.6	**2.6**± 0.5
		1:00pm	**8.9**± 1.5	**8.5**± 1.7	**7.6**± 1.7	**6.6**± 1.6	**5.9**± 1.4	**4.9** ± 1.3	**4.3**± 1.2
		5:00pm	**11.2**± 2.2	**11.4**± 2.2	**10.8**± 2.4	**8.2**± 3.3	**5.8**± 2.1	**4.6**± 2.1	**3.9**± 1.4
	T °C	8:00am	**30.2**± 2	**30.2**± 2	**29.9**± 2	**29.8**± 2	**29.8**± 2	**29.8**± 2	**29.8**± 2
		1:00pm	**33.5**± 2	**32.8**± 2	**31.6**± 2	**30.8**± 2	**30.4**± 2	**30.2**± 2	**30.2**± 2
		5:00pm	**33.4**± 2	**33.3**± 2	**32.6**± 2	**31.6**± 2	**30.9**± 2	**30.6**± 2	**30.3**± 2
B	pH	8:00am	**8.7**± 0.7	**8.7**± 0.7	**8.7**± 0.7	**8.7**± 0.7	**8.7**± 0.7	**8.7**± 0.7	**8.6**± 0.7
		1:00pm	**9.6**± 0.8	**9.5**± 0.9	**9.2**± 0.7	**9**± 0.7	**8.9**± 0.7	**8.8**± 0.7	**8.8**± 0.7
		5:00pm	**9.6**± 1	**9.6**± 1	**9.3**± 0.9	**9**± 0.8	**8.8**± 0.8	**8.7**± 0.8	**8.6**± 0.7
	DO mg/L	8:00am	**3.7**± 1.1	**3.4**± 1.1	**3.2**± 0.9	**3.1**± 0.8	**3**± 0.8	**3**± 0.7	**2.9**± 0.7
		1:00pm	**9.3**± 1.8	**9.1**± 1.9	**8.6**± 2	**7.9**± 1.9	**6.8**± 1.4	**5.9**± 1.2	**5**± 1
		5:00pm	**12.5** ± 2.2	**12.5**± 2.1	**11.7**± 2.2	**9.9**± 2.3	**7.3**± 1.8	**5.7**± 1.6	**4.8**± 1.6
	T °C	8:00am	**30.1**± 2	**30.1**± 2	**30.1**± 2	**30**± 2	**29.9**± 2	**29.9**± 2	**29.8**± 2
		1:00pm	**33.5**± 2	**33.1**± 2	**31.9**± 3	**30.9**± 3	**30.4**± 2	**30.4**± 2	**30.4**± 2
		5:00pm	**33.2**± 3	**33.2**± 2	**32.4**± 2	**31.4**± 2	**30.7**± 2	**30.5**± 2	**30.5**± 2
C	pH	8:00am	**8.8**± 0.7	**8.7**± 0.8	**8.7**± 0.8	**8.7**± 0.8	**8.7**± 0.8	**8.7**± 0.8	**8.7**± 0.8
		1:00pm	**9.6**± 0.9	**9.6**± 0.9	**9.3**± 0.8	**9.1**± 0.7	**8.9**± 0.7	**8.8**± 0.7	**8.8**± 0.7
		5:00pm	**9.7**± 1	**9.6**± 1	**9.3** ± 0.9	**9**± 0.8	**8.8**± 0.8	**8.7**± 0.8	**8.7**± 0.7
	DO mg/L	8:00am	**3.6** ± 1.3	**3.4**± 1.3	**3.3**± 1.3	**3.2**± 1.1	**3**± 0.9	**3**± 0.8	**3**± 0.8
		1:00pm	**9.3**± 1.8	**8.8**± 1.8	**8.6**± 2.2	**8.2**± 2.3	**7.2**± 2.4	**5.7**± 1.7	**4.9**± 1.7
		5:00pm	**12.7**± 2.4	**12.7**± 2.6	**10.1**± 2.4	**10.3**± 2.2	**7.9**± 2.3	**5.7**± 1.9	**4.6**± 1.3
	T °C	8:00am	**30.1**± 2	**30.1**± 2	**30.1**± 2	**30**± 2	**29.9**± 2	**29.9**± 2	**29.8**± 2
		1:00pm	**33.6**± 2	**33.3**± 2	**32**± 3	**31.2**± 3	**30.7**± 2	**30.5**± 2	**30.3**± 2
		5:00pm	**33.3**± 3	**33.3**± 2	**33.5**± 2	**31.5**± 2	**30.9**± 2	**30.5**± 2	**30.4**± 2
D	pH	8:00am	**8.8**± 0.6	**8.7**± 0.8	**8.7**± 0.7	**8.7** ± 0.7	**8.7**± 0.7	**8.7**± 0.7	**8.7**± 0.7
		1:00pm	**9.6**± 1	**9.6**± 0.9	**9.3**± 0.7	**9.1**± 0.7	**8.9**± 0.7	**8.8**± 0.7	**8.7**± 0.7
		5:00pm	**9.7**± 1	**9.6**± 1	**9.3**± 0.9	**8.9**± 0.8	**8.7**± 0.8	**8.7**± 0.8	**8.6**± 0.7
	DO mg/L	8:00am	**3.5**± 1.3	**3.2**± 1.2	**3.1**± 1.2	**3**± 0.9	**2.8**± 0.8	**2.7**± 0.7	**2.7**± 0.7
		1:00pm	**10**± 2.5	**9.6**± 2.3	**8.7**± 2.1	**7.2**± 2.2	**6.4**± 1.9	**5.6**± 1.6	**4.9**± 1.3
		5:00pm	**13.4**± 2.2	**13.2**± 2.5	**11.7**± 2.6	**9**± 2.5	**6.7**± 2.4	**5**± 2	**4.1**± 1.4
	T °C	8:00am	**30** ± 2	**30**± 2	**30**± 2	**30**± 2	**29.9**± 2	**29.9**± 2	**29.8**± 2
		1:00pm	**33.7**± 3	**33.3**± 2	**32.1**± 3	**31.1**± 2	**30.7**± 2	**30.5**± 2	**30.4**± 2
		5:00pm	**33.4**± 3	**33.3**± 3	**32.5**± 3	**31.4**± 2	**30.9**± 2	**30.6**± 2	**30.5**± 2

[a]pH, DO and Temperature were recorded 450 times at each depth; ***Avg**: Arithmetic average; **Std:** standard deviation

- ***pH, DO and Temperature profiles***

Figure 4.5 depicts pH, dissolved oxygen (DO) and temperature profiles. The pH recorded in the upper layers of Compartments A, B, C and D respectively were: 9.2, 9.3, 9.3 and 9.4, then decreased progressively to approximately 8.7 in the lower layers of all compartments (**Figure 4.5a**). These high pH values achieved in this experimental study are in agreement of those reported by Curtis et al., (1992) and Kayombo et al., (2002). Moreover, similar profile of pH was predicted by the 3D model of Sah et al., (2011), which was developed for a secondary facultative pond. It is also reported by Bolton et al., (2010) that pH could result in a decreased stability of the cells of micro-organism in the ponds (**Figure 4.2**).

Furthermore, similar profiles were also achieved for the DO. Higher values of DO were recorded in the top layers (8.1, 8.5, 8.5 and 8.8 mg/L in Compartments A, B, C and D respectively) and decreased to lower values in the lower layers (respectively to 3.6, 4.2, 4.2 and 3.4 mg/l; **Figure 4.5b**). The DO profile for this study was similar to that predicted by the 3D model of Sah et al., (2011), which was developed for a secondary facultative pond. Bolton et al., (2010), comment that DO stratification can vary significantly through the water column, with nearly all effective light being absorbed in upper layers. It is also assumed that an increase in DO would result in an increase in highly reactive oxygen species formation and therefore increase the photo-oxidation. This fact could be the reason why low *E. coli* concentrations were found in this Baffled Pond (**Figure 4.2**). The results of this study are also in agreement with those of Davies-Colley et al., (1999) who reported that the endogenous photo-inactivation of *E. coli* was strongly dependent on DO.

Temperature, pH and DO showed similar profiles. The temperature falls from 32, 33, 32 and 31◦C in the top layer (up to a depth of 15cm) to 30, 30, 30 and 28 ◦C at a depth of 75 cm in Compartments A, B, C and D respectively. However, below there, the temperature remains constant (**Figure 4.5c**). The model developed by Ukpong (2013), based on published data from ponds operated in similar climatic conditions to predict the vertical temperature profile in waste stabilization ponds shows a comparable trend (**Figure 4.5c**). The reasons for this phenomenon (thermal stratification in ponds) are well documented in the literature (Abis & Mara, 2006; Ukpong, 2013). Certainly, the higher relative temperature difference from the surface layer of the pond to the bottom layer during thermal stratification processes could be related to the complex mechanisms of thermal energy transfer between the water surface and the air.

The profiles for pH, DO and temperature in those three cases were similar to those of the Baffled Pond (**Figure 4.5).** In the upper layers of all the compartments (down to a depth of 0.60 m), higher values of pH (≥9), DO (≥9 mg/L) and temperature (≥32°C) were observed, whereas lower values were found in the lower layers (**Figure 4.5**). The results are consistent with those of Ouali et al., (2014) and Curtis et al., (1992) who found that the die-off rates of *E. coli* were dependent on pH, DO and temperature, with more rapid disinfection at higher values of these parameters

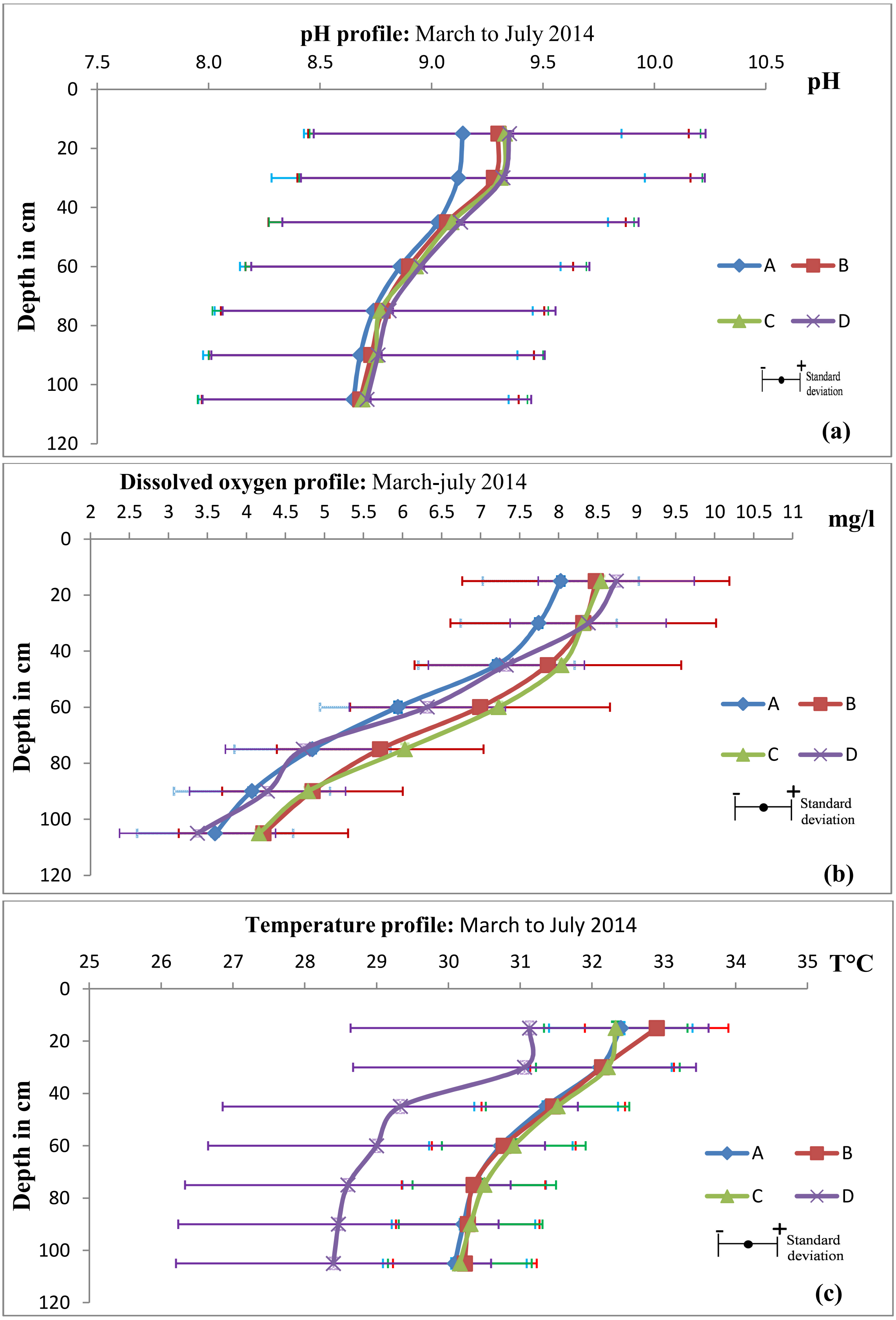

Figure 4.5: Arithmetic daily (8:00am, 1:00pm and 5:00pm) averageprofiles of: (**a**) pH, (**b**) DO and (**c**) Temperature in the four compartments of the Baffled Pond

(DO, pH and temperature were recorded 450 times at each depth)

4.4 Conclusions

The investigation carried out in the Baffled Pond to understand the non-detection (< 1 per 100 ml) of *E. coli* in its effluent has confirmed the results of the previous study of the performance evaluation of the two-stage high rate anaerobic reactor followed by a Baffled Pond with attached-growth for domestic wastewater treatment in a sub-Saharan African, warm-dry climate. The results obtained from this study have revealed the benefits of releasing the effluent from the top layer pond with respect to *E. coli* inactivation. Moreover, it was found that there was a significant difference in the removal efficiencies and die-off rates for *E. coli* between the BP and the CP. This implied that the baffles with attached media, which increased the hydraulic retention time in the BP compared to CP, may have played an important role in the removal of *E. coli.* In addition, the effluent from the baffled pond was released from the top layer, where there were consistently fewer *E. coli*. Sedimentation, combined with the synergetic effects of the physical, chemical, and environmental factors may be responsible for the inactivation of *E. coli* in this system. It is advisable to investigate the importance of predation by zooplankton in the elimination of *E. coli* from this sort of Baffled Pond. Based on the outcome of this research, it was concluded that treatment via anaerobic reactors followed by Baffled Ponds could be applied widely as a low-cost alternative to treat the wastewater low-income city-dwellers, and it would be recommended for the effluent to be used for restricted irrigation of peri-urban agriculture.

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Chapter 5

5. Biofilm characteristics and zooplankton composition in a Baffled Pond in the Sahel Region of Africa

This chapter focuses on the characteristics of the biofilm that formed in the Baffled Pond, in terms of algal biomass distribution, microbial diversity, and zooplankton species composition and distribution in the water column and on the attached media. The potential correlations among phytoplankton, zooplankton, bacteria, suspended solids, and several environmental variables are also discussed.

5.1 Importance of biofilm biomass and zooplankton development in WSP

Biofilm has been defined as ''a layer of microorganisms in an aquatic environment held together in a polymeric matrix attached to a substratum'', and these adherent microorganisms are frequently embedded within a self-produced matrix of an Extracellular Polymeric Substance (EPS) (van der Wende and Characklis**,** 1990). In the natural aquatic environment, bacteria may be planktonic (i.e., suspended in the water) or sessile (i.e., fixed to a surface). Biofilms may contain many different types of microorganisms, including bacteria, archaea, protozoa, fungi, and algae, with each group performing specialized metabolic functions (Donlan, 2002). Bacteria living in a biofilm also usually have significantly different properties from free-floating bacteria of the same species, as the dense and protected environment of the film allows them to cooperate and interact in various ways.

Although biofilm formation is sometimes considered detrimental in certain instances, it can also be harnessed for beneficial purposes (Brading et al., 1995). For instance, Bitton, (2005) pointed out an increased interest in biofilm microbiology within the field of wastewater treatment, which is leading to the development of specialized biological wastewater treatment systems. The role played by the microorganisms in wastewater treatment plant depends on the process being used and biofilms are very efficient at degrading soluble organic matter, transforming and removing nitrogen, and filtering out suspended solids and other microorganisms (von Sperling and Chernicharo, 2005). Biofilms also help to remove of phosphorus by simply creating the favourable conditions for microorganisms to carry out this process (Henze et al., 2008). Indeed, biological wastewater treatment occurs through the role of microorganisms, since domestic wastewater consists of roughly 70-80% organic matter, most of which can be consumed by these organisms (Bitton, 2005).

A mutualistic relationship between bacteria and algae also enhances the treatment that takes place in the Waste Stabilization ponds (WSP) (Kayombo et al., 2002). In addition, WSP have been shown to contain a complex ecosystem, consisting of algae, virus, protozoa, rotifers, insects, crustaceans, and fungi (von Sperling and Chernicharo, 2005). These microbial communities stabilize the organic waste and lower the levels of pathogens in the effluent. Last but not least, biofilms are known to be important components of food chains in rivers and streams and are grazed by the aquatic invertebrates that many fish later eat

(Oueda, 2009). In addition, many fish, such as the family Loricaridae, feed on the biofilm directly (Bitton, 2005).

In any case, biofilm formation is difficult to control and the attachment of microorganisms to surfaces is a very complex process, with many variables affecting the outcome (Brading et al., 1995). The principal factors include oxygen, pH, temperature, nutrients, cations, type of substrate, extracellular polymeric substance (EPS) and flow velocity (Characklis et al., 1990; Esterl et al., 2003). Characklis et al. (1990) noted that the extent of microbial colonization increases as the surface roughness increases, and this was mainly due to lower shear forces on rougher surfaces. It has also been shown that bacteria preferentially attach to a variety of surfaces (Pringle and Fletcher, 1983). Surprisingly, microorganisms attach more rapidly to hydrophobic, non-polar surfaces (e.g., Teflon and other plastics) than to hydrophilic materials (e.g., glass or metal; Fletcher and Loeb, 1979; Pringle and Fletcher, 1983; Bendinger et al., 1993; Donlan, 2002). In this context, provision of artificial plastic media may be a good option for encouraging the development of microbial biomass in wastewater treatment plants and hence enhance their performance.

In this regard, various strategies have been developed for effective control of biofilm for constructive purposes. For instance, Babu, (2011) investigated the effect of baffles on the algal-bacterial biofilm structure and the composition of zooplankton in WSP for nitrogen removal, and found that baffles improved water quality, which in turn affected the ecology of the treatment plant, and thus its performance.

Another subject of importance is the effect of zooplankton on wastewater treatment (Mitchell and Williams, 1982). There are three major classes of crustacean zooplankton in these ponds: copepods, cladocera and rotifers (Babu, 2011). These have different sizes and feeding habits, although overlap does exist (Oueda, 2009). Zooplankton have been reported to be more abundant in ponds with low organic loading (Uhlman, 1980), and they also release nutrients back into the water (Lehman, 1980; Lampert et al., 1986; Lai and Lam, 1997). Babu, (2011) found that the presence of zooplankton in ponds did not contribute much to nitrogen removal. Therefore, the effect of zooplankton on nutrient removal in ponds is still not well understood.

Although researchers are becoming increasingly aware of the importance of roles that biofilm play in wastewater treatment processes, little research has been done in Sahelian climatic conditions. Hence, a greater understanding of biofilm processes, including their distribution, dynamism, and diversity, in both sessile and planktonic placement in WSP, is key to providing good treatment and optimal system performance. Therefore, in this study, baffles with additional rough surfaces as attachment media support (plastic) were incorporated in a pilot-scale wastewater pond and investigated under Sahelian climatic conditions for algal biomass distribution, microbial diversity, and zooplankton species composition and distribution in the water column and on the attached media of the pond.

5.2 Methodology

5.2.1 Pilot plant description

The Baffled Pond with attached-growth used in this study, and its operational conditions, were as described in **Chapter 2** (**Figure 2.4**). In order to better understand the non-detection of *E. coli* in the effluent of this Baffled Pond, algal biomass distribution, microbial diversity, and zooplankton species composition and distribution in the bulk water and attached media were studied

5.2.2 Sampling and analysis

After two years of operation of the Baffled Pond, grab samples of 500 ml were taken between 8:00 to 9:00 am, at three different depths (15, 60 and 90 cm) in the water column and in each compartment (A, B, C and D). Next, the pond was carefully emptied and the entire biofilm was carefully scraped off within sample plots of 10 cm by 10 cm (0.01 m^2), at depths of 15 cm, 60 cm and 90 cm on: one transversal wall, one longitudinal wall and the two faces (Face A and Face B) of the three vertical baffles with affixed waste plastic bottle caps. The collected samples were appropriately stored and analysed for: dry weight of biofilm, Total Suspended Solids (TSS), Algal biomass (Chlorophyll-A), zooplankton and microbial diversity. The sampling positions were as described in **Chapter 4** (**Figure 4.1**).

- **Dry weight**

The biofilm samples were dried at 105 ^{o}C for 1.5 h to determine the dry weight as TSS (APHA, 2012).

- **TSS**

The TSS was determined by filtration of the water column samples through pre-weighed 0.45 µm Whatman Glass Microfiber Filter (GFC) as described in APHA (2012).

- **Algal biomass**

Chlorophyll-A, as an indicator of algal biomass, was measured using the spectrophotometric method described in APHA (2012). It consisted of a sequential procedure of filtration, centrifugation and extraction of the chlorophyll by using an organic solvent (90 % acetone). The extracted chlorophyll was then analysed by a spectrophotometer Hach Lange DR 5000 at the light absorption wavelengths of 750 nm and 665 nm, to be expressed in units of mg/L.

- **Zooplankton**

For the study of zooplankton, two types of assessment were carried out: qualitative and quantitative. The first established species richness and diversity, while the second highlighted their distributional densities (Oueda et al., 2007). Water samples were stored in bottles of 250 ml, preserved with 2% formaldehyde. However, for the scraped biofilm samples, 250 ml of distilled water were added to dilute the samples and were also preserved with 2% formaldehyde. The samples were transferred to the Laboratory of Biology and Animal Ecology (LBEA) of the University of Ouagadougou (Burkina Faso) for analysis of zooplankton. In the laboratory, the samples were washed over a 100 µm sieve (plankton net) to remove the fixative. Sub-samples of 0.5 ml were taken from well agitated samples to ensure a homogeneous distribution of organisms. The sub-samples were put on a counting chamber and examined under a microscope at both X100 magnification for taxonomic analysis and X40 magnification for counts to determine species composition and abundance, respectively. In order to optimize the quality of the results, the identification and counting procedures were repeated four times for each sample. Identifications were conducted to the lowest possible taxon using published keys and figures (Sars, 1895; Koste and Voigt, 1978; Pontin, 1978; Dussart, 1980; Pourriot, 1980; Rey and Saint-Jean, 1980; Notenboom-Ram, 1981; Amoros, 1984; Kořinek, 1999; Hamaidi et al., 2008; Oueda, 2009).

- **Microbial community**

The spread plate method was used to determine the abundance of microbial colonies in the water and biofilm samples, cultured on Blood agar, nutrient agar, MacConkey agar and Sabouraud agar to evaluate different types of microbes (**Table 5.1;** APHA, 2012).

Table 5.1: Summary of the methods for the four microbial colonies

Types of Microbes	Incubation temperature (^oC)	Incubation duration	Culture medium
Fastidious bacteria	37	24 to 48 hours	Blood agar
Non-fastidious bacteria	37	24 hours	Nutrient agar
Enterobacteria	37	24 to 48 hours	MacConkey agar
Fungi: Yeasts and Molds	26	5 to 7 days	Sabouraud Agar

- **Physical and chemical parameters**

The pH, water temperature (T) and dissolved oxygen (DO) were measured *in situ* with an integrated portable pH-T-DO meter (WTW 340i/SET) fixed to a cylindrical iron rod carefully graduated (**Section 4.2.2**). This device was previously calibrated using standard buffer solutions before any usage. The measurements were carried out between 8:00 am to 9:00 a.m., at depths of 15 cm, 60 cm and 90 cm, in each compartment (A, B, C and D) of the Baffled Pond, before it was drained.

- **Statistical analysis**

Two multivariate statistical technique, called Principal Components Analysis (PCA) and Canonical Correspondence Analysis (CCA), were applied using the XLSTAT 7.5.2 and the Paleontological Statistics Software Package for Education and Data Analysis (PAST 3.04), in order to appreciate the interactions between biotic and abiotic components of the Baffled Pond. The data were normalized before doing this CCA by dividing each variable by its own greatest value, such that all the data only ranged up to one.

- **Sludge accumulation in the Baffled Pond**

The 'White Towel Technique' was used to assess the sludge that had accumulated in the Baffled Pond after two years of continuous operation. The method consisted of wrapping a white towel around the bottom of a cylindrical iron rod attached with a graduated tape. The depth of the sludge was measured by lowering the rod vertically into the pond until it reaches the bottom; it is then slowly withdrawn to read the depth of sludge left on the towelling material (Llyod & Vorkas, 1999; Mara, 2004; Konate et al., 2013). The Baffled Pond was divided into bathymetric sections of 7 cm^2 and a total of 40 locations were measured for sludge thicknesses.

5.3 Results and Discussion

The results of this study are presented mainly via bar diagrams, in order to easily compare the biofilm distribution on both the attached media and in the bulk water. The density and types of zooplankton are also shown in tabular form. The wastewater arrives at the Baffled Pond with one set of organisms and leaves with an almost entirely different one. In particular, *E. coli*, an indicator of faecal bacteria, is abundant in the influent wastewater, but is reduced to undetectable levels at least 15 cm before the water leaves the pond.

5.3.1 Biomass on attached media and in the water column

- **Biofilm biomass on attached media**

There was great variation in the dry weight densities of biofilm on plastic baffles, plastic bottle caps and cement walls, after 2 years of operation of the Baffled Pond (**Figure 5.1)**. Thick, dense biofilm occurred in the upper sections, above a depth of 60 cm, on all of the attached media, and then decreased toward the bottom, except for the case of the plastic bottle caps. The densest biofilms, of up to 370 g/m^2 were recorded at a depth of 15 cm, on the baffle plates of Face B (counter current), on both Baffles 2 and 3. This is consistent with the findings of Babu (2011), in which the variation in dry weights could have been caused by biofilm components including algae, heterotrophs bacteria, midge larvae and detritus, as well as depending on the aerobic and anaerobic conditions. This variation could also be due to the preferential behaviour of microorganisms to attach to hydrophobic, non-polar surfaces, such as plastic (Pringle and Fletcher, 1983). Another reason for thick, dense biofilms to form on the upper structures may be the high content of algae in the biofilm (**Figure 5.3**). On the other hand, the thick, dense biofilms in the lower plastic caps could be related to the greater contribution of detritus, since their cylindrical holes retain such sediments. The introduction of baffles with affixed bottles caps has allowed producing 1.5 kg of dry weight biofilm in the Baffled pond of 15.79 m^2 attached surface area.

Biofilms were thicker on the counter-current faces (**Figure 5.1)**, possibly due to their greater protection against the shear forces induced by the water flow, which could otherwise make the biofilm slough off of the substrate (van der Wende and Characklis**,** 1990).

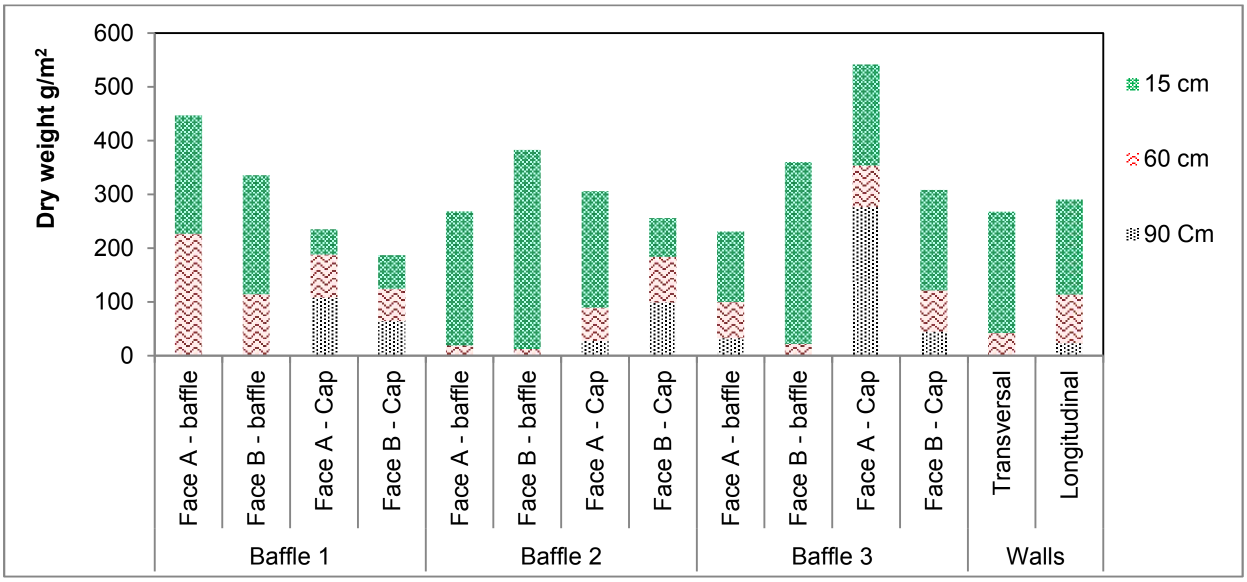

Figure 5.1: Biofilm dry weight densities on baffles, bottle caps and walls at depths of 15 cm, 60 cm and 90 cm after 2 years of operation of in the Baffled Pond

- **TSS in the water column**

There was also great variation in the amount of suspended solids at different depths, in the four compartments of the Baffled Pond (**Figure 5.2)**. High concentrations were observed in the top layers, at a depth of at 15 cm, in all compartments, with a maximum of 35 mg/l, which gradually decreased toward the bottom. These high levels of TSS in the top layers of the water column may be due to the abundance of algae (**Figure 5.3**) and other planktonic microorganisms, since algal photosynthesis mainly occurred in the top layers, where sunlight and carbon dioxide are available (Maiga et al., 2009). Nonetheless, these TSS concentrations were lower than those reported in the literature (Oliveira and von Sperling, 2011). The results are consistent with Shin and Polprasert (1987), who explain that attached biofilms could adsorb some of the dispersed planktonic microorganisms and other particles, thus reducing TSS concentrations. A total of 0.04 kg of TSS was found in the water column of the Baffled Pond of 3.2 m^3 wastewater volume.

When comparing the TSS in the water column with the attached biofilm, it was evident that more biomass was found on the substrates (1.5 kg of attached biomass) than was dispersed in the water (0.04 kg of biomass in the water column) (**Figures 5.1, 5.2**). The biomass attached on the media constituted 35.5 times of that in the water column. Therefore, the baffles enhanced the growth of attached biomass, increasing the biomass volume in the pond and consequently leading to better removal of organics and nutrients (Shin and Polprasert, 1987).

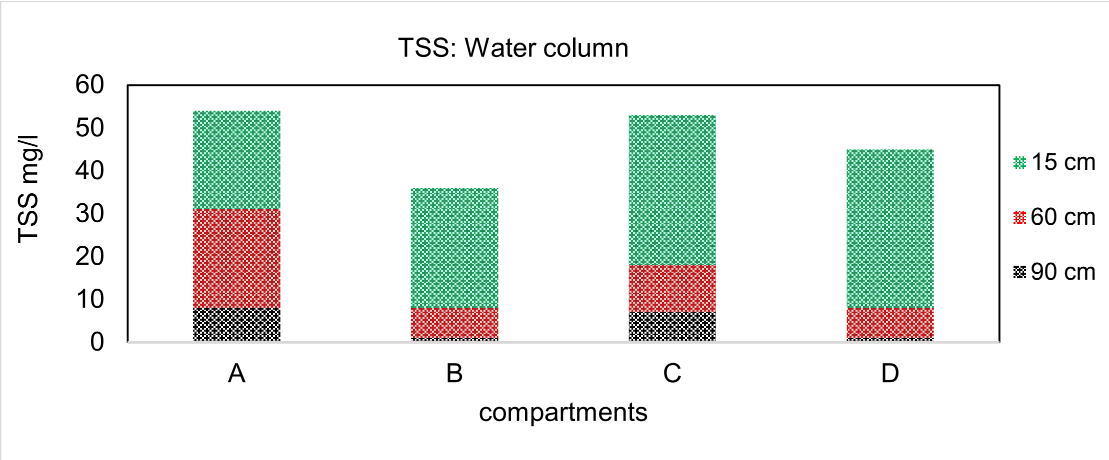

Figure 5.2: TSS distribution pattern at the depths of 15 cm, 60 cm and 90 cm, in the four compartments of the Baffled Pond

- **Attached and dispersed algal biomass**

Chlorophyll-A, which is an indicator of algal biomass was most concentrated on Baffle 2 (**Figure 5.3).** It was also fairly abundant on the entire length of each baffle. This implied that more algal activity occurred in the central region of the pond. The abundance of algae at the bottom of the baffles was consistent with the findings of Barranguet et al. (2005), who reported that when algal-bacterial biofilm develop under low light, the proportion of heterotrophic bacteria to algae increases.

On the other hand, the algal biomass in the water column, increased toward the outlet, reaching a maximum concentration of 12 mg/l in the upper layers down to a depth of 60 cm. Concentrations were considerably lower in the deeper layers, with values as low as 1 mg/l (**Figure 5.3b)**. The low concentration of Chlorophyll-A in the lower layers may be due to the fact that sunlight did not reach so deep (Maiga et al., 2009).

By comparing the results between attached and dispersed algal biomass (**Figure 5.3**), it becomes evident that the attached algal biomass is more important than that in the water column. Shin and Polprasert (1987) and Babu (2011) also found that the addition of artificial media in ponds increased the growth of algae and thus enhanced the removal of organics and nutrients. Moreover, these results are in line with the findings of Characklis et al., (1990), who explained that microbial colonization appears to increase as the surface roughness increases. As a result, the introduction of the baffles with roughened surfaces in the polishing pond seems to be an excellent way to encourage the growth of an ample biofilm in such ponds.

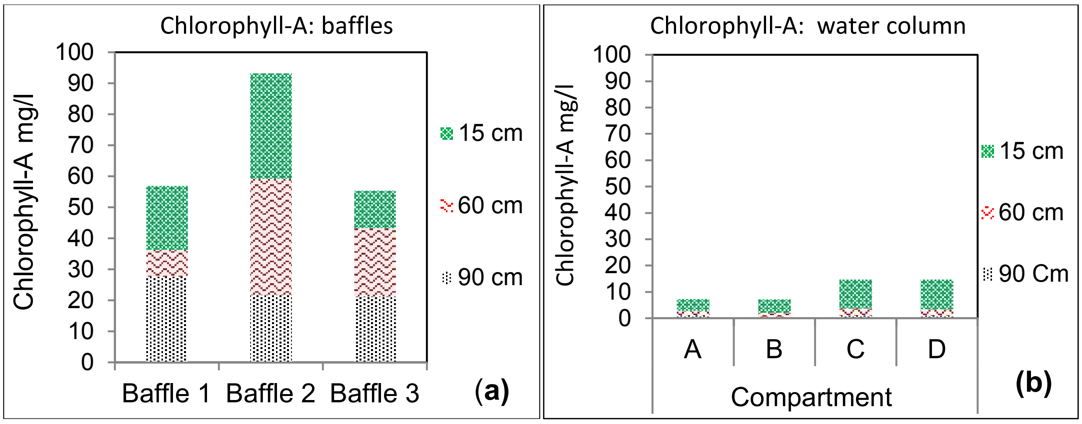

Figure 5.3: Chlorophyll-A distribution pattern at depths of 15 cm, 60 cm and 90 cm **(a)** on baffles and **(b)** in the water column of the four compartments of the Baffled Pond

- **Distribution bacterial colonies in the water column**

There was considerable variation in the distributions of fastidious bacteria, non-fastidious bacteria and enterobacteria throughout the pond (**Figure 5.4)**. Concentrations diminished with depth and also varied from one compartment to another, with the only exception of fastidious bacteria in Compartment A. This means that fastidious bacteria which include presumably the most pathogenic bacteria are getting wiped out quickly in the pond. These distributions are similar to those of TSS (**Figure 5.2**) in the water column. Furthermore, there was a strong linear correlation between non-fastidious bacteria and TSS a (correlation coefficient R^2= 0.979), and this can also be seen in the graph of the Canonical Correspondence Analysis (**Figure 5.9**). This phenomenon appears to be related firstly, to the adsorption and/or the attachment of bacteria to suspended solids and, secondly, to the symbiotic algal-bacterial activity in the course of organic matter degradation in the upper layers, due to the fast adaption of the non-fastidious bacteria.

The high concentration of the fastidious bacteria in the top layer (15 cm) of the first compartment (A) could be explained by their complex nutritional requirements, since they only grow when specific nutrients are included in their diet. Therefore, the inlet point seems to contain the specific nutrients necessary for their growth (**Figure 5.4**).

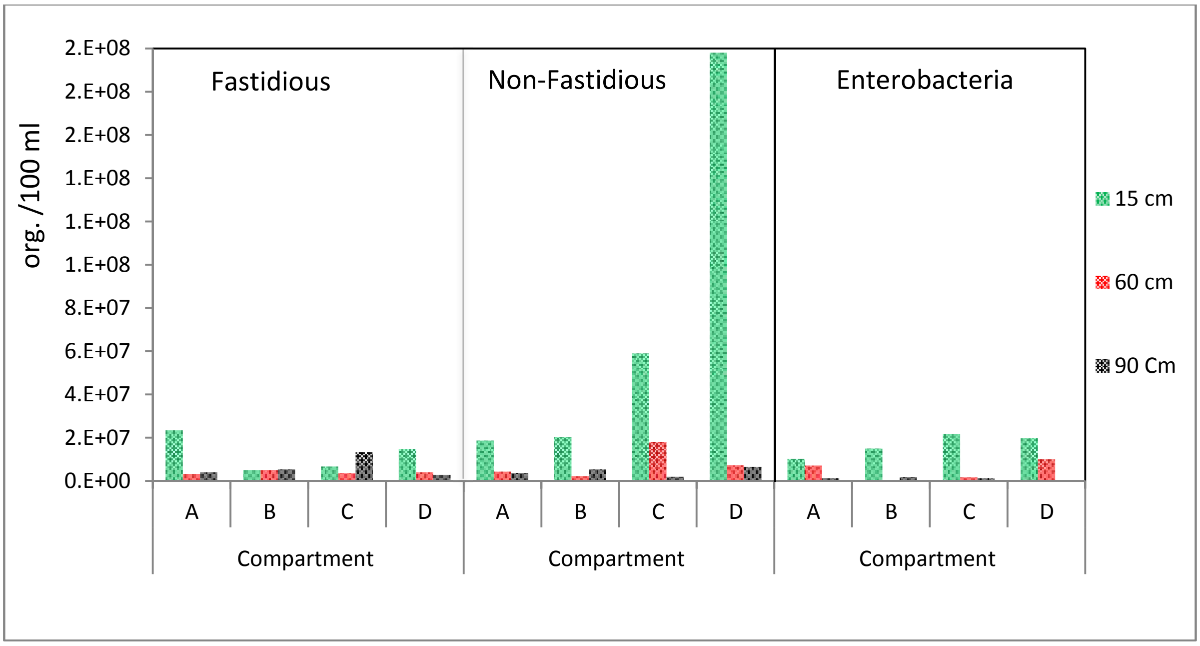

Figure 5.4: Distributions of three categories of bacteria at depths of 15 cm, 60 cm and 90 cm in the water columns of the four compartments of the Baffled Pond

- **Distribution of fungi on baffles and in the water columns**

The concentrations of yeasts and moulds on the structures inside the pond increased with depth and toward the outlet (**Figure 5.5**). On the other hand, these fungi were almost absent in the water columns throughout the pond, with only very low concentrations being observed in the bottom layers of Compartments A and D. This indicates that fungi prefer to grow on hard surfaces rather than being dispersed in the water. Moreover, there are reports indicating that the presence of fungi has two contradictory advantages in the treatment of wastewater. In the first place, fungi have the capacity to survive in acidic environments and with little nitrogen content, which makes them important for improving biological wastewater treatment processes, whereas, in the second place, certain filamentous fungi can deteriorate the sludge settleability (bulking the sludge), therefore reducing the efficiency of the treatment process (Metcalf and Eddy, 1991; von Sperling and Chernicharo, 2005, Henze et al., 2008). It is important to point out that this latter adverse effect was not the case in this study, since the fungi were attached to the baffles and may have contributed to the improved treatment performance that was achieved. In addition, unlike algae, fungi can assimilate dissolved carbon dioxide in the water phase, and they are also known to contribute to the removal of organic waste contained in sewage (Brading et al., 1995). For these reasons, the introduction of baffles with roughened surfaces may have increased the growth of fungi in the pond, which may have played an important role in improving the treatment performance of the system.

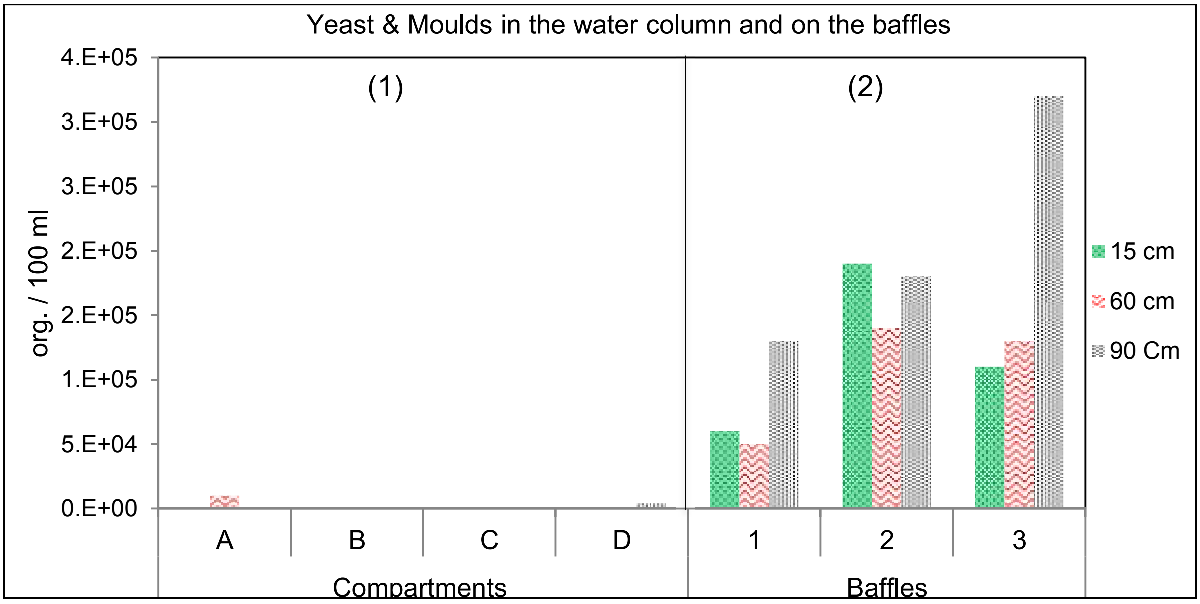

Figure 5.5: Distribution of fungi at depths of 15 cm, 60 cm and 90 cm **(1)** in the water columns of the four compartments and **(2)** on the baffles of the Baffled Pond.

5.3.2 Zooplankton in the Baffled Pond

A total of 19 zooplankton taxa were identified (**Table 5.2)**. These planktonic organisms belong to 5 families of crustaceans (*Daphnidae, Moinidae, Sididae, Cyclopidae, Diaptomidae)* and 4 families of rotifers (*Brachionidaes, Testudinellidae, Asplanchnidae, Lecanidae).* The rotifers, copepods, cladocerans and ostracods were the most abundant members of this aquatic fauna (**Figure 5.6**). It is important to point out that copepod larvae cannot be identified to species; therefore they were counted separately from adults. In terms of biodiversity and species richness, rotifers were dominant, with 14 species identified and at least one species in most of the samples, followed by the cladocerans (with 4 species) and finally the copepods (with 4 species and a high density of *Nauplii and Copepodites;* **Table 5.2, Figure 5.6**). These results are similar to those reported by Oueda et al., (2007) who found, in their study of the diversity, abundance and seasonal dynamics of zooplankton community in a south Saharan reservoir of Burkina Faso, many abundant species richness of zooplankton, despite various pressures on these water reservoirs.

The dominance of rotifers may be explained by the fact that they are very adaptable, even to such a polluted environment (Hamaidi et al., 2008). Rotifers are microscopic organisms common in fresh and brackish waters and are quantitatively dominant in zooplankton communities in lakes and calm parts of rivers, partially due to their parthenogenetic reproduction and rapid maturation. Many species of this group (e.g., *Brachionus keratella)* are used in aquaculture farms for feeding juvenile fish (Oueda, 2009). Furthermore, rotifers are known to be very efficient at consuming dispersed bacteria and small particles of organic matter. In addition, according to Metcalf and Eddy (1991) their presence in the effluent of a treatment system indicates an efficient biological purification process.

Table 5.2: Summary of the identified zooplankton species in the Baffled Pond

Class	Sub-class	Family	Species
Crustaceans	**Branchiopodes**	Daphnidae	*Ceriodaphnia cornuta Sars, 1886*
		Sididae	*Diaphanosoma excisum Sars, 1885*
		Moinidae	*Moinodaphnia macleayi King, 1853*
			Moina micrura Kurz, 1874
	Copepods	Diaptomidae	*Tropodiaptomus incognitus Dussart Gras1966*
		Cyclopidae	*Mesocyclops leuckarti Claus 1857*
			Nauplii spp
			Copepodites spp
Rotifers		Brachionidae	*Brachionus falcatus (Ehrb., 1838)*
			Brachionus caudatus (Barrois Daday, 1894)
			Brachionus angularis (Gosse, 1851)
			Brachionus quadridentatus (Hermann, 1783)
			Brachionus calyciflorus (Pallas, 1766)
			Epiphanes spp
			Keratella cochlearis (Gosse, 1851)
			Keratella tropica (Apstein, 1907)
		Asplanchnidae	*Asplanchna brightwelli (Gosse, 1850)*
			Asplanchnopus multiceps (Schrank, 1783)
		Lecanidae	*Lecane luna (Müller1776)*
		Testudinellidae	*Filinia longiseta (Ehrb., 1834)*
			Filinia terminalis (Plate, 1886)
			Hexarthra spp
ostracods			

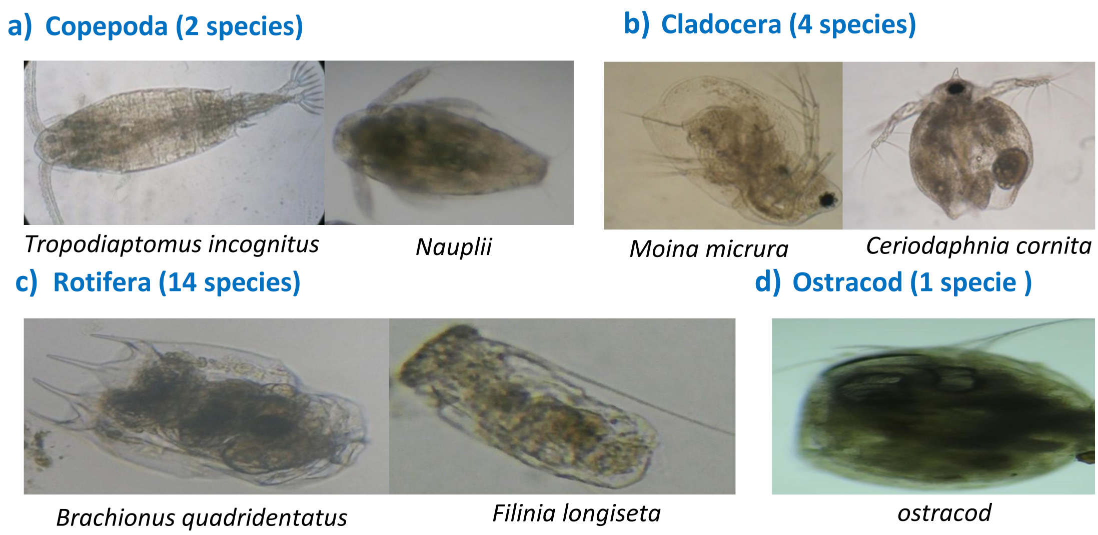

Figure 5.6: Photographs of some of the species of the four main groups of zooplankton found in the Baffled Pond

Copepodite, *Nauplii* and were the most constant and abundant zooplankton genera in the Copepoda group, being found in all depths of all compartments (**Table 5.3)**. Likewise, the *Diaphanosoma excisum* and *Moinodaphnia maleayi* of the Cladocera group were also ubiquitous. Their abundant and constant presence could be due their documented preference to feed on a wide variety of phytoplankton, from unicellular picoplankton (e.g., *Chlorella sp*.) to larger phytoplankton (e.g., *Coelastrum reticulatum coenobia)*, and to the largest algae that are found in these conditions (e.g., *Cyclotella sp, Scenedesmus opoliensis;* Pagano, 2008). Unlike the other two groups, the rotifers exhibited uneven distribution patterns in the Baffled Pond and were more species-diverse than the other groups of zooplankton. Babu (2011) found that copepods and rotifers compete for the palatable forms of algae; therefore the abundance of copepods like *Nauplii* in the Baffled Pond was disadvantageous for the rotifers. Furthermore, this dispersed nature of rotifers in the Baffled Pond could be attributed to their preference to consume dispersed bacteria and small particles of organic matter, detritus and algae in the water column (Starkweather, 1980).

Ostracods were found only in the bottom of the first compartment of the Baffled Pond (**Table 5.3**). Their identification in this system may be of great importance to domestic wastewater treatment, because they are able to ingest massive quantities of a wide variety of food items within a few minutes and then survive starvation for several weeks (Vannier et al., 1998). Ostracods eat living prey, such as *polychaete worms*, or dead animals, including fishes and squid, which may explain why they were found at the bottom of the pond (Vannier et al., 1998).

Zooplanktons were almost entirely absent from the surfaces of the baffles. This may be explained by their nature of floating freely in the water column (van der Wende and Characklis**,** 1990). However, some individuals of *Nauplii, Copepodites, Brachionus calyciflorus, Epiphanes,* and *Asplanchna brightwelli* were encountered in some surface samples, which may be due to the presence of algae there and they had gotten trapped while grazing.

In summary, the abundant presence of zooplankton could play an important role in the control of bacterial and algal populations in the Baffled Pond. More interestingly, if this technology is combined with aquaculture, it would have great potential to contribute to the alleviation of hunger in low-income neighborhoods of cities. For instance, Oueda, (2009) found various species of fish, such as *Brycinus nurse, Oreochromis niloticus, Tilapia zillii, Synodontis membranaceus*, and *Clarias anguillaris* in Sahelian lakes to consume different sorts of aquatic insect larvae, terrestrial insects, phytoplankton and zooplankton. This can be an attractive application of aquaculture in the Sahel.

Table 5.3: Zooplankton distribution pattern at depths of 15, 60 and 90 cm in the four compartments of the Baffled Pond

Zooplankton (org/ 100 ml)	Compartment A			B			C			D		
depth (cm)	15	60	90	15	60	90	15	60	90	15	60	90
Copepoda												
Tropodiaptomus incognitus **(A)***	1000	1400	1800	1000	800	2600	600	1400	1400	400	1400	1600
Mesocyclops leuckarti **(B)**	600	1000	2000	2200	800	2400	400	1600	400	200	1400	1000
Nauplii spp **(C)**	1600	2000	2200	2000	1400	1200	1000	1600	1000	600	2000	1200
Copepodites spp **(D)**	1200	2400	1600	1600	600	1200	600	1800	1400	600	1800	1400
Cladocera (Branchiopodes)												
Ceriodaphnia cornuta **(E)**						400		400				
Diaphanosoma excisum **(F)**	600	1200	600	1400	600	1400		1600	1600	1000	1600	1800
Moinodaphnia maleayi **(G)**	400	1000	1000	800	1000	1400		1000	1200	200	400	1000
Moina micrura **(H)**	400	800	400	400								
Rotifera												
Filinia terminalis **(I)**		1000	2000	600	400	2800	200	600	1000	400	600	1000
Filinia longiseta **(J)**					400	2200	400	400	200	200	400	600
Keratella cochlearis **(K)**	200	400	400									
Keratella tropica **(L)**		600	800	200								
Brachionus falcatus **(M)**	600	400	400	200	600	600			1000	400	1000	1600
Brachionus caudatus **(N)**	400	1600	1800	1200		800		400	600	400	400	800
Brachionus quadridentatus **(O)**		200							600		400	800
Brachionus angularis **(P)**		800	1800	200					400	400		
Brachionus calyciflorus **(Q)**			1400	2000	200	1400	800	1600			1600	1600
Epiphane spp **(R)**	200	200	600	400	600	800	800					
Asplanchna brightwelli **(S)**	400	1200	2200	400	400	1400	400	600	800			1000
Asplanchnopus multiceps **(T)**	200	600	400	400		800						1000
Lecane luna **(U)**		400										
Hexarthra spp **(V)**	400	1200	600	600	800	1200	400	1000	1400	400	1800	1800
Ostracoda												
Ostracods **(W)**			400									

*The letters in parentheses () represent these zooplankton species in the Principal Components Analysis (**Section 5.3.3**)

5.3.3 Interactions between biotic and abiotic aspects of the Baffled Pond

Principal Components Analysis (PCA) and Canonical Correspondence Analysis (CCA) were used to visualise the correlations between biotic components including phytoplankton (Algae); zooplankton (**Table 5.3**); bacteria (*E. coli*, fastidious bacteria (**FB**), non-fastidious bacteria (**NB**) and enterobacteria (**EB**)); fungi and abiotic components comprising total suspended solids (**TSS**), pH, temperature (**Temp**), electrical conductivity (**Cond**), and dissolved oxygen (**DO**) at three depths in each of the four compartments of the Baffled Pond.

- **Principal Components Analysis of the abundance of the zooplankton species**

The plane formed by the axes F1 and F2 expresses 61.3% of the information (**Table 5.4**); therefore, this factorial plane was used to describe the matrix of correlation of the 23 zooplankton species (**Table 5.3**) that were present at 12 points along the transit of wastewater through the Baffled Pond.

Table 5.4: Eigenvalues and percentages of variance explained by Axes F1 and F2 of a Principal Components Analysis of the zooplankton species abundances in the various sectors of the Baffled Pond (**Figure 5.7**)

Axes	Eigenvalues	Percentage of variance explained (%)	Cumulative percentage (%)
F1	2.33	42.40	42.40
F2	1.04	18.92	61.32

From the scatterplot of the PCA Analysis, one can see that all of the zooplankton species that were identified tended to increase along Axis F1 (**Figure 5.7**). Wastewater enters with very few zooplankton and the species *Brachionus angularis* (**P**), *Brachionus caudatus* (**N**), *Asplanchna brightwelli* (**S**) *Filinia terminalis* (**I**), and *Mesocyclops leuckarti* (**B**) increase rapidly until the water reaches the bottom of the first compartment. Then, in the bottom of Compartment B, *Tropodiaptomus incognitus* (**A**), *Brachionus calyciflorus* (**Q**), *Diaphanosoma excisum* (**F**) and *Filinia longiseta* (**J**) are more common, while species P, N and S are less common. Next, abundances decrease as the water goes back to the surface and at C15 reach levels very similar to the influent. As the water goes back down in Compartment C, zooplankton become abundant again, but with fewer of species P, N and S (presumably indicators of contaminated water) and more of *Hexarthra spp.* (**V**), *Diaphanosoma excisum* (**F**) and *Filinia longiseta* (**J**) (potentially indicators of water that is less contaminated by people). The zooplankton fauna under Baffle 3 is considerably less abundant than that under Baffle 1, with a different composition of species. Then abundances diminish as the water rises toward the outlet and leaves the pond with few zooplanktons (**Figure 5.7**).

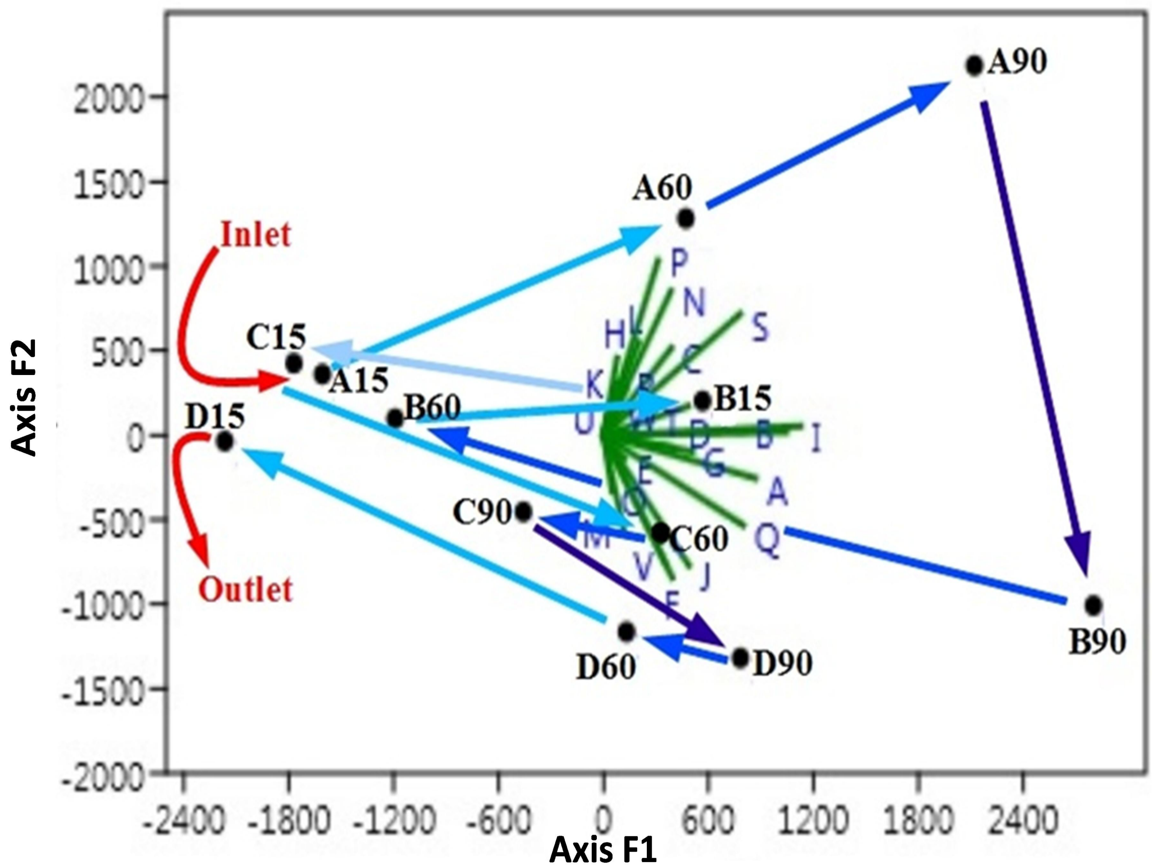

Figure 5.7: Scatterplot of a Principal Components Analysis of the abundance of 23 zooplankton species present at 12 points along the transit of wastewater through the Baffled Pond.

Note: Green lines show the direction and importance of the influence of each species (lettered in their order in **Table 5.3**). Blue arrows show the trajectory of the wastewater, with deeper tones for deeper waters.

- **Correlation between the amount of algae and of fungi developed on Baffles**

The abundances of the algae and fungi that grew on different parts of the baffles in the Baffled Pond, as described in **Section 5.3.1**, varied greatly. There is a strong correlation between the amount of algae and the amount of fungi, but with one distinct exception (**Figure 5.8**). The sample at the depth of 90 cm on Baffle 3 (**c90**) had inordinately more fungi than the other samples and was exactly adjacent to the great rise in fungi at the same depth in Compartment D (**D90**) in the water column. It is also worth noting that the highest concentrations of algae occurred in the upper two samples of the middle baffle. Possibly something inhibited them on the first baffle and there was a lack of nutrients when the water got to the third baffle.

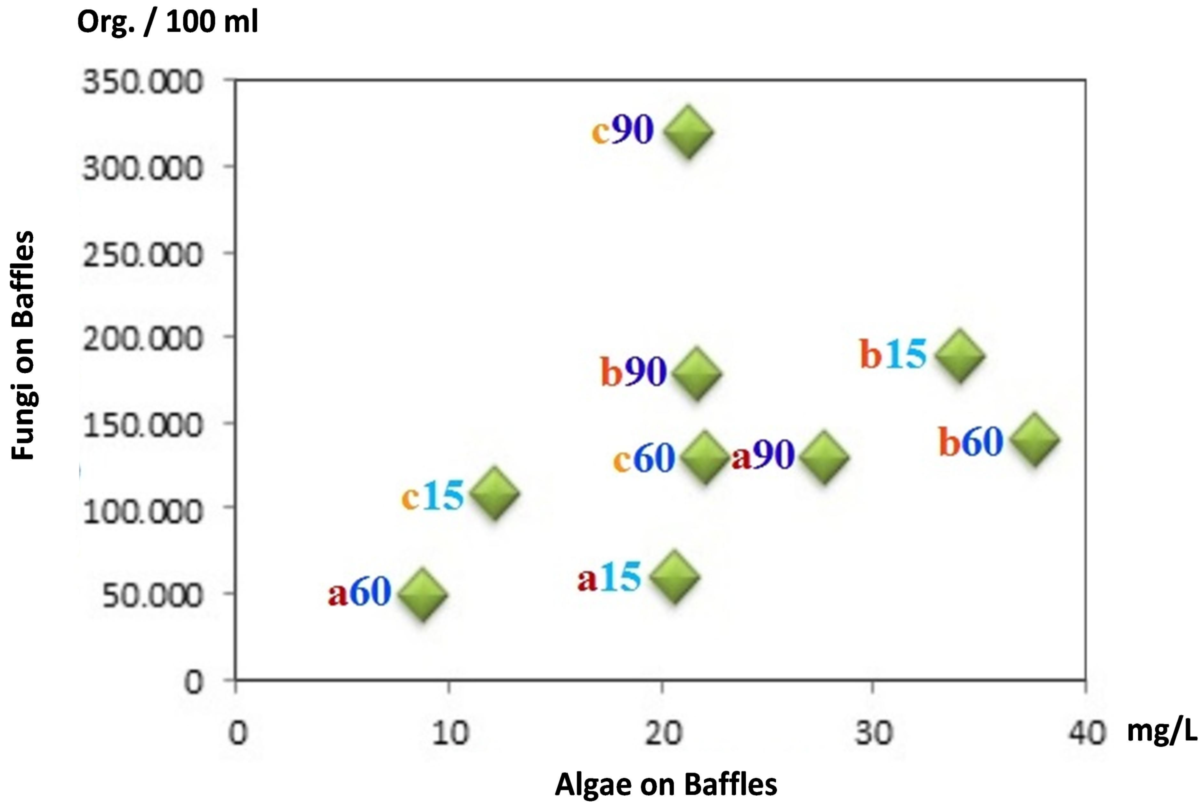

Figure 5. 8: Scatterplot of the abundances of algae and fungi on different parts of the baffles in the Baffled Pond.

Note: The three baffles are lettered (a, b, c) according to the compartment that precedes them and the numbers refer to the depth in centimetres within the water.

- **Canonical Correspondence Analysis of Physical-chemical Factors and Microbes**

The plane formed by the axes F1 and F2 expresses 93% of the variance in the data (**Table 5.5**); therefore, this factorial plane was used to describe the matrix of correlations among different types of microbes and various environmental factors across the different sectors of the pond (**Figure 5.9**)

Table 5.5: Eigenvalues of Axes F1 and F2, together with the percentages of the variance that they explain in a Canonical Correspondence Analysis of different types of microbes and physical and chemical factors in the various sectors of the Baffled Pond (**Figure 5.9**)

Axes	Eigenvalues	Percent of variance explained (%)	Cumulative percentage (%)
F1	0.35	67.8	67.8
F2	0.13	25.2	93.0

According to this multivariate ordination, the wastewater arrives with high numbers of fastidious bacteria and with a moderate amount of *E. coli* and these increase toward the bottom of each compartment, but are largely replaced by non-fastidious bacteria each time the water travels upward (**Figure 5.9**). In addition, it should be noted that the points representing Compartment B are in a straight line roughly parallel to the *E. coli*/NB + EB axis and this trend continued straight as the water flowed over the central baffle to C15, where the trend reversed as the water went down Compartment C with a slightly less straight line . By the time the water leaves the pond, the *E. coli* have been eliminated entirely (**Figure 4.2**), thus lending strong support for the effectiveness of this treatment.

The main deviation from this pattern is that in two separate points (A60, D90) the abundance of fungi increased greatly and potentially predominated over the bacterial processes mentioned above, under conditions of greater electrical conductivity and lower temperatures. The first of these occurred half-way down the first compartment (A), but upon reaching the bottom, fungi were eliminated entirely and *E. coli* was at its maximum (**Figure 5.9**). The second surge of fungi was at the bottom of the last column (with maximal electrical conductivity), immediately before the water rose to the outlet without any *E. coli*. So fungi may have played an important role in sanitizing the wastewater and potentially one could investigate ways to encourage the growth of fungi in these ponds.

One would expect the suspended solids to diminish during the transit through the pond, but the opposite is shown by this ordination to be the case, potentially with more solids being generated by the algae (**Figure 5.9**). The ordination implies and the data confirm that the water at the outlet had the highest values for dissolved oxygen, but also pH, suspended solids, algae, and non-fastidious bacteria, so this water has been sanitized in terms of *E. coli* and presumably pathogenic organisms, but it is still far from being clean water. It contains abundant nutrients, solids and microorganisms, so it should best not be released into rivers or bays, but instead used productively in aquaculture or agriculture.

Dissolved oxygen and pH showed almost exactly the same trends, so they seem to be represented by the same orange line. Also note that the adjacent points A90 and B90 had such similar conditions, at the bottom, below the first baffle, that they occupied nearly identical spots on **Figure 5.9**.

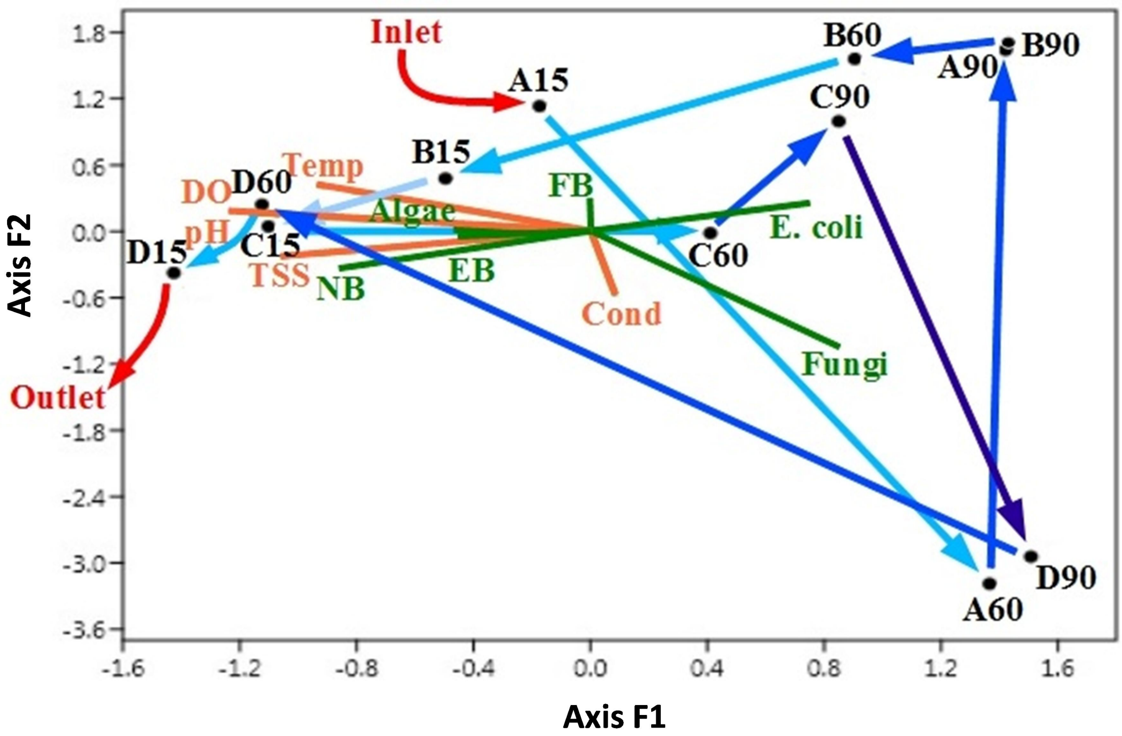

Figure 5.9: Scatterplot of a Canonical Correspondence Analysis of the abundance of algae, fungi, and four categories of bacteria (E. coli, FB- fastidious bacteria, NB- non-fastidious bacteria, and EB-enterobacteria) present at 12 points in the water column along the transit of wastewater through the Baffled Pond, also taking into account five environmental variables: total suspended solids (TSS), pH, temperature (Temp), electrical conductivity (Cond), and dissolved oxygen (DO).

Note: Orange lines show the direction and importance of the environmental variables. Green lines do the same for each of the six types of microbes. Blue arrows show the trajectory of the wastewater, with deeper tones for deeper waters.

- **Principal Components Analysis of Physical-chemical Factors, Microbes and zooplankton**

The plane formed by the axes F1 and F2 expresses 100 % of the information (**Table 5.6**); therefore, this factorial plane was used to describe the matrix of correlation among the physical-chemical factors, microbes and zooplankton. This separated the variables into three groups (**Figure 5.10**).

Table 5.6: Eigenvalues of Axes F1 and F2, together with the percentages of the variance that they explain in a Principal Components Analysis of different types of microbes, zooplankton , and physical and chemical factors in the various sectors of the Baffled Pond (**Figure 5.10**)

Axes	Eigenvalues	Percent of variance explained (%)	Cumulative percentage (%)
F1	25.85	92.34	92.34
F2	2.15	7.66	100.00

Indeed, the F1 axis clearly highlighted two opposite groups (1 and 2), whereas the F2 axis isolated zooplankton species in the compartment B in positive values (**Group 3**; **Figure 5.10a**). The first group which is positively correlated with F1 axis includes biotic components (bacteria, algae) and all monitored abiotic parameters. For the case of the second group which is negatively correlated with F1 axis comprised only zooplankton in Compartments A, C, D. Coupling this analysis with respect to the sampling depths in the factorial plan (**Figure 5.10b**), biotic components in Group 1 colonized better the top layers, while biotic components in Group 2 preferred the deeper layers. Thus, in the scale of this Baffled Pond, bacteria and algae were shown to be pelagic species and that of zooplankton in Group 2 as benthic.

Furthermore, PCA results clearly showed the good correlation among the bacteria, algae, TSS and the resulting pH and dissolved oxygen in the top layers of all compartments of the Baffled Pond. This implies a positive relationship, where an increase of one group leads to the proliferation of the other group. Therefore, the PCA analysis has proven once more evidence of the symbiotic algal-bacterial activity and abiotic parameters such as pH, dissolved oxygen and temperature interdependences in the course of organic matter degradation in the top layer of the Baffled Pond. In addition, the very strong negative correlation between Group 1 and Group 2, confirms the predation relationship of the zooplankton at one side and the bacteria and algae on the other hand. This indicates that any increase of zooplankton would lead systematically to the decrease of bacteria and algae. Zooplankton dynamics was seen to depend on factors such as temperature and pH, as has been found in other studies (Pourriot and Champ, 1982; Oueda et al., 2007).

Although, zooplankton in Compartment B had a positive correlation with F2 axis, this result may be explained by the fact that the water flowed upward. However, this trend was not observed in Compartment D which has similar flow pattern. Further study can be conducted to clarify this issue.

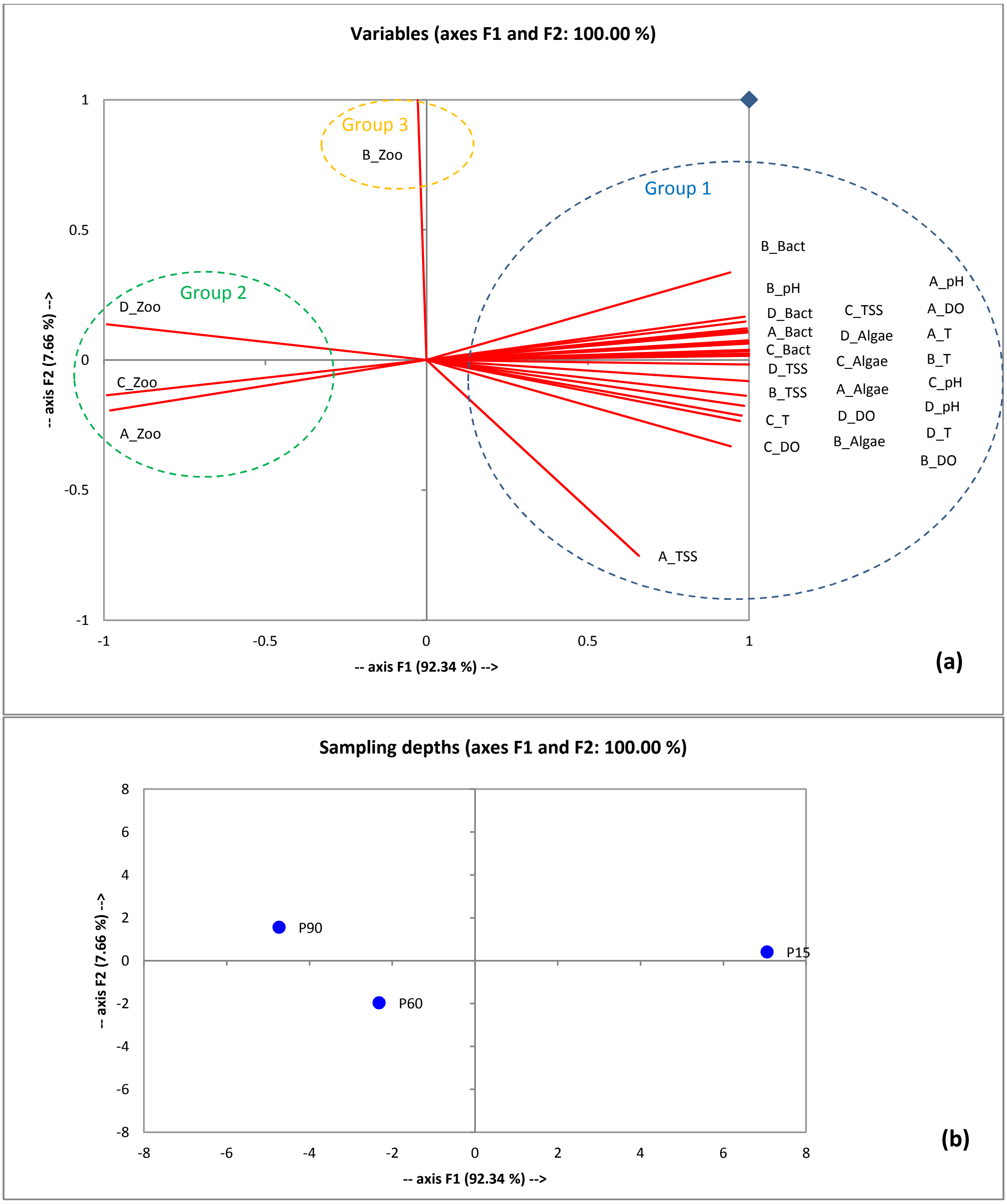

Figure 5.10: Correlation scatterplot of the PCA analysis of the Baffled Pond showing: (a) the grouping of components and (b) the sampling depths position in the factorial plan

Note: Group 1: **A_Bact, B_Bact, C_Bact, D_Bact**: Fastidious, non-fastidious and enterobacteria bacteria in compartment A, B, C, and D respectively and at depth 15, 60 and 90 cm (**P_{15}, P_{60}**and **P_{90}** respectively); **A_Algae, B_Algae, C_Algae, D_Algae**; **A_TSS, B_ TSS, C_ TSS, D_ TSS; A_pH, B_pH, C_pH, D_pH; A_DO, B_DO, C_DO, D_DO;** and **A_T, B_T, C_T, D_T:** Algae; Total suspended solids; pH, dissolved oxygen; and temperature respectively in the four compartments (A, B, C, D) at depth 15, 60 and 90 cm

Group 2: **A_zoo, C_zoo, D_zoo**: all identified zooplankton (**Table 5.3**) in compartment A, C, and D respectively and at depth 15, 60 and 90 cm

Group 3: **B_zoo**: all identified zooplankton (**Table 5.3**) in compartment B and at depth 15, 60 and 90 cm

5.3.4 Sludge accumulation rates in the Baffled Pond

After two years of operation, the distribution of sludge was very uneven with thickness varying from 3 to 7 cm (**Figure 5.11**). In addition, the maximum sludge thickness was found to occur near the inlet, outlet and around Baffle 2 in the middle (**Figure 5.11**). This distribution pattern is consistent with the findings of several authors in similar and different climatic conditions (Cavalcanti and Van Haandel, 2001; Picot et al., 2005; Keffala et al., 2011; Konate et al. 2010, 2013). Indeed, many factors have been reported to influence the sludge distribution in ponds, including: pond geometry, wind effect, pond age, sedimentation of dead biomass (algae), and rainwater infiltration (Cavalcanti and Van Haandel, 2001; Nelson et al., 2004; Picot et al., 2005; Keffala et al., 2011). Furthermore, the accumulation rate was not reported to be constant and it decreases with time due to anaerobic degradation and consolidation of sludge. For example, Hammou et al., (1992) reported an accumulation rate of 4.3 cm/year in a primary pond in Meze (France) after 8 years of operation and Picot et al., (2005) found 2.7 cm/year after 14 years of operation of the same pond.

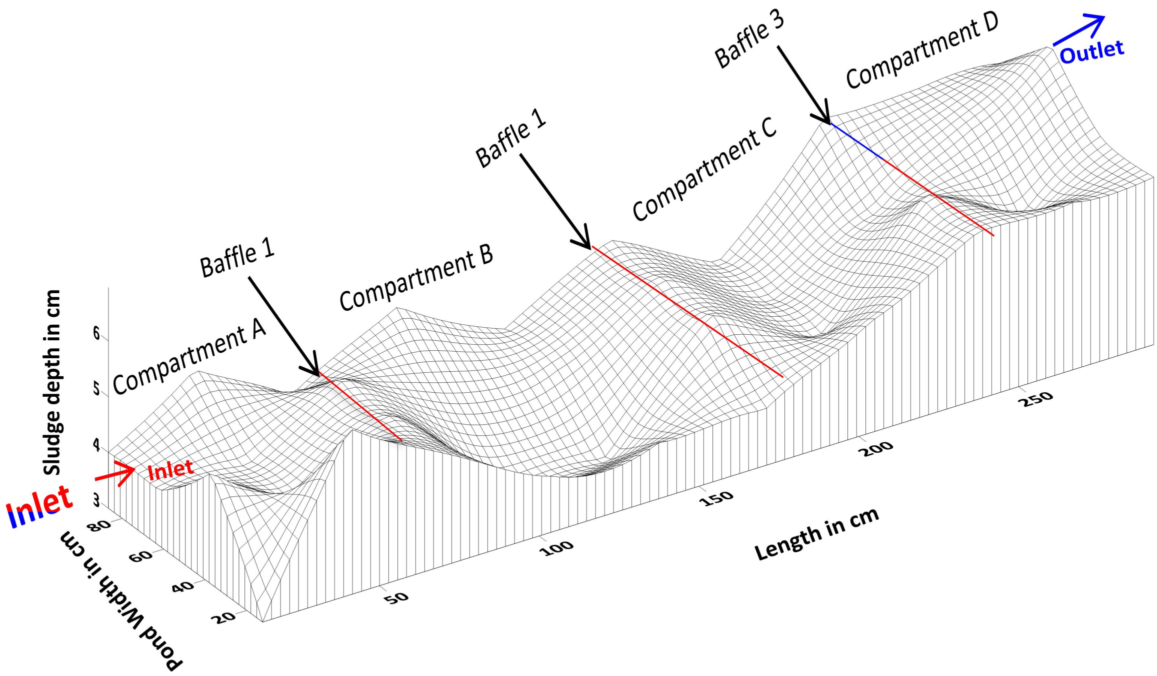

Figure 5.11: Sludge distribution pattern in the Baffled Pond after 2 years of operation

For a population equivalent of 50 PE, after two years of operation, the rate of sludge accumulation in the Baffled Pond was estimated to be 0.0014 m^3 per capita per year, which represented 4 % of the total volume of the pond (**Table 5.7**). It is generally assumed that ponds should be desludged when they are 30% full. If the current accumulation rate remains stable, the Baffled Pond would require desludging every 15 years. This coincides with the recommendation in France for the desludging interval of primary facultative ponds (Picot et al., 2005).

Table 5.7: Sludge accumulation rate in the Baffled Pond

Parameter	Baffled Pond				
	Compartment A	Compartment B	Compartment C	Compartment D	Total
Sludge volume m^3 in 2 years	0.022	0.041	0.049	0.030	0.142
Sludge volume m^3 per year	0.011	0.021	0.024	0.015	0.071
Sludge volume m^3 per capita per year*	0.0002	0.0004	0.0005	0.0003	0.0014

*For a population equivalent of 50 persons

The sludge accumulation rate was very low compared to the values reported in the literature (**Table 5.8**; Gomes de Souza, 1987; Mara and Pearson, 1998; Nelson et al., 2004; Konate et al., 2010). This low accumulation rate of sludge in the Baffled Pond under a Sahelian climate may be due to the constant mesophilic conditions that favour active sludge digestion, and also due to the fact that the Baffled Pond is preceded by two-stage high-rate anaerobic reactors, where most of the sludge is removed. Therefore, combining two-stage high-rate anaerobic reactors with a baffled pond seem to provide an excellent alternative to minimize sludge handing, and thus reduce the cost of operation and maintenance.

Table 5.8: Sludge accumulation rates in various waste stabilization ponds

Parameter	Time of operation	Capacity	Volume of sludge accumulated	Sludge accumulation rate			Reference
	(years)	(person equivalent, PE)	(m^3)	(cm/year)	(m^3/PE/year)	(kg_TSS/PE/year)	-
Baffled pond with attached growth at 2iE in Burkina Faso	2	50	0.141	-	0.0014	0.28	Current study
Maturation Pond in WSP at 2iE in Burkina Faso	5.5	448	15.5	1.3	0.007	0.26	Konate *et al.*, 2013
Maturation pond in WSP at Tunis	8	282	121.79	1.6	0.029	-	Keffela *et al.,* 2011
19 WSP in the south of France	12 to 24	120 to 10000		1 to 2.7	0.04-0.148		Picot et al., 2005

5.4 Conclusions

A great diversity and density of algae, bacteria, fungi and zooplankton were found to be living in the water column and on the attached media of the Baffled Pond, after two years of operation, and many of these certainly contributed to wastewater treatment. Thick, green biofilms formed on the upper parts of the both sides of each baffle (both on the plastic that forms the baffles and the plastic bottle caps affixed to it), reaching 370 g/m^2 and decreasing with depth to a minimum of 0.1 g/m^2. The Baffled Pond had both aerobic and anaerobic zones and may be considered as facultative pond. Three major groups of diverse zooplankton were found in the water column: Cladocera, Copepoda and Rotifera. This last group was dominant with 14 identified species, which consume a wide spectrum of natural food items, including bacteria and algae. The various types of microbes that were tested for varied greatly in their abundances in the different sectors of the water column of the Baffled Pond, especially the non-fastidious bacteria that increased near the surface (in each compartment) as the water approached the outlet. Indeed, strong correlations between certain types of the bacteria and suspended solids seem to correspond to the adsorption phenomenon of bacteria in suspended matter that is known to occur. The PCA and CCA analysis showed more evidence of the symbiotic algal-bacterial activity and the importance of abiotic parameters such as pH, dissolved oxygen and temperature in the degradation of organic matter. In addition, the very strong negative correlation between Group 1 and Group 2, confirms the predation relationship of the zooplankton at one side and the bacteria and algae on the other hand. As a result, the parasitic symbiosis distributions of phytoplankton and zooplankton have shown that the baffles had an effect on water quality which in turn has affected the ecology of the baffled pond.

The dense and abundant zooplankton community may play an important role in the control of bacterial and algal populations in the Baffled Pond. These same microscopic animals may serve as food for fish in aquaculture at the end of the wastewater treatment.

There was a very low rate of sludge accumulation in the Baffled Pond of only 0.0014 m^3 per capita per year, which indicates that sludge handing would be minimized and could be done at low cost taking into account the financial constraints prevailing in sub-Saharan Africa. The use of the Baffled Pond effluent to produce fish and vegetables in aquaculture, followed by irrigation of urban agriculture, would be an attractive area of research, to contribute to the alleviation of hunger in low-income neighborhoods.

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Chapter 6

6. Biogas production from high-rate anaerobic reactors in Sahelian climate

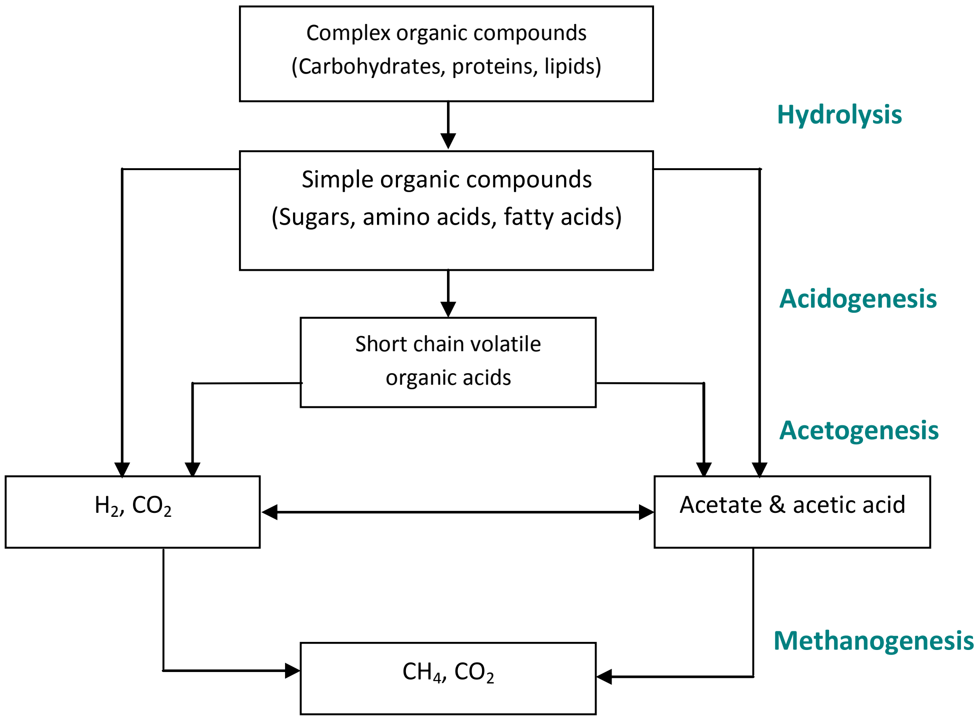
This final chapter presents and discusses the potential for biogas production, its quality, and the rate of sludge accumulation in two pilot-scale anaerobic reactors at the 2iE Campus in Ouagadougou, Burkina Faso, within the Sahel Region, just to the south of the Sahara Desert.

6.1 Biogas recovery from anaerobic treatment processes

Anaerobic wastewater treatment has been given more attention over the aerobic wastewater treatment since the energy crisis of the 1970s, together with the increased demand for industrial wastewater treatment (Henze *et al*., 2008). Indeed, anaerobic biological wastewater treatment systems are recognized for their efficient organic matter removal, low energy requirements, their potential for valuable resources recovery, such as methane-rich biogas for energy and stable sludge as organic fertilizer for agriculture (Van Lier et al., 2008; Bodík et al., 2011; Lohani et al., 2013). In addition, these systems produce more biogas in hot, tropical areas (Haandel and Lettinga, 1994).

Generally, the anaerobic degradation pathway of organic matter to produce biogas is a multi-step process of reactions in series and in parallel. This process of organic matter degradation is known to proceed in four key biological and chemical stages namely: hydrolysis, acidogenesis, acetogenesis and methanogenesis (**Figure 6.1;** Haandel and Lettinga, 1994; Van Lier et al., 2008). The methane-producing bacteria are located at the end of this food chain, which indicates the unidirectional degradation of organic matter to the end products of methane and carbon dioxide. However, in practice, Van Lier et al., (2008) reported that other reverse reactions may occur to produce high concentrations of volatile fatty acids (VFA) and alcohols, which are of capital importance in the control and operation of an anaerobic digester.

Biogas production is dependent on several environmental variables, including substrate type and quality, temperature, pH, retention time, degree of wastewater treatment, competition between methanogens and sulphate-reducing bacteria, and toxicants (El-Fadel and Massoud, 2001; Stadmark and Leonardson, 2005). For instance, the hydrolysis process, which is considered to be the rate-limiting step during anaerobic digestion of complex substrates, is very sensitive to average temperature and temperature fluctuations (Van Lier et al., 2008). On the other hand, methanogenesis is sensitive to both high and low pH and occurs between pH 6.5 and pH 8. According to Haandel and Lettinga (1994), methanogenesis is often the rate limiting step of the entire anaerobic digestion process, because it has a cell 'doubling time' of a few days compared with the few hours required for acetogenic bacteria.



Source: Haandel and Lettinga, (1994)

Figure 6.1: Reaction sequence for the anaerobic digestion of complex macromolecules

In the field of domestic sewage treatment, many types of anaerobic reactors have been designed and implemented specifically to recover biogas. The Upflow Anaerobic Sludge Blanket reactor (UASB), which was developed by Lettinga and co-workers (Lettinga et al., 1980), represents the most widely, used system for anaerobic wastewater treatment in many parts of the world. For example, in a UASB reactor, 28% to 75% of the chemical oxygen demand (COD) is transformed into energy in the form of the methane gas (Mendoza et al., 2009). Furthermore, a number of researchers have investigated the potential for biogas production from other anaerobic treatments systems, such as conventional, duckweed-based, and algae-based anaerobic ponds (Van der Steen et al., 2004; Konate et al., 2013; Sims et al., 2013). These have found a huge potential to produce biogas with a high content of methane (80%), if the necessary collection systems is constructed.

If this methane is not collected, wastewater treatment contributes significantly to total greenhouse gas (GHG) emissions is, because it involves conversion of organic matter into biogas (mainly CH_4 and CO_2) (Kärrman and Jőnsson, 2001; Van der Steen et al., 2004; Stadmark and Leonardson, 2005, 2007; Hospido et al., 2007; Margarita and Scarlette, 2007; Show and Lee, 2008; Sims et al., 2013). Wastewater treatment may account for about 5% of global methane emissions (Czepiel et al., 1993). According to Le et al., (2007), 0.3 to 0.6 kWh is required to treat one cubic meter of wastewater using membrane bioreactors and 0.2 to 0.4 kWh/m^3 is needed when using the conventional activated sludge process (Amy, 2008). In addition to the high energy consumption, the waste management sector has contributed about 3 to 4 percent of the annual global anthropogenic (man-made) GHG emissions (IPCC, 2006).

Therefore, one of the mitigation measures is to apply wastewater treatment options that consume less energy, combined with energy recovery and reuse. Hence, it is in this regard that the potential for biogas production in terms of quantity and quality from a two-stage high-rate anaerobic reactors treating domestic wastewater under a Sahelian climate is investigated. In fact, this present work could be seen as an extension of the study by Konate et al., (2013), who measured biogas production and composition generated from an anaerobic pond treating domestic wastewater in the same area. The main difference from their study resides in the type of wastewater treatment technology, since it has been reported to affect the amount of methane released per kilogram of BOD treated (Van der Steen et al., 2004). In addition, the accumulation of sludge from these anaerobic reactors was also measured.

6.2 Methodology

6.2.1 Description of the experimental setup

The two-stage high-rate anaerobic reactors used in this study were as described in **Chapter 2** (**Figure 2.3**), as were their operational conditions.

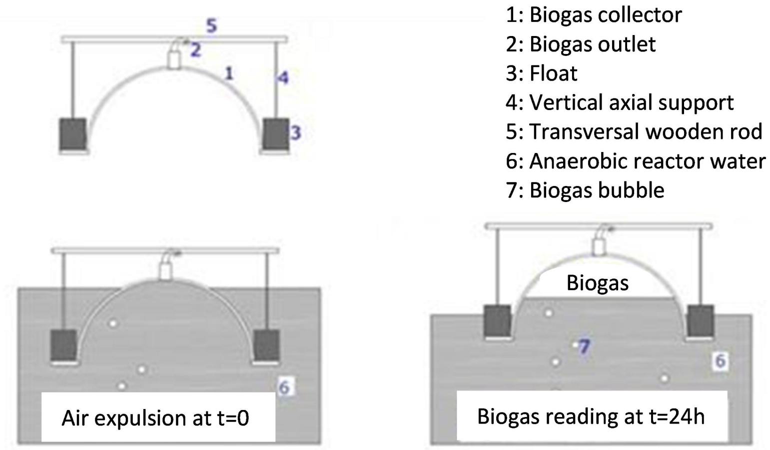
6.2.2 Biogas collection, sampling and analysis

The production of biogas was measured daily from March to August 2015 with two floating static chambers in Plexiglas adapted from the collectors described in a similar study on biogas collection in the same climatic conditions (Konate et al., 2013). These devices were used, because gas-tightness could not be achieved above the water surface inside the anaerobic reactors (the 0.5 m^3 that was left for the biogas collection and storage).

The device used here for the biogas collection consisted of a floating static chamber, with a basal area of 0.138 m^2. It is a half-sphere in shape and covered 9% of the total surface area of each anaerobic reactor (1.538 m^2; **Figure 6.2)**. All the two collectors were supported at the surface with floats and anchored with strings to the top of the chamber to prevent any disturbance due to internal mixing of the reactor. The collectors were opaque and also the reactors lids were set in order to stop ultraviolet (UV) penetration and to prevent algae from growing within the system (Picot et al., 2003). Therefore, the increase in O_2 concentrations in the collectors was avoided as reported by previous studies (Brockett, 1976; Sharpe and Harper, 1999). Every 24 h, the buoyancy position of the collector was read with a graduated scale established on each collector, which gave the corresponding volume of biogas that had been produced. This reading of the biogas volume had an estimated accuracy of 3%. The measurements of biogas production were done every day for six months. The two gas collectors were placed in the middle of the water surface inside the reactor. The ideal gas formula (**Equation 6.1)** was used to correct for the real volume of the collected biogas (R= 0.0821 at P= 1atm, T= 20°C, and n= 1 mole).

PV=nRT **(6.1)**

Where: **P** is the pressure of the gas (atm)
V is the volume of the gas (m^3)
T is the temperature (°C)
n is number of moles
R is the universal gas constant (R= 0.0821 at 1 atm, 20°C and 1 mole)



Source: Konate et al., (2013)

Figure 6.2: Schematic view of the biogas collector

The composition of the biogas in terms of CH_4, CO_2, H_2S and other trace gas was determined three times a week directly *in situ*, using a portable biogas analyser (Geotech GA 5000; **Figure 6.3**). This apparatus was designed specifically to measure gases from landfills and other anaerobic sources, with certified allowable ambient temperatures between - 10 ºC and + 50 ºC. It should be noted that the inlet and outlet pressures should not exceed +/- 500 mbar and +/- 100 mbar with respect to atmospheric pressure respectively. Calibration of the apparatus was carried out using various types of gas depending on the purpose the study.

Besides, the biogas from the collectors of both anaerobic reactors were sampled once a month in tedlar bags and transported to 2iE laboratory for analysis with micro-gas chromatograph (Micro-GC). This was done in order to cross check the results from the field analyses. In order to optimize the quality of the results, each biogas sample was analyzed in triplicate.

To appreciate the additional value of each anaerobic reactor (R1 and R2), statistical tests were conducted with STATISTICA 8.0 software (IBM). The t-test was used to test significant differences between the two anaerobic reactors in terms of biogas production at the 5% significance level.

Note: **B** and **E** are the only ports of interest in this study.
B: Biogas inlet
E: Biogas outlet

Figure 6.3: Portable biogas analyser GA 5000

6.2.3 Wastewater characterization

Grab samples of 500 ml were taken each week during the six months, between 8:00 and 9:00 a.m., at the following points in the system:

- the influent wastewater (RW),
- the effluent of the first anaerobic reactor (R1), and
- the effluent of the second anaerobic reactor (R2)

The collected samples were stored at 4°C and analysed within 3 hours for TSS, BOD_5, COD, and VSS in the 2iE Laboratory (**Table 2.6**), whereas pH, temperature, dissolved oxygen (DO) and redox potential were measured immediately, *in situ* with an integrated portable pH-T-DO meter (WTW 340i/SET), according to the Standard Methods APHA (2012). Volatile fatty acids (VFA), expressed as acetic acid, were determined by an alkalimetric method using a two-stage sequential titration (Anderson and Yang, 1992).

6.2.4 Sludge accumulation assessment

The "white towel technique" was used to assess the sludge that had accumulated in R1 and R2 after two years of continuous operation. The method consisted of wrapping a white towel around the bottom of a cylindrical iron rod attached with a graduated tape. The depth of the sludge was measured by lowering the rod vertically into the tank until it reaches the bottom; it is then slowly withdrawn to read the depth of sludge through the thick mark left on the towelling material (Konate et al., 2013). The anaerobic reactors were divided into bathymetric sections, each with a radius of 7 cm, and a total of 11 locations for each reactor were measured for sludge thicknesses.

6.3 Results and Discussion

6.3.1 Characteristics of the influents and effluents of R1 and R2

Table 6.1 summarizes the averages and standard deviations of influent and effluent concentrations for the parameters that were analysed during the six months. The temperature of the influent wastewater averaged 26 ± 2 °C , it increased to 39 ± 3 °C upon leaving R1, and lowered slightly to 38 ± 4 °C in the course of going through R2 (**Table 6.1**). Therefore, the use of anaerobic reactors painted black in the sunny climate of the Sahel resulted in an increase in temperature (**Section 2.3.2**). Temperature plays a decisive role in treatment processes, decreasing land requirements, enhancing conversion processes, increasing removal efficiencies, and boosting biogas production. Increasing temperatures in anaerobic digesters may even shift the microbial diversity from acetoclastic methanogens to hydrogenotrophic methanogens, which produce more methane (Liu et al., 2002; Pender et al., 2004).

The pH values of the influent sewage and the effluents of the anaerobic reactors were in the neutral range, which indicates favourable anaerobic conditions for bacterial growth and biological degradation of organic matter to produce biogas (Peña, 2002; Foresti et al., 2006). Dissolved oxygen was consistently low in the influent wastewater and the effluents of R1 and R2 (**Table 6.1**), as is expected in anaerobic conditions. In addition, the redox potential measured in the influent and effluents of both anaerobic reactors had negative values, thus conditions were favourable for methanogenesis as the exclusive terminal microbial process (Kotsyurbenko et al., 2004). There was a decrease in VFAs from 66 ± 46 mg/l in the influent wastewater to 36 ± 16 and 25 ± 9 mg/l in the effluents of R1 and R2 respectively, suggesting that a good conversion from acitogenesis to methanogenesis was taking place (Pender et al., 2004).

COD, BOD_5, TSS and VSS concentrations of the influent wastewater were lower than normal for the domestic wastewater of developing countries (Metcalf & Eddy, 2003; von Sperling & Chernicharo, 2005; Henze *et al.,* 2008). Besides, its COD/BOD_5 ratio was 2.1, indicating that it was easily biodegradable domestic wastewater (Metcalf & Eddy, 2003; Henze *et al*., 2008). In summary, the results of this six-month monitoring session are similar to those of **Chapter 2** on the performance of the anaerobic reactors, demonstrating the reproducibility of the experiment.

Table 6.1: Influent and effluent characteristics of the anaerobic reactors R1 and R2

Parameters	RW		R1		R2		n
	Avg*	STD⁺	avg	STD	avg	STD	unit
T (°C)	26.73	2.12	39	2.89	38.75	3.64	24
pH (value)	7.28	0.21	7.21	0.10	7.29	0.13	24
Disolved oxygen (mg/l)	0.81	0.4	0.66	0.7	0.57	0.20	24
Redox potential (mV)	-29.6	12	-24.3	10.0	-29.0	12.0	24
Total DBO_5 (mg/l)	166	49	120	59	82	46	24
Total COD (mg/l)	357	127	235	87	173	39	24
TSS (mg/l)	61	14	38	7	20	13	24
VSS (mg/l)	49	17	20	10	10	8	24
VFA (mg/l of acetic acid)	66	46	36	16	25	9	24

6.3.2 Biogas production rates

The daily variability in biogas production in the two anaerobic reactors was closely related to temperature during these six months (**Figures 6.4 and 6.5**). This correlation with water temperature is well established (Toprak, 1995; Hodgson and Paspaliaris, 1996; Picot et al., 2003, 2011; Konate et al., 2013). In the current study, water temperature varied from 30.1 ˚C to 42.7 ˚C with an average of 39.0 ˚C for R1, while in R2 it varied from 29.1 ˚C to 41.4 ˚C with an average of 38.7 ˚C, thus they operated in mesophilic temperatures. For both reactors, the highest production rates were recorded when the temperature in the reactors was highest, and decreased when the temperature was lower. The variation in the biogas production rate also varied according to the organic loading rate, as has been previously demonstrated (Konate et al., 2013).

The first anaerobic reactors of this study (R1) produced an average daily volume of biogas of 107 ± 17 L/day (9.7 ± 1.5 L/m^{2}day) or 2.5 L/g of volatile suspended solids (VSS) removed, while the second one (R2) generated 105 ± 14 L/day (9.5 L/m^{2}day) or 1.8 L/g VSS removed. R1 produced slightly more than R2 because it was first in the series and received a higher organic load, which may also explain its higher temperature. Although biogas production rates were mostly higher in R1, t-tests showed no significant difference between the two reactors. Moreover, the biogas production rates were found to be lower than those that Konate et al., (2013) obtained from ananaerobic pond treating domestic wastewater under the same climatic conditions, but with a higher organic load. This difference in biogas production may also have been due to other environmental factors and operational conditions, such as the running period of the system. In any case, organic loading rate is a key factor in biogas production (El-Fadel and Massoud, 2001; Stadmark and Leonardson, 2005).

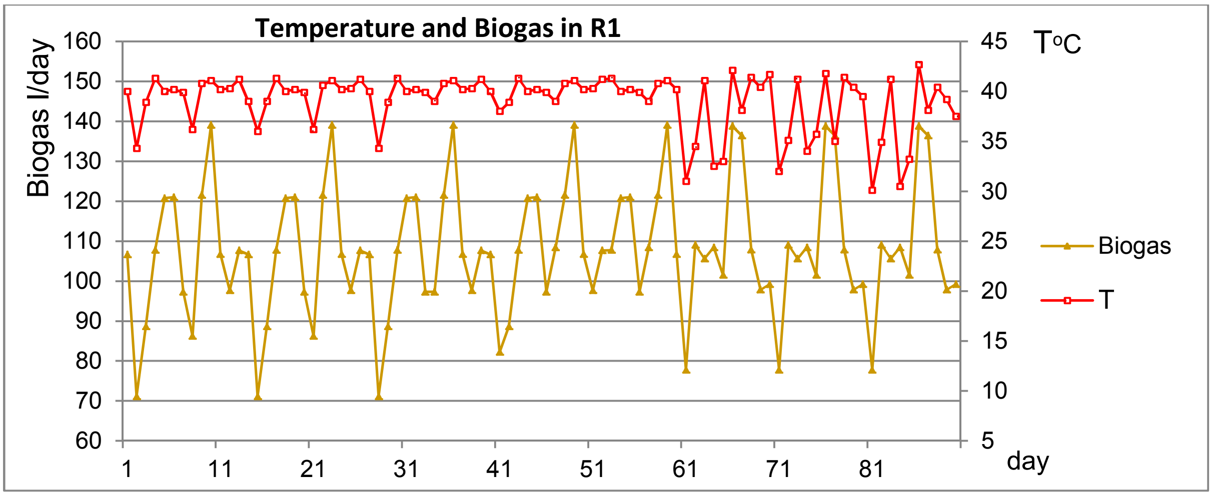


Figure 6.4: Variations in biogas production and internal water temperature over time in the first anaerobic reactor (R1)

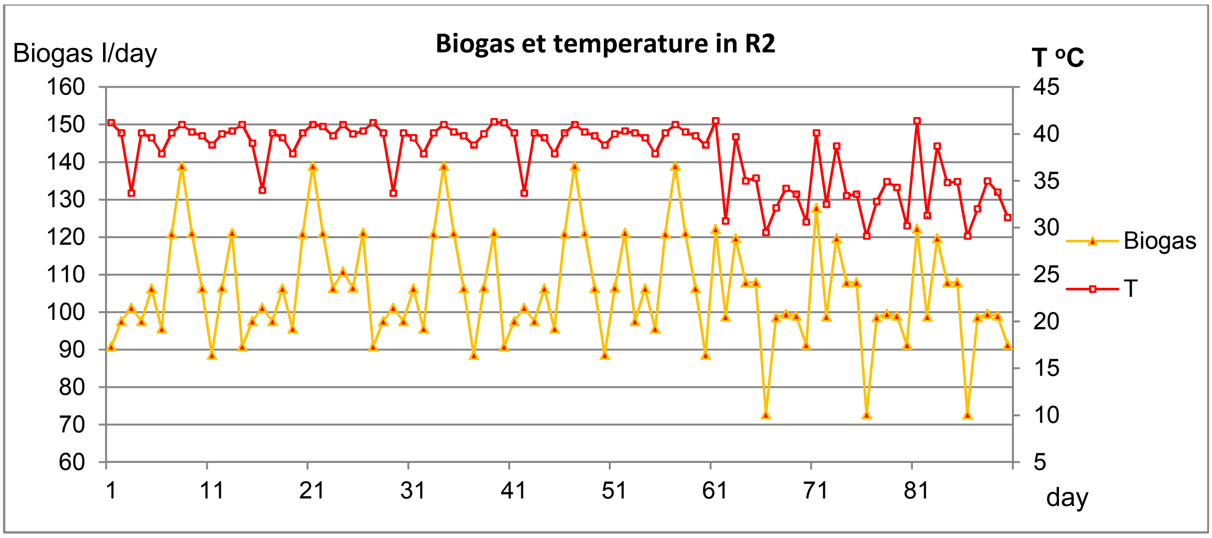


Figure 6.5: Variations in biogas production and internal water temperature over time in the second anaerobic reactor (R2)

6.3.3 Biogas composition

The biogas collected during this six-month monitoring period (72 samples) had the following composition:

- 54 % ± 10 methane, 6% ± 1 carbon dioxide, 8 % ± 2 N_2 and 32 % of other gases (H_2, H_2S, H_2O, …) for R1
- 44 % ± 5 methane, 12 % ± 2 carbon dioxide, 9 % ± 1 N_2 and 34 % of other gases (H_2, H_2S, H_2O, …) for R2 (**Figure 6.6**).

Methane content was higher in the first anaerobic reactor, potentially due to its higher temperatures and organic loadings being more conducive to the activity of the microorganisms. The hydrogen sulphide (H_2S) content for both reactors was very low, even negligible in R2 (1 ppm and 0 ppm in R1 and R2 respectively). Konate et al., (2013) attributed this to the scarcity of sulphates in the wastewater of Burkina Faso. On the other hand, the content of other gases was not negligible, since about 32 % of the biogas consisted of other gases, such as H_2, H_2O. This may be attributed to denitrification processes occurred in the anaerobic reactors. Nonetheless, the methane content found in this study was similar to that found elsewhere (Hodgson and Paspaliaris, 1996; Kotsyurbenko et al., 2004).

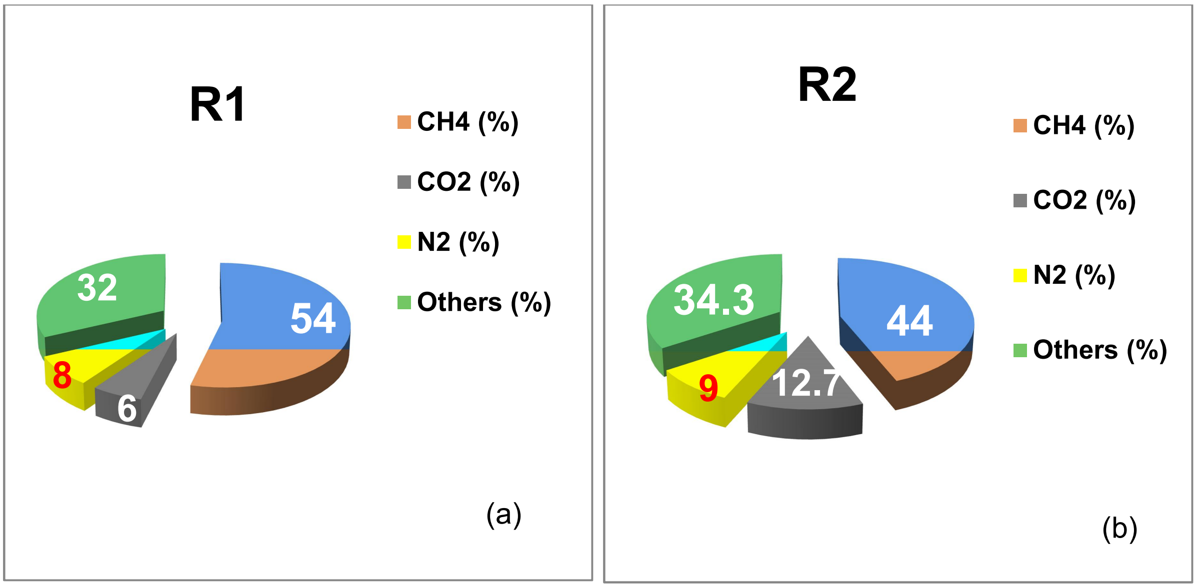


Figure 6.6: Average composition of the biogas produced from (a) R1 and (b) R2

6.3.4 Sludge accumulation rates

As has been seen in anaerobic ponds (Konate et al., 2010, 2013), sludge distribution within the anaerobic reactors, after two years of operation, was uneven (**Table 6.2)**. The deepest sludge was found in the middle, and near the outlets of the reactors, as has been found elsewhere (Picot et al., 2005; Keffala et al., 2011; Konate et al., 2010, 2013). Sludge was deeper in the first reactor because most suspended solids settled there before continuing to the second reactor.

On the other hand, for a population equivalent of 50 PE, after two years of operation, the rates of sludge accumulation in the anaerobic reactors were estimated to be only 0.0006 and 0.0002 m^3 per capita per year, in R1 and R2 respectively (**Table 6.2**). These accumulation rates were very low, compared other cases (Gomes de Souza, 1987; Mara and Pearson, 1998; Keffala et al., 2011; Picot et al., 2005) and even those of similar climatic conditions (Nelson et al., 2004; Konate et al., 2010, 2013). Firstly, this low accumulation rate may be due to the constant mesophilic temperatures in the Sahelian, which may have favoured intense sludge digestion. Secondly, the influent wastewater may have had an especially low content of suspended solids and organics (**Table 6.1**), combined with settling and digestion that may have occurred in the buffer tank at the inlet to the pilot project. In summary, this configuration of two-stage high-rate anaerobic reactors seems to provide an excellent option for minimizing sludge handing, and hence, reduce the cost of operation and maintenance.

Table 6.2: Sludge distribution and accumulation in the anaerobic reactors R1 and R2

Parameters		Sampling position										
		Inlet(P1)	P2[+]	P3	P4	P5	P6	P7	P8	P9	P10	Outlet (P11)
Sludge depth in cm	R1	3	3	5	5	5	3	3	3	3	3	4
	R2	1	1	1	2	2	2	2	1	1	1	2

	R1	R2	R1 + R2
Sludge volume m^3 per 2 years	0.06	0.02	0.079315
Sludge volume m^3 per year	0.0287	0.0109	0.0397
Sludge volume m^3 per capita year*	**0.0006**	**0.0002**	**0.0008**

[+]Position

*For a population equivalent of 50 persons.

6.4 Conclusions

There is great potential for recovering biogas from the two-stage high-rate anaerobic reactors treating domestic wastewater under a Sahelian climate, with the added benefit of very low sludge accumulation rates. Over 9 liters per meter square per day of biogas were produced per 18 liters per gram of VSS removed with a high methane content of 44 to 54%. The industrial or domestic use of biogas would be a viable energetic option of energy in African countries. Biogas production rates were closely associated with the water temperature inside each reactor. More importantly, very low sludge yields were recorded in both anaerobic reactors (0.0008 and 0.0002 m^3 per capita per year). This low production of sludge may be due to the high biodegradability of the domestic wastewater, combined with the prevailing warm, dry climate. Therefore, the low sludge accumulation rates from this pilot project, combined with efficient biogas production, would be an attractive option for domestic wastewater treatment in the Sahel. However, before developing this on a massive scale, an efficient setup for biogas collection and treatment should be designed and implemented for use in these anaerobic reactors.

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Chapter 7

7. Conclusion and perspectives

7.1 Overall conclusion

The design, implementation and evaluation of these two-stage high-rate anaerobic reactors, followed by a baffled pond with attached-growth or a wet-dry sand filter for domestic wastewater treatment in the sub-Saharan African, warm, dry climate of Ouagadougou were conducted and optimized. Based on the results obtained in this study, the two combinations have revealed an efficient and attractive alternative to treat domestic wastewater in West Africa, under a Sahelian climate. Great variability was observed in the effluent concentrations and in the removal efficiencies, considering all of the analysed treatment units. It was discovered that high pathogen removal efficiencies were achieved in both treatment options. The anaerobic reactors, followed by the sand filter, achieved a high nitrification rate, while the same reactors followed by the Baffled Pond with attached growth revealed better efficiency in removing ammonia nitrogen and *E. coli*. Furthermore, no *E.coli* were ever detected in effluent of the Baffled Pond, nor did clogging occur in the Sand Filter, during the entire study.

A detailed investigation was later carried out comparing the Baffled Pond (BP) with a control pond (CP) to understand this non-detection (< 1 per 100 ml) of *E. coli* in the effluent of the Baffled Pond. Among other things, this work showed the benefit of releasing effluent from the top layer of the pond, since *E. coli* concentrations were lower near the surface of each of the four compartments of the BP, with an undetectable level in the last compartment down to a depth of 0.60 m.

The actual mean hydraulic retention times for the BP and the CP were 4.1 and 3.2 days respectively. This implies that by introducing three vertical baffles in a pond the mean retention time was increased by approximately 22%. Therefore, these findings show that there is significant potential for size reduction and cost optimization to be achieved by the incorporation of properly designed baffles in ponds in tropical climates. Moreover, it was found that the Baffled Pond's volume was used more efficiently for wastewater treatment than the unbaffled pond, since more than half of the latter's volume was considered 'dead' or inactive. This tracer experiment showed that the outcome of the fluid flow pattern of these ponds can be fairly approximated as resulting of one and two mixed reactors in series with high dispersion for the Baffled and Control Ponds respectively. Consequently, both the Dispersed and Mixed-Reactors-in-Series Models are more appropriate for predicting the performance of the Baffled Pond. Because of the high tracer recovery from both ponds, one can conclude that loss of water through seepage is negligible and that sodium chloride can be considered as an inexpensive and conservative tracer.

The efficiency of removing *E. coli* was significantly different between the BP and the CP. This shows that the baffles with attached biofilm played an important role in the removal of *E.*

coli. In addition to the effluent being released from the top of the Baffled Pond, sedimentation and the synergetic effects of physical, chemical and environmental factors were responsible for the inactivation of *E. coli* in this system.

Another important aspect revealed by this research was the fact that the introduction of baffles with affixed plastic bottle caps in a pond can tremendously improve the ecology of algae, zooplankton, and other microscopic organisms, thus enhancing sanitation. Indeed, the results have shown dense and diverse biodiversity on both the attached media and in the water column of the Baffled Pond. The biofilm was thick and green on the upper parts of both sides of the baffles and the associated plastic caps.

Three major groups of diverse zooplankton were found in the water column at depths of 15-90 cm: Cladocera, Copepoda and Rotifers. The last group was dominant, with 14 identified species, which are attracted to a wide spectrum of natural food items. The microbial community was sampled for *E. coli*, fastidious bacteria, non-fastidious bacteria, enterobacteria, and fungi in the water column of the Baffled Pond. Strong correlations between certain groups of the bacteria and the density of suspended solids seem to correspond to the adsorption phenomenon of certain bacteria by suspended matter, as described in the literature. A Canonical Correspondence Analysis showed that the activity of algae, fungi and different types of bacteria varied considerably from one part of the Baffled Pond to another, probably in accordance with the variation of abiotic parameters, such as pH, dissolved oxygen, electrical conductivity, and temperature. In addition, the very good negative correlation between Group 1 and Group 2, confirms the predation relationship of the zooplankton at one side and the bacteria and algae on the other hand. As a result, the parasitic symbiosis distributions of phytoplankton and zooplankton have shown that the baffles had an effect on water quality which in turn has affected the ecology of the baffled pond. A Principal Components Analysis showed that the dense and abundant zooplankton community also varied considerably among the different zones of the pond and may play an important role in the control of the bacterial and algal populations found there.

Last but not least, an investigation of the potential of recovering biogas from the two-stage high-rate anaerobic reactors treating domestic wastewater under Sahelian climate revealed a high potential of producing this energetic resource. Over 9 liters per meter square per day of biogas were produced per 18 liters per gram of VSS removed, with a good methane content of 54%. Biogas would be a viable energy source in Sahelian countries, considering their advantageous climate. Biogas production rates were observed to closely vary with the temperature of the water inside the reactors. More importantly, very low sludge yields were recorded in both anaerobic reactors (0.0006 and 0.0002 m^3 per capita per year) and in the Baffled Pond (0.0014 m^3per capita per year). This low production of sludge could be due to the high biodegradability of the local domestic wastewater, combined with the prevailing warm, dry climate. Therefore, the low sludge accumulation rates from this pilot plant, combined with an efficient biogas reuse would be an attractive option for domestic wastewater treatment in the countries of the Sahel Region of Africa, especial since sludge handing would be minimized and can be done at low cost, taking into account the prevailing financial constraints.

Based on the outcome of this research, it was concluded that, both treatment options could be applied as alternative, low-cost wastewater treatment technologies for African cities and it is recommendable to use the effluent for restricted aquaculture and/or irrigation of peri-urban agriculture.

7.2 A prospectus for future research

In order to contribute to the consolidation and dissemination of these alternatives, low-cost two-stage high-rate anaerobic reactors, followed by a baffled pond with attached-growth or a wet-dry sand filter, for domestic wastewater treatment in sub-Saharan African cities merit further research into the resource recovery that may be applied to generate additional values for their promotion.

Recognizing the sensitive local perceptions towards the use of sanitation by-products due to existing cultural and religious beliefs, thorough evaluations of the users' perceptions and acceptability should be included in any further full-scale development of the technology, together with active campaigns of education and consciousness-raising. Although, it may not be easily acceptable within the local cultures, the effluent of this Baffled Pond could be used in aquaculture, given the abundant presence of highly nutritious food items for fish, including a great diversity of algae and zooplankton. Consequently, in order to contribute to the alleviation of hunger in low-income urban neighborhoods, further investigation for the use of the baffled pond effluent in aquaculture should be an attractive area of research.

Another great challenge of developing countries, especially in West Africa, is the development of affordable and renewable sources of energy. Due to increased energy consumption and rising prices, these alternative technologies are all the more important. Biogas production is especially convenient in hot climates, where high temperatures throughout the year contribute to efficient microbial processes. Nonetheless, before this is developed on a large scale, an efficient setup for biogas collection and treatment should be designed and implemented with these anaerobic reactors.

Even though these alternative technologies seem to be very profitable and low-cost, cost benefit analysis with a user-friendly prototype should be carried out before investments are made in such a venture.

By doing so, undoubtedly, a great achievement to the consolidation and dissemination of this technology for the treatment of domestic wastewater sub-Saharan Africa, and thus contribute to controlling the environmental challenges, such as deforestation, poor sanitation and disease.