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DYNAMICS OF WASTE ASSIMILATION IN A TROPICAL COASTAL WATER: A CASE STUDY OF THE OKRIKA CREEK IN SOUTH-EASTERN NIGERIA

Using the Streeter-Phelps method as modified for tidal regimes, the pollutant dispersal, re-aeration and waste stabilisation characteristics of the Okrika Creek in the coastal zone of Nigeria were evaluated as a basis for obtaining estimates of waste loading necessary to maintain ecologically acceptable levels of dissolved oxygen. The dispersion and deoxygenation coefficients varied from $0.4 \text{ km}^2 \cdot \text{day}^{-1}$ to $1.2 \text{ km}^2 \cdot \text{day}^{-1}$ and 0.3 day^{-1} to 0.25 day^{-1} , respectively, during the ebb and flood stages. Reaeration coefficient ranged from 0.64 day^{-1} to 0.46 day^{-1} during the corresponding flow stages. The dissolved oxygen profiles showed higher concentration levels during tidal current in-rush and early full flood stage. Much lower values were recorded during the late full flood and low ebb stages, indicating a dependence of oxygen content on tidal regime. Unionised ammonia, a major pollutant in the Creek associated with the nearby fertilizer production plant effluent, showed profiles having their peak concentrations at progressively further downstream locations over a number of isolated stages of tidal cycles. This suggests a dispersive-advective transport mechanism capable of moving pollutants oceanwards over a number of consecutive tidal cycles. The optimal BOD loading is $28 \text{ mg} \cdot \text{dm}^{-3}$, which is about 12% above the nationally recommended level. The implied higher tolerance of the Creek may be due to the strong dispersive-advective and reaeration effects. Timing of waste input with the tidal regime characteristics would provide for an improved pollutant dispersal, flushing and attenuation, and hence the sustained protection of the ecosystem.

1. INTRODUCTION

1.1. BACKGROUND

Balancing the immediate human needs and activities with the long-term ecological effects, particularly in the coastal zone, is a delicate and critical problem. Several methods have been suggested to address this problem. These include making optimum use of tidal dispersion and flushing to dilute and move pollutants oceanwards into the deep seas, reduction of influx wastes to within the assimilative capacity of ecosystem

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as dictated by its self-purification potential, reduction of important growth stimulation nutrients, use of closed-cycle systems, and balancing nutrient quality to achieve optimal productivity (NEMEROW [1]). Some of these approaches have been studied by NWANKWOR and OKPALA [2] in respect of an inland stream draining into the Atlantic coastal waters (using stream self-purification analysis), and O'BRIEN and GERE [3] for the tide dominated St. Jones River, New York, based on tidal dispersion and biochemical self-purification characteristics.

The Okrika Creek, in the mangrove swamp of Nigeria, constitutes one of such coastal waters where it is to be expected that incorporation of the above measures into the environmental management scheme could prove effective in pollution mitigation and prevention. The Creek receives effluent discharges from a fertilizer manufacturing plant as well as wastes of domestic origin. The swamp environment is fragile and shows a remarkable sensitivity to stress [4], [5], thus suggesting the need to maintain influx wastes to within the assimilation capacity of the Creek.

IPS [6] described the Creek and its main distributary, the Bonny River, as having high tidal volumes as well as high ebb and flood velocities. These tidal regimes would suggest moderate to high dispersion and flushing characteristics. Their analyses were however qualitative and therefore do not provide the basis for a quantitative evaluation of the tidal characteristics from which an optimal waste loading into the Creek can be determined. EZEALA [7] evaluated the impact of the fertilizer plant effluent on the Creek and associated groundwater. The concentration profiles of unionised ammonia (UJA), phosphate and urea in the creek water showed an apparent dependence on tidal stages. Based on this apparent dependence, further studies were undertaken on the pollutant level-tidal regime relation as a basis for evaluating the waste assimilation capacity of the Creek. Evaluation of the tidal dispersion and flushing, and hence the waste assimilation capacity of such a Creek is of fundamental engineering value. Their knowledge is important for the design of waste treatment plants by industries which desire to discharge their wastes into such surface water bodies. The purpose of this study is to evaluate the pollutant dispersion, flushing and organic waste stabilization characteristics of the Okrika Creek, as a basis for estimating the optimal waste loading into the Creek. The loading was estimated by applying the Streeter-Phelps analytical method as modified for estuaries to the watershed drainage data, tidal regime, dissolved oxygen (DO) and biochemical oxygen demand (BOD) profiles that were measured along the Creek course.

1.2. ENVIRONMENTAL SETTING

The Okrika Creek watershed covers an area of about 10 km². The Creek, which is about 70 m wide, flows approximately southeastward for about 4 km, and then joins the Bonny River en route to the Atlantic Ocean (figure 1). Drainage is of the deranged pattern characteristic of swamp areas. The Creek hydraulics is tidally influenced. Average monthly water levels at high and low tides as determined from the 1995 tide

records from the Bonny River are 2.0 and 0.5 m, respectively. The watershed is underlain by a shallow unconfined sand aquifer. Available rainfall records show that the average annual rainfall amount is about 3000 mm. Humidities are high (about 80%) particularly during the rainy season months of March to November.

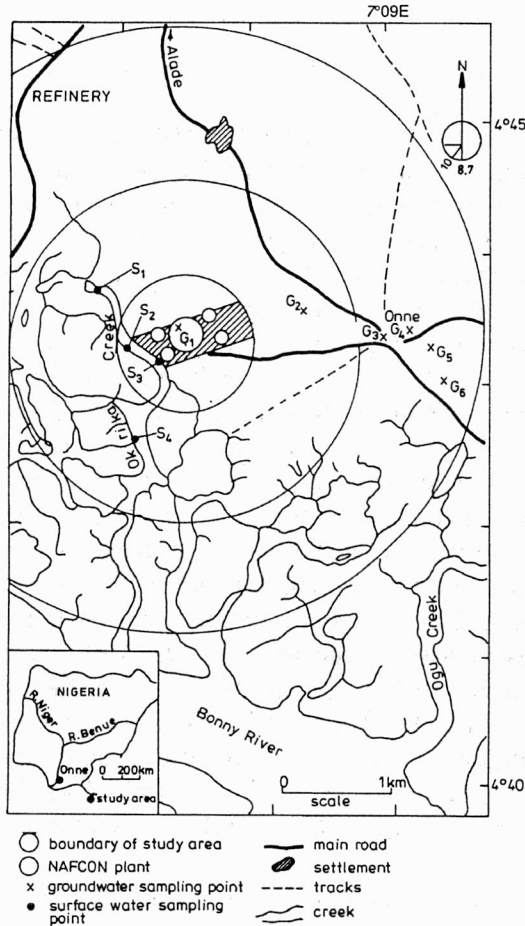


Fig. 1. Map of the study area showing sampling locations

2. METHODS OF STUDY

2.1. CREEK HYDRODYNAMICS

Aspects of the Creek hydrodynamics that are relevant for assessment of the dispersion and flushing processes include water flow velocity, water depth and patterns of freshwater inflow. Water flow velocity was estimated during low flood (tidal inflow)

and high flood/low ebb (quasi-static tidal stage) using a price current meter. No streams and rivers discharge into the Creek; freshwater inflow is therefore by surface runoff and groundwater discharge. Surface runoff contribution occurs only during the rainy season, and was estimated from the rational equation for runoff. Groundwater discharge was estimated from the static water level data using a modified form of the Darcy equation.

2.2. SAMPLING STRATEGY AND NETWORK

Water sampling network was based on the tidal regime, distribution of points of effluent waste inflow and the need to characterise the groundwater inflow chemistry. The Creek water sampling was thus carried out during the four tidal phases of low flood, high flood, high ebb and low ebb, and for three periods of time (August 30–31, September 27–28 and December 5, 1997). Sampling locations S_1 – S_4 (figure 1) were selected around the single major effluent inflow point, with S_1 located upstream of that point. At each location, samples were collected in sterilised bottles at four positions across the channel, and then composited. The composite samples were taken to the laboratory in ice-cooled containers for salinity, dissolved oxygen (DO), biochemical oxygen demand (BOD), unionised ammonia and pH measurements. Salinity (measured as electrical conductivity) profiles were important for tidal dispersion coefficient determinations, while the BOD and DO measurements were used in estimating the organic matter stabilisation characteristics of the Creek. Groundwater samples were also collected from wells G_1 – G_6 for DO analysis.

Unionised ammonia (UIA) is a major waste product of the fertilizer plant. Its stability is determined principally by the pH of the receiving water (HAMPSON [8]). Under conditions of moderate variations in pH of 6.7–7.3 in the Creek as measured in an earlier preliminary Creek survey (EZEALA [7]) it is to be expected that its total mass may not change considerably over an appreciable distance from an input point. Its attenuation would then be mainly by mixing and physical dispersion in the Creek water. Its profiles were then used in assessing the tidal dispersion and flushing effects.

2.3. TECHNIQUES OF CHEMICAL ANALYSES

Electrical conductivity was determined by the electrometric method using the YSI conductivity meter.

Creek water samples for DO determinations were collected in 250 cm³ amber bottles stoppered under water in order to exclude air bubbles. The samples were pretreated by the addition of 1 cm³ of concentrated sulphuric acid. The samples were then analysed for dissolved oxygen using the Winkler method [9].

The BOD determination was run according to the standard methods [9] procedure.

The UIA was determined using the Kjeldahl distillation method. Pretreated water samples were distilled into boric acid absorbent solution which then was titrated unto hydrochloric acid to a pink end point.

3. DATA ANALYSIS AND RESULTS

3.1. DISPERSION COEFFICIENT

Pollutant dispersion is effected by the forward and backward motion of the tidal Creek water. For a conservative pollutant, e.g. Cl^- , a measure of the dispersion is provided by the distance–electrical conductivity (salinity) profiles as hydrodynamically established during the flood and ebb tide flows (figure 2). For non-conservative spe-

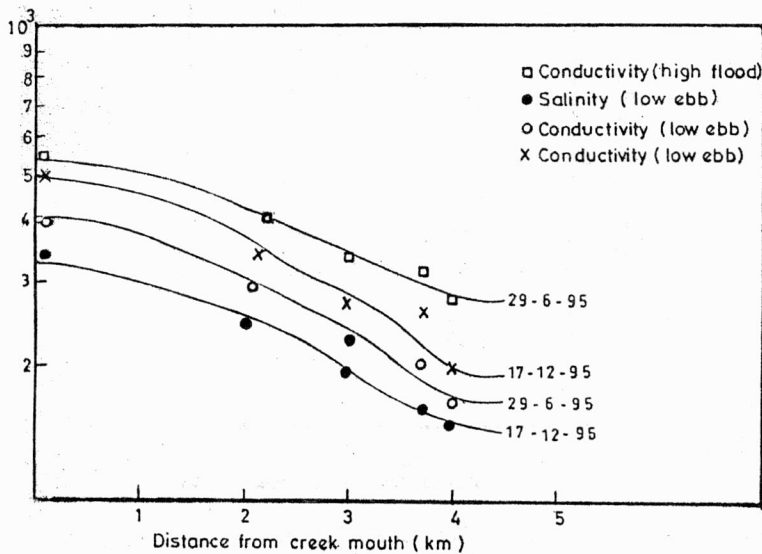


Fig. 2. Variations of electrical conductivity with distance from creek mouth

cies, the dispersion is effected by both the physical process of tidal flow and biochemical reactions. A measure of the dispersion is provided by the dispersion coefficient (E), and is estimated from the flow velocity data and conductivity profiles (figure 2) using the relation of NEMEROW [1]

$$E = \frac{U}{\frac{\ln c_1 c_2}{x_2 - x_1}}, \quad (1)$$

where U is the water flow velocity, c is the electrical conductivity, x is the distance from the mouth of the Creek, and subscripts 1 and 2 refer to the interval over which the conductivity gradient is to be determined. Relation (1) provides for dispersion coefficient values that range from $1.15 \text{ km}^2 \cdot \text{day}^{-1}$ at high flood to $0.41 \text{ km}^2 \cdot \text{day}^{-1}$ during low ebb, giving an average value of $0.78 \text{ km}^2 \cdot \text{day}^{-1}$.

3.2. BOD CREEK REACTION COEFFICIENT

The BOD values were plotted against distance upstream (i.e. from the Creekmouth towards the point of waste influx (figure 3)). As would be expected, the values generally increased upstream (i.e. decreased downstream towards the Creekmouth). The

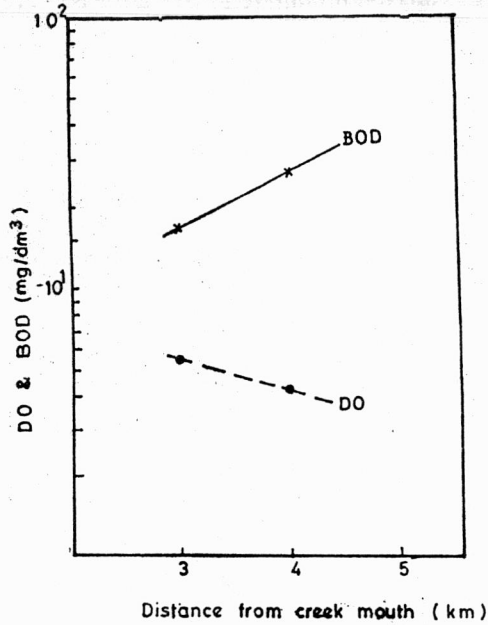


Fig. 3. BOD and DO profiles

decrease per unit distance offers a measure of the rate of organic matter decomposition and attenuation through biochemical and, to some extent, physical dispersive processes. The coefficient expressing this measure J_1 is :

$$J_1 = \frac{\ln \frac{c_2}{c_1}}{x_2 - x_1}. \quad (2)$$

Its value ranges from 0.75 to 0.98 day⁻¹.

3.3. CREEK DEOXYGENATION CONSTANT

Oxygen consumption in the Creek can be expressed in terms of J_1 and E as:

$$J_1 = \frac{U}{2E} \left[1 - \sqrt{1 + \frac{4K_d E}{U^2}} \right], \quad (3)$$

where K_d is the Creek deoxygenation constant, an index of DO consumption per unit time. The estimated values range from 0.25 to 0.3 day⁻¹.

3.4. REAERATION CHARACTERISTICS

The reoxygenation constant K_2 was determined from relation (4) as:

$$K_2 = \frac{(D_1 U)^{1/2}}{H^{3/2}}, \quad (4)$$

where D_1 is the oxygen diffusivity constant at 20 °C and H is the water depth at a given tidal level. For the measured H values of 0.5 m and 2.2 m at low and high tides, respectively, relation (4) gives K_2 values of 0.64 day⁻¹ and 0.46 day⁻¹. Using these K_2 values, the DO Creek reaction coefficient J_2 , a measure of DO replenishment per unit distance along the Creek, was estimated to be between -0.85 km⁻¹ and -0.49 km⁻¹ from equation (5):

$$J_2 = \frac{U}{2E} \left(1 - \frac{\sqrt{1 + 4K_2 E}}{U^2} \right). \quad (5)$$

3.5. DISSOLVED OXYGEN AND UNIONIZED AMMONIA PROFILES

Figures 4 and 5 present the DO and UIA profiles at the four stages of tidal cycle based on the sampled locations. Dissolved oxygen has generally higher concentration profile values (5.0–6.2 mg·dm⁻³) during the low flood period than other subsequent stages. Unionised ammonia, on the other hand, shows higher concentration profile

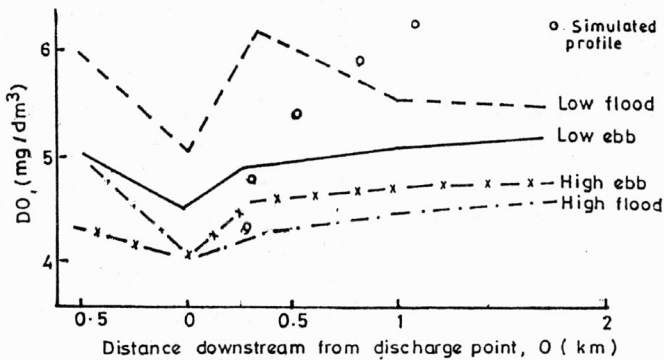


Fig. 4. Variations of DO with distance downstream at various tidal stages (0 is the point of waste influx)

values (0.1–5.6 mg·dm⁻³) during high flood. The profiles during subsequent tidal stages show generally lower concentrations and with peak values occurring at pro-

gressively further downstream locations. The pH varies over a narrow range of 6.2 to 7.8, while temperature was between 27 and 28 °C over all the tidal phases.

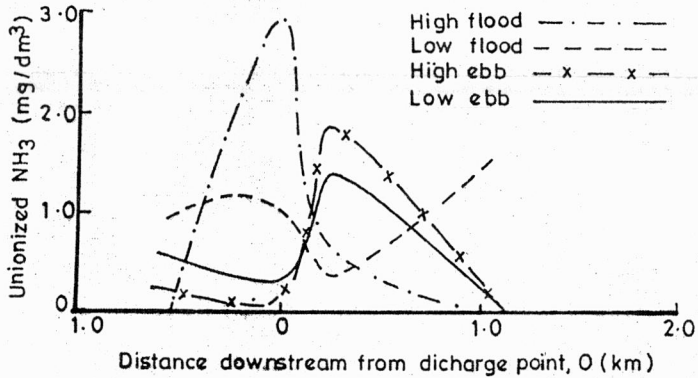


Fig. 5. UIA profiles at various tidal stages (0 is the point of waste influx)

4. DISCUSSION

4.1. DISSOLVED OXYGEN AND UNIONISED AMMONIA DISTRIBUTION PATTERNS

It is evident from the profiles in figure 4 that the DO distribution pattern is tide-dependent. The high DO content during the low flood stage may be attributed to the seawater current rush into the Creek. The turbulent nature of the incoming seawater allows it to entrap atmospheric oxygen as it moves along its pathway. It also enhances reaeration and results in elevated levels of DO concentration. Turbulence effect is an important mechanism of aeration in sewage treatment plants [10]. Furthermore, the fresh input of seawater from the ocean has not been exposed to the contaminating influence of effluents. The DO concentration is lower during the high flood stage for which water samples were collected about two hours after the cessation of tidal inflow. This can be accounted for by a decline in reaeration level due to quasi-static flow condition, increased water depth and deoxygenation processes. Dissolved oxygen concentration however increases again during the high ebb phases when flow becomes more turbulent relative to the high flood stage. The DO level however falls short of the low flood values because of the lack of fresh input of water as was the case during that (low flood) stage.

These fluctuating conditions of flow result in the Fairs number (a measure of the predominance of reaeration over deoxygenation) ranging from as low as 0.4 during the low ebb/high flood to as high as 5.4 for the low flood/high ebb stages. The low values compare well with that (0.2) obtained by NWANKWOR and OKPALA [2] for the sluggish stretch of River Nworie in the upper coastal area of Nigeria as well as with the value of 0.5 that was determined by O'BRIEN and GERE [3] in respect of the tide-

dominated St. Jones River, Delaware, USA. Similarly, the higher Fairs number is well within the range, though somewhat higher than the value of 4.5 determined for the fast flowing segments of River Nworie. The river flow is not characterised by turbulence. The profile of the Fairs number gives the indication that during tidal inflow (low flood) and recession (high ebb), the Creek behaves as a fast flowing river with high self-purification potential. During high flood–low ebb intervals both of which exhibit static flow conditions, self-purification is relatively low, and dissolved oxygen level remains about or even less than the critical threshold of 4.0–4.5 mg·dm⁻³.

The DO profile was simulated using the relation:

$$D = \frac{K_D L_0}{K_2 - K_d} (e^{-J_1 x} - e^{-J_2 x}) + D_0 e^{-J_2 x}, \quad (6)$$

where D is the DO deficit at any point X along the Creek course, D_0 is the deficit at $X = 0$ m and L_0 is the BOD measured at $X = 0$ m. The DO concentration is estimated as 8.0 mg·dm⁻³ (i.e. the theoretical DO saturation value at the measured temperature) less than the calculated D value. The simulated DO profile is qualitatively consistent with the measured values in that they all have a common DO maximum sag point, and thereafter show a general trend towards increasing DO levels with increasing downstream distance. The simulated profile, however, exhibits a higher rate of increase in dissolved oxygen causing significant deviations from the measured value. The observed lack of agreement between the simulated and measured values, particularly during the high ebb–flood stages in the downstream areas, could be as a result of changing dynamics of flow during the various tidal stages which are not completely accounted for in the simulation. Furthermore, uncertainties possibly associated with sample collection under tidal turbulence as well as sporadicity in effluent input pattern at source would also have the potential to cause the observed deviation effect (see figure 4). Sources of the dissolved oxygen include atmosphere, fresh in-rushing tidal water, discharging groundwater (5.0 to 6.5 mg·dm⁻³) and from algal photosynthetic activity [2].

4.2. POLLUTANT DISPERSAL AND ATTENUATION

Using the static high flood as a starting point of a tidal cycle, the profiles in figure 5 indicate that peak concentrations of UIA occur at progressively further downstream locations during later tidal stages. Urea and PO₄ also show concentration profiles qualitatively similar to that of UIA. Despite the fact that the profiles were not obtained during consecutive stages of one tidal cycle, and the effluent input was non-steady, an inference can be made on the potential for dispersive-advective transport during a tidal cycle. The progressively lower peak concentration values observed at downstream locations taken over the tidal stages are most probably the effect of the tide-aided dispersive-advective process, dilution and biochemical transformation that

occur as the pollutant slugs encounter larger and relatively fresh bodies of water. Based on the available data, it is to be expected that within one tidal cycle, the UIA will undergo transportation in a manner consistent with the above profiles. Over a large number of cycles, an opportunity will be created for a net seaward movement and attenuation of effluents. Such an opportunity is manifested by the apparent oceanward displacement of the peak concentration positions as observed in figure 5.

4.2.1. FLUSHING POTENTIAL

Channel flushing as a mechanism of pollutant attenuation would be effected by periodic input of pollutant-free water (freshwater) into a waterway. Such freshwater input occurs as tidal inflow, groundwater influx and surface runoff. The groundwater inflow magnitude is about $400 \text{ m}^3 \cdot \text{hr}^{-1}$, while surface runoff as computed using the rational equation is $230 \text{ m}^3 \cdot \text{hr}^{-1}$. The latter two components constitute approximately 30% of low tide discharge, particularly during the rainy season. The three discharge components potentially would provide the freshwater input needed to enhance flushing effect.

4.3. CRITICAL BOD LOADING

The critical BOD loading was estimated as the input effluent strength which when exceeded would cause ecologically unacceptable levels of dissolved oxygen. DO deficit at the point of maximum (DO) sag was evaluated for several assumed initial Creek BOD loading (L_0) as a basis for obtaining the L_0 that yielded the maximum but allowable deficit, particularly during the static high flood/low ebb period. The optimum L_0 was then used to determine the critical BOD loading taking into consideration the effluent discharge rate.

The table shows the values L_0 as well as the corresponding critical distances X_c (the distance from an effluent input location to the point of maximum DO sag) and the DO deficit (D) as calculated using relations (6) and (7), respectively:

$$X_c = \frac{1}{J_1 - J_2} \ln \left\{ \frac{J_2}{J_1} \left[1 - \frac{D_0(k_2 - k_d)}{L_0 k_d} \right] \right\}. \quad (7)$$

D_0 is the initial DO deficit with values of 3.8 to 1.8 $\text{mg} \cdot \text{dm}^{-3}$ based on the low ebb/high flood data. In eqn. (6), X_c substitutes for X .

For the range of L_0 values, $L_0 = 9 \text{ mg} \cdot \text{dm}^{-3}$ produced the maximum but allowable DO deficit of 4.0–4.5 $\text{mg} \cdot \text{dm}^{-3}$. If the upstream BOD is 2 $\text{mg} \cdot \text{dm}^{-3}$, a value within the range of 1–8 $\text{mg} \cdot \text{dm}^{-3}$ for moderately contaminated streams, it can be shown that the allowable BOD is 4.7 $\text{mg} \cdot \text{dm}^{-3}$ at the plant effluent discharge rate of $1.5 \cdot 10^3 \text{ m}^3 \cdot \text{day}^{-1}$. This amounts to allowable raw effluent BOD of 24 $\text{mg} \cdot \text{dm}^{-3}$ at the average Creek flow

Table

BOD loading (L_0) and the corresponding critical distance (X_c) and dissolved oxygen deficit (D)

BOD loading concentration L_0 ($\text{mg}\cdot\text{dm}^{-3}$)	X_c (km)	D ($\text{mg}\cdot\text{dm}^{-3}$)
5	2.5	1.6
7	1.6	2.5
9	0.5	4.5
15	0.1	6.2

rate of $1.2\cdot 10^5 \text{ m}^3\cdot\text{day}^{-1}$. It can further be shown that a raw effluent BOD as high as $28 \text{ mg}\cdot\text{dm}^{-3}$ would be assimilated during conditions of low flood (tidal inrush) and high ebb (current outflow) when reaeration effects are much more dominant. The estimated BOD load of $24\text{--}28 \text{ mg}\cdot\text{dm}^{-3}$ is somewhat higher than the nationally recommended limit of $25 \text{ mg}\cdot\text{dm}^{-3}$. This may be due to the strong dispersive and reaeration effects.

5. CONCLUSIONS

Tide-aided dispersion–advection process, flushing and dilution effects constitute the major mechanisms of attenuation and oceanward transport of pollutants in the Okrika Creek. The optimal waste load is $24 \text{ mg}\cdot\text{dm}^{-3}$ and can be increased to $28 \text{ mg}\cdot\text{dm}^{-3}$ under low flood–high ebb conditions when reaeration and mechanical mixing effects are measurably higher.

Timing of waste input with the tidal phases and regime characteristics would provide for an improved pollutant dispersal, flushing and attenuation, and hence the sustained protection of the ecosystem.

ACKNOWLEDGEMENTS

The authors wish to acknowledge the financial support of the Federal University of Technology, Owerri, Nigeria, under its Research Grant Scheme. The computing facilities provided by Dr. O.E. Osuagwu, the Centre for Computer Engineering, Owerri, are gratefully acknowledged.

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DYNAMIKA ASYMLACJI ŚCIEKÓW W PRZYBRZEŻNYCH WODACH TROPICALNYCH: PRZYPADK ZATOKI OKRIKA W POŁUDNIOWO-WSCHODNIEJ NIGERII

Korzystając ze zmodyfikowanej dla pływu metody Streetera-Phelpsa, oceniono rozproszenie polutanta, napowietrzanie naturalne i stabilizację ścieków w Zatoce Okrika w strefie przybrzeżnej Nigerii. Otrzymane wyniki stanowiły podstawę oszacowania ładunku ścieków, dla którego można utrzymać ekologicznie dopuszczalny poziom rozpuszczonego tlenu. Współczynnik rozproszenia wynosił od 0,4 do 1,2 km²/dzień a współczynnik zużycia tlenu – od 0,3 do 0,25 1/dzień podczas odpływów i przypływów. Podczas odpowiadających im przepływów współczynnik napowietrzania naturalnego mieścił się w zakresie 0,64–0,46 1/dzień. Profile rozpuszczonego tlenu pokazują, że ma on większe stężenie podczas pływu i wczesnych stanów całkowitego zalewu. Znacznie niższe stężenia stwierdzano podczas późnego całkowitego zalewu i małych odpływów, co świadczy o zależności stężenia tlenu od pływu. Profile niedysocjowanego amoniaku, który jest głównym zanieczyszczeniem zatoki związanym ze ściekami z pobliskiej fabryki nawozów sztucznych, mają szczytowe stężenia związane z miejscami położonymi z biegiem rzeki podczas pewnych zamkniętych cykli pływu. Można więc sądzić, że dzięki dyspersyjno-adwekcyjnemu transportowi zanieczyszczenia są prawdopodobnie niesione do oceanu podczas następujących po sobie cykli pływu. Optymalna wartość BOD wynosi 28 mg·dm⁻³, o około 12% przekracza zalecany poziom. Ta większa tolerancja zatoki może być spowodowana przez silne dyspersyjno-adwekcyjne efekty i naturalne napowietrzanie. Zsynchronizowanie w czasie zrzutu ścieków fazy pływu zagwarantowałoby lepszą dyspersję polutanta, jego przepłukanie i osłabienie, a w rezultacie długotrwałą ochronę ekosystemu.

dynamika
asymlacji ścieków
wody przybrzeżne
wody tropikalne
Zatoka Okrika
Nigeria
metoda Streetera-Phelpsa