Preface

This is the third volume of Journal of Hydrology and Environment Research (JHER). The JHER is a peer reviewed international journal, which publishes high quality research papers in the fields of hydrology, water and environment. The JHER is published from Sydney, Australia.

This volume of JHER contains 6 technical papers, one review paper and one technical note. The 1st paper focuses on water balance validation of a temperature-dependent parameter value of the Priestley-Taylor equation of evaporation. The 2nd paper presents the production and characterization of low-tech activated carbon from coconut shell. The 3rd paper examines the changes and geomorphology in the coastal region of Tiswadi Taluka of Goa state in India. The 4th paper presents a comparison between upflow anaerobic sludge blanket and anaerobic filter for treating the wheat straw washwater. The 5th paper discusses about the bio-chemical separation and purification of heavy metal from industrial waste water. The 6th paper presents an analysis of groundwater quality in communities surrounding the Dompoase Landfill, Kumasi, Ghana. The 7th paper presents the impacts of sea level rise in Bangladesh. The 8th paper is a technical note presenting the comments on “Revised coefficients for Priestley-Taylor and Makkink-Hansen equations for estimating daily reference evapotranspiration” by N.C. Cristea, S.K. Kampf and S.J. Burges.

We acknowledge greatly the reviewers who have spent their valuable time in reviewing the papers contained in this volume. We thank Md Mahmudul Haque for editorial coordination. We also thank Mr Tauqir Ullah and Mr Imran Rahman for assistance with the cover page design and website maintenance. We welcome papers for 4th volume which will be published in mid-2016. Authors are advised to visit website of JHER (www.jher.org) for information on preparation and submission of manuscripts for possible publication in this journal.

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# Table of Contents

**Technical papers:**

- Water Balance Validation of a Temperature-Dependent Parameter Value of the Priestley-Taylor Equation of Evapotranspiration
  
  Jozsef Szilagyi

- Production and Characterisation of Low-tech Activated Carbon from Coconut Shell
  

- Shoreline Changes Analysis and Coastal Geomorphology of Tiswadi Taluka of Goa State in India
  
  Kuldeep Pareta and Upasana Pareta

- Comparison of an Upflow Anaerobic Sludge Blanket and an Anaerobic Filter for Treating Wheat Straw Washwater
  
  Syazwani Idrus, Charles J. Banks, Sonia Heaven

- Bio-Chemical Separations and Purification of Heavy Metal from Industrial Waste

- Water: A Review on Adsorption and Precipitations
  
  Kemal Mohammed, Kidist Worku, Omprakash Sahu

- Potential Migration of Leachate from an Active Landfill: Spatial Analysis of Groundwater Quality in Communities Surrounding the Dompooase Landfill, Kumasi, Ghana
  
  A Sulemana, P Antwi-Agyei, J N Hogarh

**Review Paper:**

- Impact of Sea Level Rise in Bangladesh: A Socio Engineering Overview
  
  Sabrina Ali and Faruk Kader

**Technical Note:**

  
  Jozsef Szilagyi

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Water Balance Validation of a Temperature-Dependent Parameter Value of the Priestley-Taylor Equation of Evapotranspiration

Jozsef Szilagyi

Abstract: New research suggests that the Priestley-Taylor (PT) parameter, α, displays a temperature dependency with a significant increase in its generally accepted constant value of 1.26 as air temperature (T) drops. Multi-year water-balance data from two humid, cool climate experimental watersheds—Hubbard Brook in New Hampshire and Lookout Creek in Oregon, USA—and from a warm, humid catchment at the border of Mississippi and Louisiana as a control, support a temperature-dependent α(T) value. Mean annual evapotranspiration rates obtained with α(T) were superior to those of α=1.26 and fell within 3% of the water-balance derived (precipitation minus runoff) values. As a consequence, humid, cold-region evaporation/evapotranspiration may be significantly underestimated via the PT equation when the classical value of 1.26 is employed in place of a temperature-modulated α(T).

Keywords: Evapotranspiration, evaporation, Priestley-Taylor equation, Priestley-Taylor parameter.

1. Introduction

The Priestley-Taylor (PT) equation (1972) describes the evapotranspiration (ET) rate of a wet environment when water availability at the surface is not limiting

\[ ET = \alpha \frac{\Delta}{\Delta + \gamma} R_n \]

Here \( R_n \) is the available energy (i.e., net radiation) at the wet surface, specified in water depth per unit time (e.g., mm d\(^{-1}\)), \( \Delta \) is the slope of the saturation vapor pressure curve at the air temperature (T), and \( \gamma = c_p \rho / (0.622 L) \) is the psychrometric constant, where \( c_p \) is the specific heat of air at constant pressure (p) and \( L \) is latent heat of vaporization for water. The coefficient \( \alpha \) is generally accepted to express the evaporation-enhancing effect of large-scale entrainment of drier free-tropospheric air resulting from the growing daytime convective boundary layer (Brutsaert, 1982; deBruin, 1983; Culf, 1994; Lhomme, 1997; Heerwaarden et al., 2009). Employing warm season (11 °C < T < 33 °C) data from both hemispheres, Priestley and Taylor (1972) found that \( \alpha \) assumes a value of about 1.26, which has been used ever since in the literature as the standard value of the PT parameter.

Recently Szilagyi et al. (2014), employing ERA-Interim global reanalysis data (http://www.ecmwf.int/products/), found that the value of \( \alpha \) changes with temperature (T), and specified the best-fit polynomial (Figure 1) as presented in equation 2.

\[ \alpha(T) = -3.89 \cdot 10^{-6} T^3 + 4.78 \cdot 10^{-4} T^2 - 2.54 \cdot 10^{-2} T + 1.64 \]

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Equation 2 is valid in the 0 – 30 °C range and yields an $\alpha$ value of 1.2 at 30 °C, and 1.64 at 0 °C, which is a significant difference.

The above equation has not been validated in cold/cool climates by comparing PT-derived ET rates using the classical $\alpha = 1.26$ value or the current $\alpha(T)$ relationship. Therefore water balance data of two humid, cool-climate experimental watersheds plus those of a warm, humid catchment (as control) are to be employed and the multi-year water-balance derived ET rate as the difference in precipitation ($P$) and runoff ($Q$) compared to the PT-predicted mean annual ET fluxes.

2. Description of the watersheds

Out of the many experimental watersheds with detailed hydrological and multi-site meteorological data, maintained either by the Agricultural Research Service of the US Department of Agriculture (ars.usd.gov) or the Long Term Ecological Research Network (www.lternet.edu), only two were found (i.e., Hubbard Brook and Lookout Creek) situated in a humid, cool climate that had the necessary multi-year data of air and dew-point temperature, solar radiation, precipitation and stream discharge (Figure 2).

The Hubbard Brook experimental watershed (31.6 km$^2$) within the White Mountain National Forest in New Hampshire is made up of several forested sub-catchments for which multi-year, accurate daily meteorological measurements are available, however runoff is not measured from the main watershed itself. A common time-period for which all required data were available for altogether eight sub-catchments (at altitudes between 442 and 905 m) was found to be May 1, 1983 till April 30, 2009. The eight sub-catchments are predominantly exposed either to the south or north. While precipitation has been measured at several locations adjacent to the sub-catchments, solar radiation is measured only at one central location resulting in only one multi-year mean annual PT ET estimate. The watershed enjoys about 1400 mm precipitation a year, evenly distributed among the months. The winter periods (when freezing may occur) are long and cold. January averages are about -9 °C, but long periods of -12 °C to -18 °C are common. Occasional midwinter thaws can result in elevated streamflow. The average July temperature is ~18 °C during the short and cool summers.

Lookout Creek (drainage area of 64 km$^2$) is found in the Andrews Experimental Forest of the western Cascade Range of Oregon at elevations from 410 to 1630 m. Streamflow is measured at several sub-catchments, most of them occupying only a usually steep mountain slope, beside the main outlet. Both precipitation and other meteorological variables (including solar radiation) are measured at several locations (six for temperatures and four for solar radiation) within the basin, but typically at valley locations which makes obtaining accurate radiation balances for the steep slopes difficult. Within the Lookout Creek watershed one larger sub-catchment with monitored flow exists, called Mack Creek with a drainage area of 5.81 km$^2$.

The maritime climate has wet, mild winters and drier, cool summers. At the main meteorological station at 430 m elevation, mean monthly temperatures range from 1°C in January to 18 °C in July. Precipitation falls primarily from November through March, and varies with elevation.
Water Balance Validation of a Parameter of the Priestley-Taylor Equation

Szilagyi J.

Journal of Hydrology and Environment Research

from 2300 to over 3550 mm at higher elevations. In the wintertime, rain is mixed with snow in the lower portion of Lookout Basin and snow is more persistent at higher elevations (above 1000 m). The time-period with the most abundant data falls between October 1, 1998 and September 30, 2011. See www.lternet.edu for more detailed information of the two experimental catchments.

Figure 2 Location of the watersheds employed in the study. HB: Hubbard Brook in New Hampshire; LC: Lookout Creek in Oregon, and; AR: Amite River near Denham Springs in Louisiana

As a control case, a third, warm-climate humid watershed (shared by Mississippi and Louisiana), minimally affected by human alterations, the Amite River near Denham Springs, LA (drainage area of 3,315 km$^2$ with a mean elevation of ~100 m), was also selected, for analysis of its 30-year runoff data together with the similarly long Solar and Meteorological Surface Observation Network (SAMSON) data set from nearby Baton Rouge, LA. The watershed enjoys about 1500 mm precipitation a year, more or less evenly distributed among the months. Winter periods are short and mild with January averages of 10 °C, while July temperature is around 27 °C during the long and hot summers.

3. Estimation of mean annual evapotranspiration rates

The Hubbard Brook website specifies mean annual ET as about 500 mm, while the Andrews Forest LTER site does not disclose it. Since at Hubbard Brook streamflow is not measured at the main outlet, watershed-representative water balance ET ($ET_{wb}$) could only be obtained as the arithmetic mean of the eight $P - Q$ sub-catchment values (Table 1). The so-obtained 493 mm annual ET rate is very close to what the site specifies. Note that the long period of about 25 years allows for not using hydrologic years (i.e., from October till November), as performed for Lookout Creek where the time-period is about half of what is available at Hubbard Brook. Due to the somewhat milder climate and more abundant precipitation, mean annual water balance ET rates became 632 mm for Mack Creek and almost the same, 638 mm for Lookout Creek, which encompasses the former. The presence of deep valleys between steep slopes largely hinders the estimation of individual sub-catchment ET rates either with the water-balance approach or the PT equation. This is so because neither the closest valley bottom precipitation nor the similarly situated solar radiation measurements are representative of these steep, long slopes that contain the small sub-catchments. The meteorological measurements, especially when obtained from multiple locations, are more representative of larger, more complex catchments that comprise of many slopes with different length, steepness and aspect, and also significant valley segments, such as found at Mack Creek and at the larger Lookout Creek.

Application of the PT equation requires the net radiation term, $R_n$. The estimation method (called WREVAP) of Morton et al. (1985) was employed for this task at a monthly time-step since daily $R_n$ estimates are typically laden with large uncertainties. The estimates become significantly improved for time periods of five days or longer (Morton et al., 1985). A recent study by McMahon et al. (2013a, b) found the WREVAP program the most reliable practical ET estimation tool for monthly or annual time-scales, therefore the software’s net radiation outputs were also applied in the present study. Table 1 includes the WREVAP ET values ($ET_{M}$), as a by-product as well. In the winter months with near freezing-point temperatures the estimated $R_n$ values often become negative. Equation 1 in such months would yield negative ET rates therefore ET was set to zero when this happened.
In the lack of pressure values, \( p \) in the \( \gamma \) term of equation 1 was estimated by the barometric formula as

\[
p = 1013 \left( \frac{T + 273.16}{T + 273.16 + \lambda z} \right)^{5.26},
\]

where \( g \) is gravitational acceleration, \( R \), the specific gas constant of air, \( \lambda \) the dry adiabatic lapse rate (0.0065 Km\(^{-1}\)), and \( z \) the altitude. Equation 3 is not sensitive to the air temperature \( T \), therefore it was applied with a constant \( T = 15 \) °C for all watersheds, resulting in \( \gamma \approx 0.6 \) for Hubbard Brook \((z \approx 700 \) m\), in 0.58 for Lookout Creek \((z \approx 1100 \) m\) and in 0.65 for the Amite River \((z \approx 100 \) m\).

Szilagyi and Jozsa (2008), Szilagyi et al. (2009), and Szilagyi (2014) proposed the evaluation of \( \Delta \) at the wet-environment air temperature \((T_w)\) estimated by a recursive algorithm in place of the actual air temperature of equation 1. This is not necessary for the current watersheds due to their deliberate selection for humidity (i.e., for the applicability of the PT equation), since under such conditions actual air temperature, \( T \) is close \( T_w \).

**Table 1** Mean annual ET estimates by method \((ET_{wb} — \) water balance; \( ET_{\alpha(T)} — \) equations 1 and 2; \( ET_{\alpha=1.26} — \) equation 1; \( ET_M — \) WREVAP model) and location

<table>
<thead>
<tr>
<th>Location</th>
<th>Hubbard Brook, NH (05-01-85 – 12-31-09)</th>
<th>Lookout Creek, OR (10-01-98 – 9-30-11)</th>
<th>Amite River near Denham Springs, LA (01-01-61 – 12-31-90)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean of eight sub-catchments</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>( ET_{wb} ) (mm yr(^{-1}))</td>
<td>493</td>
<td>632</td>
<td>638</td>
</tr>
<tr>
<td>( ET_{\alpha(T)} ) (mm yr(^{-1}))</td>
<td>481</td>
<td>606</td>
<td>626</td>
</tr>
<tr>
<td>( ET_{\alpha=1.26} ) (mm yr(^{-1}))</td>
<td>438</td>
<td>546</td>
<td>540</td>
</tr>
<tr>
<td>( ET_M ) (mm yr(^{-1}))</td>
<td>461</td>
<td>490</td>
<td>569</td>
</tr>
</tbody>
</table>

4. Discussion and conclusions

Equation 1 with the classical constant \( \alpha \) value of 1.26 significantly underestimates both cool-climate water-balance derived mean annual ET rates by about 10% for Hubbard Brook and by 15% for Mack and Lookout Creeks. The underestimation greatly improves with the application of the temperature-dependent parameter value, to less than 3% for Hubbard Brook and Lookout Creek and to 4% for Mack Creek. These values are well within the combined accuracy of the precipitation and runoff measurements. WREVAP yielded ET estimates with accuracies intermediate of the two versions of equation 1 for Hubbard Brook and Lookout Creek, and gave the worst performance of all for Mack Creek. WREVAP, due to the application of the complementary relationship of evaporation (Bouchet, 1963) to predict actual ET rates, is sensitive to the vapor pressure deficit value. The low number (only two) of meteorological stations applied over a highly variable topography, neither within the boundary of Mack Creek, may explain this large error.

Table 1 illustrates well the importance of employing a temperature-dependent formulation of the Priestley-Taylor coefficient, \( \alpha \), when applied with low temperatures, since under such conditions the resulting Priestley-Taylor ET rates may differ by more than 20% (i.e. \( 1 - 1.26 / 1.6 \)) from those obtained by the classical constant value, leading to a substantial underestimation of the wet-environment latent heat fluxes in cold regions in the latter case.
For warm and humid climates the classical 1.26 value of $\alpha$ yields practically the same ET rates as the $\alpha(T)$ case (1128 vs 1134 mm yr$^{-1}$) due to a) the milder slope of the $\alpha(T)$ polynomial (equation 2) for warmer air temperatures (Figure 1) and, b) the fact the 1.26 value is more or less an average value of $\alpha(T)$ for temperatures between 15 and 30 °C, the predominant temperature range for the Amite River catchment. Both ET estimates are about 11% higher than the water-balance derived value of about 1000 mm yr$^{-1}$. Due to the high temperatures in summer and early fall, water may occasionally become a limiting factor for the very intensive ET rate specified by equation 1, reflected in the slight overestimation of the water-balance derived value. The Morton ET method, due to its complementary relationship, detects such water supply shortages via the vapor pressure deficit and yields a reduced ET rate, although somewhat smaller (by 6%) than the actual water-balance value. These latter results also indicate that the significant underestimation of ET by the PT equation when $\alpha = 1.26$ for the cool-climate watersheds is not the result of an underestimated net radiation term, $R_n$ of equation 1, since then the same underestimation should have been expected for this warm-climate catchment.

Based on the above, a temperature-modulated version of the Priestley-Taylor coefficient, $\alpha$, as specified by equation 2, is always recommended over the classical constant value of 1.26 when applying the PT equation for practical estimates of the wet-environment evapotranspiration rates, especially for cool or cold climates.

5. Acknowledgements

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Production and Characterisation of Low-tech Activated Carbon from Coconut Shell

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Abstract: In this paper, prospect of employing a low-tech preparation of coconut shell as activated carbon was investigated. Activated carbon was prepared from locally available coconut shell with simple, inexpensive and low-tech production process. The influence of carbonisation temperatures (i.e. 500, 600, 700 and 800°C) and residence times (i.e. 1.5 and 2 hours) of the prepared activated carbon on the percentage of yield, adsorption capacity and the development of pores were examined. The charcoal was ground to particles size ranges of 0.6-1.0, 1.0-2.36 and 2.36-4.75 mm, respectively and was activated by saturated NaCl solution for 24 hours. Results show that the carbon yield reduces with both increasing temperature and residence time, but the adsorption capacity increases up to an optimum temperature of 700°C. The pore diameter was progressively increased from the carbonisation temperatures 500 to 700°C and stabilised even at higher temperature. Increasing residence time to two hours obviously increased the pore diameter but the structural pore walls started to collapse at 700°C. It is further found that 1.0-2.36 mm particle sized activated carbon has the best adsorption capacity of 58% compared with the other particle size range. The optimum condition obtained is at carbonisation temperature of 700°C and 1.5 hours residence time. The results of the study demonstrate that the low tech process of self-prepared coconut-shell activated carbon is feasible and may be utilised for water treatment.

Keywords: Activated carbon, coconut shell, pore development, low-tech preparation.

1. Introduction

Activated carbon (AC) is considered as universal adsorbent for water treatment and is commonly used for decolourization and detoxification of liquids due to its porosity and large specific surface area (Bansal and Goyal, 2010). AC from agricultural by-products and natural biomass waste has received considerable interest as these are cheaper, renewable and abundantly available. Numerous agricultural wastes have been successfully experimented and produced effective AC such as oil-palm stone (Guo and Lua, 1999), olive-waste cakes (Bacaoui et al., 2001), pistachio-nut shell (Lua and Yang, 2004), olive-seed (Stavropoulos and Zabaniotou, 2005), corn cob (Cao et al., 2006), cassava peel (Sudaryanto et al., 2006), cherry stones (Olivares-Marin et al., 2006), rubber wood saw dust (Prakash et al., 2006), jute and coconut fibre (Phan et al., 2006), apricot stones (Şentorun-Shalaby et al., 2006), sugar beet bagasse (Onal et al., 2007), palm shell (Adinata et al., 2007), sunflower oil cake (Karagöz et al., 2008), sugar cane bagasse and rice husk (Kalderis et al., 2008) and coconut husk (Tan et al., 2008). The application of these residual waste is seen as more economic, sustainable and conserving the environment by minimizing the use of chemicals, in particular for water treatment.

In industrial practices, the most common sources for the production of ACs are coal and coconut shells. However, these readily available manufactured AC are rather expensive and provide limited access for the communities in rural areas. This study attempts to explore the prospect of employing a low-tech preparation of coconut shell as AC. In tropical climate countries, such as Malaysia, coconut is abundantly

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available and thus provides unlimited sources. The coconut has three layers where the outermost layer is typically smooth with a greenish colour is called the exocarp. Inside this, the fibrous husk or mesocarp surrounds the hard woody layer called the endocarp, which is commonly known as the coconut shell. Both the coconut husk and shell are rich in carbon content and have been previously investigated as potential source for activated carbon (Tan et al., 2008). Researchers found out that coconut husk is more efficiently carbonised than coconut shell. However the coconut husk AC is not suitable as filter medium due to fine charcoal residues leaching into the water during filtration (Cobb et al., 2012). Furthermore, it is less durable and physically breaks down very quickly. At the same time, coconut shell based AC is more durable and free from fine charcoal residues, where the carbonisation of one tonne coconut shells can produce almost 120 kg of AC (Bhatnagar et al., 2010).

The activation process of the AC requires chemical reagent including ZnCl₂ and H₃PO₄, K₂CO₃ (Hayashi et al., 2002; Adinata, 2007; Kalderis, 2008; Singh and Singh, 2012). The alkali hydroxide solutions such as KOH and NaOH also have been recently adopted as activation reagents as it gives the AC a high specific surface area (Hu and Srinivasan, 1999; Stavropoulos and Zabaniotou, 2005; Sudaryanto et al., 2006; Cazetta et al., 2011). Despite its desirable effect on the quality of AC, KOH and NaOH are expensive and corrosive chemicals, which need proper handling of trained personnel. On the other hand, the activation dehydrating agents such as ZnCl₂ or CaCl₂ usually employ steam activation which is excessively costly. As such, for rural communities with restricted equipment and trained personnel, a low-tech preparation of AC would benefit them to ensure water security in their area.

The present study investigates the performance of low tech prepared coconut shell AC. It also evaluates the optimum conditions (temperature and residence time) for preparation of the AC. The analyses include the carbon yield, adsorption capacity and the pore characteristics of activated products.

2. Materials and methods

Raw coconut shells were locally collected, washed and dried under the sun light for ten hours. The shells were then cleaned thoroughly with steel brush and were crushed into pieces of 5-7 cm size before putting into the stainless steel cylindrical combustion chamber placed in a furnace. Two combustion chambers were fabricated where each had one opening which can be closed by screwing the iron stopper. This keeps the chamber completely enclosed during the high temperature heating. The flow diagram is shown in Figure 1.

![Flow diagram of the methodology adopted in this study](image)

Carbonisation of the precursor was carried out by preheating the reactor at 500°C for 15 minutes. Two batches of 800g coconut shells were put for combustion simultaneously in the two chambers. The coconut shells were carbonised for 1.5 hour residence time (t) and the temperature was set at 500°C. This temperature was selected as the basis as it was suggested to be the highest devolatilation temperature (Fisher et al, 2002). Keeping the residence time fixed at t =1.5 hr, the pyrolysis temperatures (T) were systematically increased at 100°C, giving the temperature range of 500 to 800°C. In addition, the experiments were extended to carbonisation duration of two hours for batches prepared at 600 and
700°C temperatures, respectively. After carbonisation, the samples were cooled at room temperature for two hours and the pyrolysed char was taken out. The percentage of carbon yield (CY) was determined as follows:

\[
CY(\%) = \frac{W_c}{W_o} \times 100
\]

where \( W_c \) and \( W_o \) denote the charcoal dry weight (g) and the coconut shell dry weight (g), respectively.

In line with the low tech preparation, the commonly available NaCl was used as the activation reagent. The solution was prepared by mixing 35g of NaCl into 100 ml of distilled water, giving a concentration of 3.59 mol/L. The charcoal was first ground and sieved in a mechanical shaker for 20 minutes and separated into different particle sizes i.e. 0.6-1.0 mm, 1.0-2.36 mm, 2.36-4.75 mm, respectively and soaked in the NaCl solution for 24 hours inside a plastic container. The char was mixed with the NaCl solution and the impregnation ratio of 2:1. The AC particles were washed thoroughly with deionised water and dried in oven at 105°C for 24 hours to remove moisture.

Characteristics of the AC were determined by its performance to adsorb methyl orange in the liquid-phase adsorption experiments. The aqueous solution was prepared by mixing an appropriate amount of methyl orange with 1000 ml of deionised water. Adsorption experiments were conducted by mixing 10 g of the carbon sample to 50 ml of the methyl orange solution, in a glass flask. The flask was then put in a shaker for 25 minutes until it reached equilibration. The solutes from the flask were filtered prior to analysis to minimise disturbance from the carbon fines. The concentration of the adsorbate in the filtered effluent samples were determined using a Hach brand DR 6000 spectrophotometer, which measures the transmittance from 400 to 700 nm and converts to a set of abstract numbers. For each set of analysis, a control reference of methyl orange solution was prepared. The American Dye Manufacturers Institute (ADMI) colour value was recorded for each effluent sample of the prepared AC. The adsorption capacity (Ad%) for each prepared AC was presented as the percentage of adsorption, which can be calculated using the relation:

\[
A_d \text{ (\%)} = \frac{A_i - A_e}{A_i} \times 100
\]

where \( A_e \) and \( A_i \) denote the ADMI colour value of the solutes and the reference (control) value, respectively.

Analysis of Scanning Electron Microscopy (SEM) of the prepared AC was carried out for all samples to investigate the surface texture and the development of the porosity. SEMs were taken with 1250 times magnification by the Variable Pressure Scanning Electron Microscope (VPSEM) of ZEISS EVO MA 10 (UK) model. The pore diameters were measured as an averaged value from a total 25 populations for each sample.

3. Results and discussion

The carbon yield obtained and the adsorption capacity of methyl orange for each parameter varied is given in Table 1. Note that as each varying parameter has two batches of prepared AC, the averaged values are presented. Overall, the percentage of carbon yield for all samples is encouraging where it ranged from 14-33%. The highest carbon yield was obtained at the lowest temperature of 500°C and the lowest carbon yield was produced at the highest temperature T= 800°C. It can be seen in Figure 2 that pyrolysis temperature and yield percentage are inversely proportional. Increasing the T was consistently decreasing the yield percentage of AC and vice-versa. The temperature increment correspondingly increased the adsorption performance until 700°C. Although carbonisation process at T=500°C yields the highest obtained AC, it has the lowest capacity to adsorb methyl orange. AC prepared at the highest temperature T= 800°C has 41% efficiency to adsorb the methyl orange which is about 26% lower than the highest adsorption capacity at T= 700°C.

Previous research have studied the characteristics of AC which was determined by adsorption of methylene blue (MB) and the adsorption capacity was reported as milligram of MB adsorbed by unit gram of AC (mg/g). With some modified calculation to correspond to this study, the adsorption values obtained by previous research are 43.48% (Tan et al., 2008), 12.29% (Azevedo et al., 2007) and 27.79% (Kannan and Sundaram, 2001) respectively. It is found that the adsorption capacity of the low tech prepared AC is within the range obtained in other studies.

Table 1 and Figure 2 shows that the optimum conditions for good percentage of adsorption capacity is carbonisation temperature T= 700°C and residence time t=1.5 hour. Note that the optimum conditions obtained in this study has higher T and longer t of 200°C and 0.5 hours respectively, than that developed by Fisher et al. (2002) where the pore development was assisted with the usage of N2 gas. However, it is interesting to note that the optimum carbonisation temperature obtained here is comparable with other agricultural by-product studies such as...
palm shell at 800°C (Adinata et al., 2007), cassava peel at 650°C (Sudaryanto et al., 2006) and bagasse and rice husk at 700°C (Kalderis et al., 2008).

Table 1 Carbon yield and adsorption of AC at different temperature and residence time

<table>
<thead>
<tr>
<th>Sample No</th>
<th>Temperature T, (°C)</th>
<th>Residence Time t, (hours)</th>
<th>Shell Mass (g)</th>
<th>Char coal Mass (g)</th>
<th>CY (%)</th>
<th>ADMI Colour Value</th>
<th>Ad (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>800</td>
<td>1.5</td>
<td>800</td>
<td>110</td>
<td>14</td>
<td>1789</td>
<td>41</td>
</tr>
<tr>
<td>2</td>
<td>700</td>
<td>2</td>
<td>800</td>
<td>150</td>
<td>19</td>
<td>1455</td>
<td>52</td>
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<tr>
<td>3</td>
<td>700</td>
<td>1.5</td>
<td>800</td>
<td>200</td>
<td>25</td>
<td>1304</td>
<td>57</td>
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<tr>
<td>4</td>
<td>600</td>
<td>2</td>
<td>800</td>
<td>215</td>
<td>27</td>
<td>1425</td>
<td>53</td>
</tr>
<tr>
<td>5</td>
<td>600</td>
<td>1.5</td>
<td>800</td>
<td>225</td>
<td>28</td>
<td>1516</td>
<td>50</td>
</tr>
<tr>
<td>6</td>
<td>500</td>
<td>1.5</td>
<td>800</td>
<td>265</td>
<td>33</td>
<td>2092</td>
<td>31</td>
</tr>
<tr>
<td>Control</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>3032</td>
<td></td>
</tr>
</tbody>
</table>

Analysis of SEM was conducted for all samples to investigate the development of pores, shown here in Figures 3-5. The micrograph of the control virgin coconut shell, demonstrates no pores (Figure 3). In this study, two varying parameters were investigated here i.e. the carbonisation temperature and the residence time. For coherent discussion, the effect of varying temperature will be first discussed. The pyrolysis process clearly develops pores in the coconut-shell AC. At T=500°C and t=1.5 hours, pores were created and begin to be visible although they are to well developed (Figure 4a). Thick wall structure can be found in Figure 4a with little porosity, where the averaged pore diameter was found to be 2.86 μm and was uniformly developed within the sample. Larger and widened pores with simultaneous thinner pore
Production and Characterisation of Activate
d Carbon from Coconut Shell
Shaheed et al.
Journal of Hydrology and Environment Research

wall were clearly visible as the temperature increased to T=600°C and 700°C, as shown in Figure 4b and 4c, respectively. The pores developed at T=600°C show a progressive increase of size at 7.1 μm and at AC prepared at T=700°C show a more orderly structural skeleton with larger pores of 17.3 μm (Table 2). At fixed residence time t=1.5 hours, increasing T by 100°C from initial temperature 500°C gave an increment of pore diameter to 148% and 503% at T=600°C and 700°C, respectively. The development of pores, however reached their maximum diameter of approximately 17 μm even when the temperature is increased to 800°C.

Figure 3-4 SEM images of coconut shell AC pyrolysed for 1.5 hours at varying temperatures T; (a) 500°C (b) 600°C (c) 700°C (d) 800°C (mag. 1250×)

This pore widening is due to further devolatilation from the pore surface and merging of adjacent pores. Increasing the temperature to T=800°C shows the structural pore walls were either started to collapse or have already disintegrated and merged with the adjacent pores (Figure 4d). The collapsed pore walls may have blocked the micropore channels and inhibit the accessible adsorption pore sites (Yang, 2010; Shabanzadeh, 2012). This supports the methyl orange analysis where the capacity to adsorb was significantly reduced for the AC prepared at T=800°C. SEM analysis demonstrate gradual development of pores in the coconut-shell AC, i.e. creation of new pores at T =500°C and widening of existing pores at T =600°C to 700°C. The well-developed pores in the prepared AC are the main factor that leads to the high adsorption capacity as previously shown in Section 3.1.

With increasing residence time t=2 hours and at T=600°C the pore walls remain well-structured (Figure 5a) but the pore walls were slightly disintegrated when T =700°C (Figure 5b). Even so, both conditions show a comparable adsorption capacity at 53% as shown in Table 1. Increasing t obviously resulted an increment of the pore diameter; when t was made 0.5 hours longer at T=600°C and 700°C, the averaged pore size increased 62% and 16% respectively from that of lower residence time (Table 1). The pore walls at t=2 hours and T=700°C started to collapse, giving a more favourable options for the AC prepared at T=700°C but with shorter residence time i.e. t=1.5 hours. It can be said that the diffusion of NaCl into the pores was promoted with increasing t and T, allowing a better NaCl-carbon reactions. The opening and enlargement of the pores enhanced the surface area and pore volume of the AC, thus increasing the adsorption capacity as shown previously in the methyl orange analysis. To give a better illustration, the averaged pore diameters are listed in Table 2, along with the statistical parameters.
measured. The minimum, maximum, average and median diameters measured are given in μm. The standard deviation is also presented. Depending on these data, a box plot is prepared.

![SEM images of coconut shell AC pyrolysed for 2 hours at carbonisation temperatures of (a) 600°C (b) 700°C (mag. 1250×)](image)

**Figure 5** SEM images of coconut shell AC pyrolysed for 2 hours at carbonisation temperatures of (a) 600°C (b) 700°C (mag. 1250×)

<table>
<thead>
<tr>
<th>Temperature, T (°C)</th>
<th>500</th>
<th>600</th>
<th>600</th>
<th>700</th>
<th>700</th>
<th>800</th>
</tr>
</thead>
<tbody>
<tr>
<td>Residence Time, t (hours)</td>
<td>1.5</td>
<td>1.5</td>
<td>2</td>
<td>1.5</td>
<td>2</td>
<td>1.5</td>
</tr>
<tr>
<td>Average diameter in μm</td>
<td>2.86</td>
<td>7.08</td>
<td>11.47</td>
<td>17.26</td>
<td>20.01</td>
<td>17.12</td>
</tr>
<tr>
<td>Standard Deviation</td>
<td>1.44</td>
<td>2.46</td>
<td>3.13</td>
<td>7.39</td>
<td>2.98</td>
<td>3.17</td>
</tr>
<tr>
<td>Minimum diameter</td>
<td>1.19</td>
<td>2.72</td>
<td>6.26</td>
<td>7.76</td>
<td>13.78</td>
<td>11.37</td>
</tr>
<tr>
<td>median</td>
<td>2.78</td>
<td>7.51</td>
<td>11.29</td>
<td>15.26</td>
<td>20.29</td>
<td>17.01</td>
</tr>
<tr>
<td>Maximum diameter</td>
<td>5.16</td>
<td>10.20</td>
<td>16.89</td>
<td>29.12</td>
<td>23.67</td>
<td>21.83</td>
</tr>
</tbody>
</table>

**Table 2** Pore diameter of the prepared AC at different Temperature and Residence time

Figure 6 shows the box plot analysis of the measured pore diameter for all samples. It is evident that the pore diameters of the AC reached maximum at t=1.5 hour and T= 700°C before stabilized independent of increasing residence time and temperature. The standard deviation (σ) of the pore diameters distribution remains within the range of 1.44 to 3.17 for all pyrolysis conditions except at t=1.5 hour, T=700°C, where σ = 7.4. The high σ value is due to the wide range of pore size distribution from 7.76 to 29.12 μm where the pores can be newly created, widening of existing pores and merging of some neighbouring pores.

The performance of the AC was also examined when the prepared AC particle sizes were varied. Table 3 shows the adsorption capacity of the prepared AC for three ranges of particle sizes, with the minimum diameter of 0.6 mm and the largest of 4.75 mm. This table shows three types of particle size ranges; type I (0.6-1.0 mm), type II (1.0-2.36 mm) and type III (2.36-4.75 mm) to account for its different group. The results indicate that the adsorption capacity increased to some extent with the decreasing particle size of the adsorbent. AC particles type II have the highest adsorption capacity of 58%, whilst the largest type III have lower capacity to remove the methyl orange at only 32%. Smaller AC particles have greater portion of the total pore volume to adsorb molecules and is more closely to a true equilibrium than the larger particles (Daifullah et al., 2004). As the AC particles was produced by grinding larger particles to its designated group, it is likely that a certain percentage of the pore volume and area of porous AC is unavailable to the methyl orange molecules because of the “ink-well” or “bottle-neck” constrictions (Weber et al., 1983). Furthermore, the decreasing adsorption was promoted as the distance from its external surface was decreased as a result of reduced accessibility of its inner regions to activation during the preparation process (Selles-Pérez and Martín-Martínez, 1991).

It is expected that as the particle size decreasing, the specific surface area for adsorption is increased (Osman and Samuel, 1983). However, in this study, type I particles with the smallest particle size range have lower adsorption percentage than type II at 41%. It can be deduced that the enlargement of micropores to mesopores reduced the specific surface area for type I particles, although this is yet to be asserted with confidence.
4. Conclusion

Low tech preparation of AC using coconut shell was conducted using commonly available NaCl as the activation reagent. Effects of the carbonisation temperature and residence time were investigated on the yield percentage of pyrolysed carbon and their performance on the methyl orange adsorption. The pyrolysis optimum condition to prepare the low tech AC from coconut shell is 700°C temperature and 1.5 hours residence time. This was concluded by the high percentage (25%) of carbon yield and the highest adsorption capacity, i.e. 58% removal of methyl orange. At this optimum condition, the diameter of the pore is approximately 17.3 μm. The coconut-shell AC with particle sizes ranges from 1.0-2.36 mm proved to have better adsorption capacity, where below and above this range reduced the adsorption performance. SEM analysis shows that the pore size was gradually developed with creation of new pores, widening and merging of adjacent pores at T=500°C, 600°C and 700°C, respectively. This low tech process of preparing AC from coconut shell not only will be able to save energy and cost, but also promotes sustainability by utilising readily available bio-waste material. It is foreseen that the water treatment process can be widened to the rural communities, with minimum equipment and simple procedures.

6. Acknowledgements

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References


Shoreline Changes Analysis and Coastal Geomorphology of Tiswadi Taluka of Goa State in India

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1 Water Resource Division, Spatial Decisions, New Delhi, India
2 Department of Mathematics, PG Collage, District Sagar (M.P.) India

Abstract: This study examines soil, drainage network, geology, coastal geomorphology and shoreline changes using remote sensing and GIS techniques. Multi-sensor, multi-resolution and multi-temporal satellite imageries of IRS/LandSAT have been used to extract the shoreline from 1973 to 2014. The statistical techniques called end-point rate (EPR) and weighted-linear-regression-rate (WLR) have been used to find out the rate of change during the period of 1973-2014 by Digital Shoreline Analysis System (DSAS) tool, which is an extension of ArcGIS software. Average erosion rate and accretion rate (m/year) were found to be -0.73 (EPR) & -0.21 (WLR), and 1.22 (EPR) & 0.53 (WLR), respectively. The shoreline vectors were likened to each other to determine the shoreline change. It is estimated that average erosion area per year from 1973 to 2014 was 0.168 km², while the average accretion area per year for same period was 0.124 km². Preliminary results show erosion happening near to human activities in shoreline of Tiswadi taluka. The study of geomorphology of Tiswadi taluka coast of Goa state includes the image interpretation of IRS/LandSAT satellite imageries, CartosAT/ASTER DEM data and SoI topographical maps with limited field check. The landforms of the study area are the results of the combination of lithology, structure, interruption and sea-level change. The findings of this study can be useful guide to similar studies for identification of shore lines, which has particular significance in the sea level rise studies in the changing climate condition.

Keywords: Geology, geomorphology, shoreline change, Tiswadi taluka, remote sensing, GIS.

1. Introduction

Growing population along with urban, economic, industrial and agricultural growth has an adverse impact on world’s environment especially on the coastal area. From the time immemorial the coastal lands have attracted human population to spread their nest around its’ natural beauty. During the course of time, small villages turn into big towns and eventually into big cities. This resulted in a high degree of shore development. These developmental activities either destroyed or reduced the coastal vegetation like mangroves, coastal dunes and dune plants. The combined effects of development and depletion of coastal vegetation led to increase of coastal runoffs and drainage flow, which trigger shoreline change. The human induced developmental activity not only affects natural resources, but also modifies the natural coastal processes to a considerable degree. Major coastal constructions like reclamation piers, or changes in the coastal profile, can strongly influence waves and current refraction and therefore the direction of long swells and dominant current direction and speed. This results in undesired and unpredictable erosion or accretion along the coast line.

Remote sensing and GIS data have widely been used for shoreline change analysis, coastal landforms and geomorphological studies along with management and scientific issues in the coastal areas. Due to its repetitive, multispectral and synoptic nature, remote sensing data has proved to be extremely useful in providing multi-spectral information on various components of the coastal environment, viz. coastal wetland conditions, mangrove and coral reef degradation, coastal landforms and shoreline changes, tidal boundaries, brackish water areas, suspended

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sediment dynamics, coastal currents and air pollution (Nayak et al., 2001). IRS LISS-III and LandSAT-MSS, TM, ETM+ and OLI which have proved to be extremely useful in the mapping details of ports and geological/geomorphological mapping as well as in assessing damage due to cyclones in the coastal areas and shoreline changes (Nayak et al., 1996; Navalgund and Bahuguna, 1999; Mitra et al., 2000).

The coastal geomorphology of the study area has been discussed by Telles, 1911; Feio, 1956; De Souza, 1968; & Vaidyanadhan, 1968 and then the availability of satellite image data in different forms has ensured many successful applications in coastal geomorphology mapping e.g. Wagle et al., 1975; Desai et al., 1978; Sriram et al., 1979, 80; Guzder, 1980; Wagle, 1982; Kale et al., 1983; Rao et al., 1985; Bhattacharya et al., 1985; Pant et al., 1985; Sinha, 1985; Wagle, 1987; Kunte, 1990; Wagle, 1993; Kunte et al., 1994. The use of remote sensing and GIS for shoreline extraction and shoreline changes analysis have been described by Pethick, 1984; Shaikh et al. 1989; Pal and Pal, 1993; White et al., 1999; Nayak, 2000, 2002; Ron Li et al., 2001; and Dellepiane et al., 2004.

2. Study area

Goa is one of the most famous tourist coasts of the world, which is situated along the central-west coast of India, representing part of the Konkan coast. The study area chosen for the geomorphological study is a Tiswadi taluka of the north Goa district - Goa, covering an area of about 213.5 km². It lies between 15°24'30" & 15°34'05" North latitudes, and 73°46'46" & 73°57'41" East longitudes. The area for shoreline change analysis lies between 15°19'26" & 15°35'19" North latitudes, and 73°43'55" & 73°53'43" East longitudes, it has a coastline of about 79.33 km (Figure 1). The study area falls in Survey of India (SoI) topographic maps (1:50,000 Scale) number 48E/14 & 48E/15.

A general lithological characteristics of the area are beach sand, laterites, quartz-sericite schist, and greywacke with conglomerate. The main geomorphological unit in the study area are coastal plains, islands, dissected table land, low cut terraces, laterite mesas, lineaments/dykes, estuary, beach, tidal flat, tidal river, channel spit, beach ridge, salt pans, mangroves, estuary island and exposed rocks. Major soil types are laterites, alluviums, sandy coastal soils, saline soils and marshy soils. The area is surrounded by the Mandovi river Zuari river, both having estuarine characteristics. Primarily the underlying rocks govern the drainage system in the area. The drainage pattern is generally dendritic type. The study experiences a tropical & humid climate. Rain occurs due to the south-west monsoon winds from June to September. The average annual rainfall (1971-2013) is about 2,932 mm, bulk of which is received during monsoon months of June to September. Almost 32% of the annual rainfall is received in the month of July. The maximum temperature recorded is 36°C and the minimum is 23°C during May and January respectively. Total population of Tiswadi Taluk is 177,219 living in 42,241 houses, spread across total 20 villages and 20 panchayats.
3. Data used and sources

The primary source of data is IRS or LandsAT satellite data, while ancillary data are the survey of India toposheet, geological map and soil map for shoreline changes analysis and coastal geomorphology study. The data used for this study were multi-temporal, multi-sensor, and multi-resolution satellite imageries of Landsat series such as Landsat - MSS, TM, ETM+, OLI; IRS series such as ResourceSat-2 LISS-III; IRS-P5 CartoSAT-1 DEM, and ASTER (DEM) which have been acquired on cloud free days, over the chosen period (1973 to 2014). The advantage of these satellite imageries for coastal applications is that it is made available at no cost to user communities. Survey of India (SoI) topographical maps at 1:50,000 Scale 48E/14, and 48E/15 have been used for preparation of base map, and drainage map. Soil map and geological map were used for preparation of geomorphological map with limited field check. In this study, ERDAS Imagine-2013 and ArcGIS-10.3 software including DSAS tool have been used for digital/vector analysis, transect wise shoreline changes analysis, and hydrological/geomorphological analysis. Data used and sources are shown in Table 1.

<table>
<thead>
<tr>
<th>S.No.</th>
<th>Data Layer / Maps</th>
<th>Sources</th>
</tr>
</thead>
<tbody>
<tr>
<td>1.</td>
<td>Topographical Map</td>
<td>Survey of India topographical map at 1:50,000 Scale</td>
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<tr>
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<td>48 E / 14, and 48 E / 15</td>
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<td>2.</td>
<td>Remote Sensing Data</td>
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<td>LandsAT-5 TM (30m): 19th November, 1989</td>
</tr>
<tr>
<td></td>
<td></td>
<td>LandsAT-7 ETM (30m): 14th November, 1999</td>
</tr>
<tr>
<td></td>
<td></td>
<td>LandsAT-7 ETM (30m): 02nd November, 2006</td>
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<td></td>
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<td>IRS ResourceSat-2 LISS-III (23.5 m): 14th October, 2009</td>
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<td></td>
<td>LandsAT-8 OLI (30m): 09th July, 2014</td>
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<td>ASTER (DEM) Data (30m): 02nd December, 2003</td>
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<tr>
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<td></td>
<td>IRS-P5 CartoSAT-1 DEM Data (30m): 26th September, 2009</td>
</tr>
<tr>
<td>3.</td>
<td>Drainage Map</td>
<td>Drainage network has been generated in GIS environment using ASTER (DEM) data, CartoSAT-1 (DEM) Data and ArcHydro Tool in ESRI ArcGIS-10.2.2 software</td>
</tr>
<tr>
<td>4.</td>
<td>Soil Map</td>
<td>Soil map of Goa has been collected from National Bureau of Soil Survey and Land Use Planning (NBSS&amp;LUP) and updated through satellite data</td>
</tr>
<tr>
<td>5.</td>
<td>Geological Map</td>
<td>Goa state geological map has collected from GSI and updated through IRS LISS-III &amp; Landsat-8 OLI satellite data with limited field check</td>
</tr>
<tr>
<td>6.</td>
<td>Geomorphological Map</td>
<td>Geomorphological map has been prepared using the Landsat-8 OLI data, CartoSAT-1 DEM / ASTER-DEM data, and other ancillary data i.e. SoI topographical map, GSI geometrical map</td>
</tr>
</tbody>
</table>

4. Results and discussion

4.1 Drainage network

The study area is drained by a network of two estuarine rivers, namely Mandovi, and Zuari River. These rivers originate in the Western Ghats, but soon lose their energy as they wander through the midlands and the coastal plains to discharge into the Arabian Sea. They are characterized by imperceptible gradients in the lower reaches resulting in the tidal waters entering several km inland.

The drainage network of the study area has been generated in GIS environment using ASTER (DEM) data, CartoSAT-1 (DEM) data, Survey of India (SoI) topographical maps (1956) of 1:50,000 scale (Toposheet No. 48E/14, and 48E/15) and Arc Hydro tool of ESRI ArcGIS-10.3 software. Therefore, drainage network has been digitized by ArcGIS-10.3 construction tools after geometric transformation of SoI toposheets, satellite imageries, and ASTER/CartoSAT DEM data in UTM coordinate system (WGS 1984 / Zone 43N). The drainage pattern is generally dendritic type. Authors have carried out the stream ordering system based on the method proposed by Strahler (1952). It has also been noticed that there is a decrease in stream frequency as the stream order increases. A drainage network map of the study area is shown in Figure 2.
4.2 Soil type

The soil map was generated in GIS environment using soil map collected from National Bureau of Soil Survey and Land Use Planning (NBSS&LUP) - Nagpur and were updated through LandSAT-8 OLI (Operational Land Imager) and IRS ResourceSAT-2 LISS-III satellite data by using ArcGIS-10.3 software with limited field check (Figure 3). The soil map obtained from the NBSS&LUP was geo-metrically registered to the base data to match Landsat & IRS satellite imageries. The geo-referenced soil map was used to assist in visual classification of satellite imagery for obtaining soil categories. According to “Task Force on Eco-Development Plan for Goa Report, 1982”, generally five main types of soils are found in the study area i.e. laterites, alluviums, sandy coastal soils, saline soils and marshy soils. Forest Survey of India (FSI) has published a report on “Resource Survey of Goa Forest” in 1985. FSI has classified the soil type of Tiswadi taluka area in nine classes as shown in Table 2.

4.3 Regional geology and structural landforms

A general lithological map can help understanding of the geomorphology of an area. A geological map of the study area has been prepared by using LandSAT-8 OLI (Operational Land Imager) (30m MSS & 15m PAN merged satellite imagery), IRS-P5 CartoSAT-1 DEM data (2.5m), Geological Survey of India (GSI) map for Goa, Survey of India (SoI) topographical map at 1:50,000 Scale (Figure 4). The general geology of the study area has been described by the various geologists such as Telles (1911), Feio (1956), De Souza (1968), Vaidyanadhan (1968), Ahmed (1972), Wagle et al. (1975), Desai et al. (1978), Sriram et al. (1979, 80), Guzder (1980), Wagle (1982), Kale et al. (1983), Rao et al. (1985), Bhattacharya et al. (1985), Pant et al. (1985), Sinha (1985), Wagle (1987) and Wagle (1993). They recorded the principal rock formations as described in Table 3.

Major part of the study area is covered by the formation of Goa Group belonging to Dharwar Super-Group of Archaean to Proterozoic age. The rock types representing the Dharwars are quartzites, quartz-sericite schist, and meta-greywackes with conglomerate, which is distributed in a general NW-SE direction. The rock types of Goa Group have suffered considerable faulting; all the faults are not exposed on surface owing to the extensive cover of laterite. The rocks have been subjected to lateritisation of varying thickness during the recent times. Laterites
and lateritic soils are found to occur capping the rocks forms the high plateaus of the Sahyadri down to the sea level. Thus, laterite occurs extensively covering almost all the formations in area. Beach sand occurring along the coastal plains consists of fine to coarse sands.

<table>
<thead>
<tr>
<th>Soil Series</th>
<th>Classification</th>
<th>Major Soil Type</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Padi</td>
<td>Loamy-skeletal mixed. isohyperthermic Ustoxic Dystropepts</td>
<td>Padi Gravelly Sandy Loam</td>
<td>Uncultivated. The moderately shallow, gravelly sandy loam soils of Padi series occurring on hill slopes are stony and rocky.</td>
</tr>
<tr>
<td>Nagowa</td>
<td>Fine, mixed. isohyperthermic Ustoxic Dystropepts</td>
<td>Nagowa Gravelly Sandy Clay Loam</td>
<td>Cultivated. The deep, gravelly sandy clay loam soils of Nagowa series occurring on dissected hills with 8-15 per cent slope are unfit for annual crop production.</td>
</tr>
<tr>
<td>Mandavi</td>
<td>Mixed, isohyperthermic Typic Paanmaquents</td>
<td>Mandavi Sand</td>
<td>Uncultivated, The deep, sandy, poorly drained soils of Mandavi series occur along river banks and on mud flats.</td>
</tr>
<tr>
<td>Madgaon</td>
<td>Loamy-skeletal, mixed, isohyperthermic Fluventic Ustopepts</td>
<td>Madgaon Gravelly Loamy Sand</td>
<td>Cultivated, Deep soils of Madgaon series are gravelly sandy in surface and gravelly clay loam in subsoil and occur on hill slopes.</td>
</tr>
<tr>
<td>Karven</td>
<td>Clayey-skeletal, kaolinitic, isohyperthermic Oxic Ustopepts</td>
<td>Karven Gravelly Clay</td>
<td>Forest. The deep, gravelly clay soils of Karven series occur on escarpments of hills. Stoniness. Rockiness and excessive slope of the lands make it unfit for annual crop production.</td>
</tr>
<tr>
<td>Kalangut</td>
<td>Fine-loamy, mixed, isohyperleriennic Typic Tropaquents</td>
<td>Kalangut Loamy Sand</td>
<td>Cultivated, Kalangut soils occurring on nearly level low lands have sandy surface texture and sandy loam sub-soils.</td>
</tr>
<tr>
<td>Karmali</td>
<td>Clayey, mixed, isohyperthermic Lithic Dystropepts</td>
<td>Karmali Gravelly Clay</td>
<td>Cultivated, The shallow, gravelly, clay soils of Karmali series occur on flat topped laterite hills.</td>
</tr>
<tr>
<td>Dabolim</td>
<td>Loamy, mixed, isohyperthermic Lithic Ustorthents</td>
<td>Dabolim Sandy Clay Loam</td>
<td>Pasture Land. The very shallow soil of Dabolim series occurring on flat topped laterite hills with extensive exposure of laterite is unfit for any kind of crop production.</td>
</tr>
<tr>
<td>Chapora</td>
<td>Fine-loamy, mixed, isohyperthermic Typic Ustopepts</td>
<td>Chapora Sandy Loam</td>
<td>Cultivated, Chapora soils occurring on gently to moderately sloping lands of coast are deep and sandy loam in texture.</td>
</tr>
</tbody>
</table>

Source: National Bureau of Soil Survey and Land Use Planning (NBSS&LUP) & Forest Survey of India (FSI)

4.4 Coastal geomorphology

Coastal geomorphology relates to landform features and land forming processes that are shaped by atmospheric, terrestrial and marine processes. Coastal landforms are dynamic systems that function over a range of temporal and spatial scales. Due to this dynamism, the understanding and prediction of coastal geomorphological behaviour is a continually evolving science (Whitehouse, 2009). The capability of satellite remote sensing to provide synoptic, repetitive and multispectral data has proved to be very useful in the inventory and monitoring of coastal feature, such as tidal wetlands, coastal landforms/geomorphology, potential aquaculture sites, mangroves, estuary dynamics/shoreline changes and offshore aspects like suspended sediment dynamics and coastal currents, near-shore bathymetry, internal waves etc. Several geomorphologists have contributed to diverse geomorphological aspects of the study area, notable among these are discussed below.

Telles (1911) and Feio (1956) defined the geomorphic problems of Goa disbursing special consideration to the Goa Escarpment. The escarpment was well-thought-out by them as an erosional. They said that the difference of levels of the order of 1000m between the Deccan Plateau and the coastal region show a high probability of tectonism. De Souza (1968) physiographically classified the Goa area into three regions: (i) the coastal tract consisting of beaches, rugged sea-cliffs, small isolated pocket braches, recent broad alluvial plains, vast estuaries, older spits, dunes, beach ridges and broad hard wave cut platforms; (ii) the tract between the coast and the ghats; (iii) the high ranges of the western ghat rising to elevation of 900 m to 2500 m above meal sea level. He has also described the geomorphic features of the laterites of study area in 1968 and accordingly laterite covers the major portion of the area and typically occurs as plateau landforms. Additionally, the
plateau-top laterite is the area showing characteristics identical to the high level type and it has differentiated layers similar to other reported in situ deposits. However, it is possible to recognise a high and a low level variety of laterite within the study area on a single hill feature and its adjoining lowlands. Vaidyanadhan (1968) has marked the geomorphological features i.e., filled bays, marshy area, sea cliffs, stacks, tidal flats, tom bolos, etc. based on the topographic maps.

Figure 3 Soil map of Tiswadi taluka, Goa

Figure 4 Geological map of Tiswadi taluka, Goa
Table 3 Stratigraphy of the study area

<table>
<thead>
<tr>
<th>Age</th>
<th>Group</th>
<th>Formation</th>
<th>Rock Type</th>
</tr>
</thead>
<tbody>
<tr>
<td>Quaternary</td>
<td></td>
<td></td>
<td>Beach Sand</td>
</tr>
<tr>
<td>Cenozoic</td>
<td></td>
<td></td>
<td>Laterites</td>
</tr>
<tr>
<td>Upper Cretaceous</td>
<td>Clopset Granite</td>
<td></td>
<td>Granite</td>
</tr>
<tr>
<td>to Eocene</td>
<td>Peridotite, Gabbro, Norite</td>
<td>Pyroxenite, Peridotite, Serpentinite, Gabbro</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Vageri Formation</td>
<td>Carbonate-Quartz-Chlorite Schist with Greywacke</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Dolomitic Limestone</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Quartz-Sericite Schist</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Banded Iron Formation</td>
<td></td>
</tr>
<tr>
<td>Archaean to Lower</td>
<td>Bicholim Formation</td>
<td>Chert and Quartzite</td>
<td></td>
</tr>
<tr>
<td>Proterozoic</td>
<td>Goa Group (Dharwar Super-Group)</td>
<td>Quartz-Chlorite-Biotite Schist with Layers of Chert, Iron</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Sanvordem Formation</td>
<td>Oxide, Carbonate, Meta-basalt and Meta Gabbro</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Greywacke with Conglomerate</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Barcem Formation</td>
<td>Quartzite</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Quartz-Chlorite Schist</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Meta-acid Volcanics</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Meta-Basalt</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Orthoquartzite</td>
<td></td>
</tr>
<tr>
<td>Archaean</td>
<td>Peninsular Gneissic Complex</td>
<td>Granite Gneiss, Migmatites and Granites</td>
<td></td>
</tr>
</tbody>
</table>

Source: Geological Survey of India

Wagle and Misra (1975) have observed the intensive headward erosion of the rivers based on aerial photo interpretation and field check. They stated that the present western face of the Western Ghats is simply a product of circumdenudation of an old table land. Desai and Peshwa (1978) studied the drainage irregularities on the west coast region of Maharashtra and Goa based on remote sensing data and explained that the lithology and structure controlled the drainage. Sriram and Prasad (1979, 80) noticed in the northern part of Goa that the off-cuts of planar surfaces showing deep lateritic weathering profiles. They also described that the geomorphological evolution of the lateritic terrain of Goa composed of a sequence of events that are recognised as repetitive, slow regional appearance with recurrent phases of quiescence leading to the formation of the planation surface. According to Guzder (1980) the occurrence of laterite gravel close to sea level in Goa is evidence of sea level oscillations during the Quaternary phase.

Wagle (1982) has identified fluvial, marine and eolian features such as tidal flats, river terraces, mesas, wave-cut, platforms, old beach ridges, dune, etc. based on aerial photos interpretation of Goa. He has described the coastal laterite height ranging from 40 m to 100 m. He has also verified these heights through the field checks. The tops are covered with massive laterite ranging in thickness from 20 m to 40 m. According to Kale and Rajaguru (1983), the landscape of Goa is characterised by well indurated lateritic hilltops or by non-lateritic plateau on the coast. Resting on the slopes of these coasts are lateritic terraces with a homogenous or heterogeneous character. Rao and Subramanian (1985) have studied the geomorphology and morphotectonic features of Goa based on the aerial photos and landsat imagerys. They stated that the geomorphology of Goa is the result of a combination of several geomorphic processes associated with climatic and eustatic sea level changes. Three main physiographic zones are identified, i.e. the narrow coastal plains on the west, followed by a vast, dissected tableland and the steeply rising Western Ghats.

Bhattacharya, Perumal and Roy (1985) had studied the geology, geomorphology and lineaments of the west coast including the Goa area by using aerial photographs and landsat satellite imagery. This study has not given any direct evidence of faulting along the west coast. According to Pant et al. (1985) a cross section of the Goa landscape comprises an imposing chain of hills of Western Ghats in the east, the undulating topography with moderate hills alternating with low broad valleys in the central part and the coastal plains in the west. Sinha (1985) has marked the geomorphic features using physiographic boundaries such as hills (laterite soils), rolling to undulating upland (lateritic soil and red loam soil), coastal plains (coastal alluvium) and flood plains (saline and non-saline soil).
Wagle (1987) has described the geomorphology and evolution of the coastal and offshore area of Maharashtra and Goa presented a general geomorphological map of the coastal area and the processes operating therein. Kunte (1990) has applied various image analysis techniques to demarcate the geological features and lineaments of the study area using the landsat satellite imagery. Wagle (1993) has studied the geomorphology of Goa and Goa coast through aerial photographs, landsat satellite imagery and ground checks. He has demarked the physiography into three regions i.e. coastal tracts, sub-ghat region and high ranges of the Western Ghats. Generally landforms are the result of a combination of structure, lithology, interruptions, denudation and sea level changes. Kunte and Wagle (1994) have analyzed the aerial photographs, landsat and SOPT satellite imageries of the Goa and identified the fluvial, estuarine, marine, vegetation & near shore features.

The geomorphological map (Figure 5) of the study area has been prepared by using LANDSAT-8 OLI (Operational Land Imager) satellite imagery, IRS-P5 Cartosat-1 DEM data, ASTER (DEM) data, and SoI maps of 1:50,000 scale with limited field checks. GSI Geological maps (structural and lithological) have also been referred. Coastal landforms/geomorphology classification of the study area was based on the landforms, geology, sediment pattern, some other factors, including the geomorphic process and land cover/vegetation were also considered. The various geomorphic units and their component were identified and mapped. The description of different geomorphic units of Tiswadi Taluka has been given in Table 4.

**Table 4 Important geomorphic units of Tiswadi taluka, Goa**

<table>
<thead>
<tr>
<th>Map Symbol</th>
<th>Geomorphic Units</th>
<th>Lithology</th>
<th>Location</th>
<th>Association</th>
<th>Description or Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>DCP(Sc)</td>
<td>Coastal Plains</td>
<td>Schist</td>
<td>Adjacent to coast</td>
<td>-</td>
<td>A coastal plain is an area of flat, low-lying land adjacent to a seacoast</td>
</tr>
<tr>
<td>DCP(Gc)</td>
<td>Coastal Plains</td>
<td>Greywacke with Conglomerate</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>DIS(Sc)</td>
<td>Islands</td>
<td>Schist</td>
<td>Adjacent to coast, river, water</td>
<td>Water</td>
<td>Piece of sub-continental land that is surrounded by water</td>
</tr>
<tr>
<td>DIS(Gc)</td>
<td>Islands</td>
<td>Greywacke with Conglomerate</td>
<td>bodies</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>DDTL(La)</td>
<td>Dissected Table Land</td>
<td>Laterite</td>
<td>-</td>
<td>-</td>
<td>Dissected table land are characterized by a variety of interwoven landforms preserved by substantial uplift</td>
</tr>
<tr>
<td>DDTL(Sc)</td>
<td>Dissected Table Land</td>
<td>Schist</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>DDTL(Gc)</td>
<td>Dissected Table Land</td>
<td>Greywacke with Conglomerate</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>DLCT(Gc)</td>
<td>Low Cut Terraces</td>
<td>Greywacke with Conglomerate</td>
<td>River</td>
<td>River</td>
<td>Narrow flat area often found at the base of a sea cliff or along the shoreline of a lake, bay, or sea</td>
</tr>
<tr>
<td>DLCT(La)</td>
<td>Low Cut Terraces</td>
<td>Laterite</td>
<td>-</td>
<td>-</td>
<td></td>
</tr>
<tr>
<td>DRT(Sc)</td>
<td>River Terraces</td>
<td>Schist</td>
<td>Along the river channel</td>
<td>River</td>
<td>River terraces lie parallel to and above the river channel and its floodplain</td>
</tr>
</tbody>
</table>

**Coastal Geomorphology: Structural**

<table>
<thead>
<tr>
<th>Map Symbol</th>
<th>Geomorphic Units</th>
<th>Lithology</th>
<th>Location</th>
<th>Association</th>
<th>Description or Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>SLm(La)</td>
<td>Laterite Mesas</td>
<td>Laterite</td>
<td>-</td>
<td>-</td>
<td>Laterite Mesas is a flat, elevated landform</td>
</tr>
<tr>
<td>SLs(Gc)</td>
<td>Lineaments</td>
<td>Cut across various lithology</td>
<td>Fault, fractures etc.</td>
<td>Fault line, fractures, joints, shear zone, contact zones, other linear features and straight stream courses</td>
<td></td>
</tr>
<tr>
<td>SD(Gc)</td>
<td>Dykes</td>
<td>Greywacke with Conglomerate</td>
<td>-</td>
<td>-</td>
<td>Quartz intrusions that cut across the rock</td>
</tr>
<tr>
<td>SDf(Gc)</td>
<td>Dip of Foliation</td>
<td>Greywacke with Conglomerate</td>
<td>-</td>
<td>-</td>
<td>Strike &amp; Dip of Foliation / Banding / Schistosity</td>
</tr>
</tbody>
</table>
Shoreline changes analysis and coastal geomorphology of Tiswadi Taluka

Coastal Geomorphology: Marine

<table>
<thead>
<tr>
<th>ME(Gc)</th>
<th>Estuary</th>
<th>Greywacke with Conglomerate</th>
<th>Adjacent to coast Funnel, Semi circular</th>
<th>Part of the lower river course that is affected by the mixing of salt water with the fresh water</th>
</tr>
</thead>
<tbody>
<tr>
<td>MB(Bs)</td>
<td>Beach</td>
<td>Beach Sand</td>
<td>Adjacent to coast on land-water boundary Linear, Open coast</td>
<td>Made up of fine to medium coarse sand</td>
</tr>
<tr>
<td>MTr(La)</td>
<td>Tidal Flat</td>
<td>Laterite</td>
<td>Irregular, Between high &amp; low waterline</td>
<td>Bordering the estuarine region of rivers, creek &amp; banks</td>
</tr>
<tr>
<td>MTr(Sc)</td>
<td>Tidal Flat</td>
<td>Schist</td>
<td>Part of the beach extending into the sea Linear, Open coast</td>
<td>Narrow embankments of land consisting of sand / gravel with one end attached to the mainland and other terminating into the sea</td>
</tr>
<tr>
<td>MTr(Gc)</td>
<td>Tidal Flat</td>
<td>Greywacke with Conglomerate</td>
<td>Adjacent to coast</td>
<td>Between two rocky headlands</td>
</tr>
</tbody>
</table>

Coastal Geomorphology: Fluvial

<table>
<thead>
<tr>
<th>FTr(La)</th>
<th>Tidal River</th>
<th>Laterite</th>
<th>Intertidal area River</th>
<th>Intricate networks of narrow inlets</th>
</tr>
</thead>
<tbody>
<tr>
<td>FTr(Gc)</td>
<td>Tidal River</td>
<td>Greywacke with Conglomerate</td>
<td>Part of the beach extending into the sea Linear, Open coast</td>
<td>Narrow embankments of land consisting of sand / gravel with one end attached to the mainland and other terminating into the sea</td>
</tr>
<tr>
<td>FCs(Sc)</td>
<td>Channel Spit</td>
<td>Schist</td>
<td>Intertidal area</td>
<td>Irregular, Quiet depositional area Grows on substrates of mud / sand composition, percentage of mud is more</td>
</tr>
</tbody>
</table>

Coastal Geomorphology: Aeolian

<table>
<thead>
<tr>
<th>ABr(Gc)</th>
<th>Beach Ridge</th>
<th>Greywacke with Conglomerate</th>
<th>Adjacent to coast</th>
<th>Between two rocky headlands Develops due to formation of Prolific headlands</th>
</tr>
</thead>
</table>

Coastal Geomorphology: Others

<table>
<thead>
<tr>
<th>OSp(La)</th>
<th>Salt Pans</th>
<th>Laterite</th>
<th>Near the high tide boundary On the land, within mudflat area</th>
<th>Rectangular, white</th>
</tr>
</thead>
<tbody>
<tr>
<td>OSp(Sc)</td>
<td>Salt Pans</td>
<td>Schist</td>
<td></td>
<td></td>
</tr>
<tr>
<td>OSp(Gc)</td>
<td>Salt Pans</td>
<td>Greywacke with Conglomerate</td>
<td></td>
<td></td>
</tr>
<tr>
<td>OMg(La)</td>
<td>Mangroves</td>
<td>Laterite</td>
<td>Intertidal area</td>
<td>Irregular, Quiet depositional area Grows on substrates of mud / sand composition, percentage of mud is more</td>
</tr>
<tr>
<td>OEi(Sc)</td>
<td>Estuary Island</td>
<td>Schist</td>
<td>Land-water boundary -</td>
<td>-</td>
</tr>
<tr>
<td>OE(La)</td>
<td>Exposed Rocks</td>
<td>Laterite</td>
<td>Adjacent to Rocky cliffs</td>
<td>-</td>
</tr>
<tr>
<td>OE(Sc)</td>
<td>Exposed Rocks</td>
<td>Schist</td>
<td>Coast</td>
<td></td>
</tr>
<tr>
<td>OE(Gc)</td>
<td>Exposed Rocks</td>
<td>Greywacke with Conglomerate</td>
<td></td>
<td></td>
</tr>
<tr>
<td>OAs(Oc)</td>
<td>Arabian Sea</td>
<td>Ocean</td>
<td>Ocean</td>
<td>Ocean</td>
</tr>
</tbody>
</table>

4.5 Shoreline changes

Shoreline is one of the rapidly changing landform of the earth. The shoreline comprises a major element of the earth’s landscape and the procedures that shape it are exceptionally complex (Pethick, 1984). The actual definition of shoreline, mapping and using them is a complicated task (Nayak, 2002). According to Li Ron et al. (2001), the shoreline is a most unique feature of the earth surface. It is one of the 27 features recognized by the International Geographic Data Committee (IGDC) and a rapidly changing landform in the coastal area. It provides a more detailed picture of shoreline change through time and of how adjacent shore types evolve in concern with the associated coastal landforms. The geomorphic processes of erosion & sedimentation, periodic storm, flooding and sea level changes continuously modify the shoreline (Nayak, 2002). The rising and falling sea level along with the waves, tides and sediment available to them develop a great range of coastal landform. These forms can be related to the variation in the energy level, structural stability of the area, waves & tides and combination of these processes. The accurate demarcation and monitoring of shoreline (long-term, seasonal and short-term changes) are...
necessary for understanding of coastal processes. The historical and functional approaches to study shoreline changes along with various landforms help in deciphering the coastal processes operating in an area (Shaikh et al., 1989, Nayak, 2000).

4.5 Shoreline extraction

The shorelines from satellite remote sensing data can be extracted from many methods i.e. visual image interpretation, semi-automatic or automatic segmentation techniques (White et al., 1999 and Dellepiane et al., 2004), but there is no any single method, which can be considered good for all multi-temporal, multi-sensors and multi-spatial resolution satellite images (Pal and Pal, 1993). The conceptual flowchart for the shoreline change analysis and mapping is shown in Figure 6. The methodology used for the extraction of the shoreline is based on automatic segmentation classification techniques as well as visually image interpretations (manually digitization) to delineate land-water boundary for six different periods i.e. 1973, 1989, 1999, 2006, 2009 and 2014 (Figure 7). For calculation of shoreline rate-of-change statistics for above stated time series, the ESRI ArcGIS software based Digital Shoreline Analysis System (DSAS) version 4.3 extensions has been used. By using this tool, the shoreline change rate has been calculated by End Point Rate (EPR) for short term and Weighted Linear Regression (WLR) for long term period.

4.6 Calculation of shoreline change rates

Rates of shoreline change have been generated by using ArcGIS-10.3 with the Digital Shoreline Analysis System (DSAS) v4.3, an ArcGIS tool developed by the USGS (Thieler et al., 2009). DSAS tool has been used to generate transects starting from a reference baseline and intersecting the shoreline positions at 20m intervals. A number of transects were used to calculate the rates along the coastal area of Tiswadi taluka using two approaches i.e. End Point Rate (EPR) for short term and Weighted Linear Regression (WLR) for long term period. The end-point-rate (EPR) was calculated by dividing the distance of shoreline movement by the time elapsed between the earliest and latest measurements i.e. the oldest and most recent shoreline. The weighted-linear-regression-rate (ELR) was calculated by plotting the shoreline positions with respect to time. The WLR statistical method is more reliable because it takes into account the uncertainty field to calculate the long-term rates of shoreline...
change. Annual rates of changes by using End Point Rate (EPR) for short term and Weighted Linear Regression (WLR) for long term period are summarized in Table 5.


<table>
<thead>
<tr>
<th>S. No</th>
<th>Calculation Approach for Shoreline Rate-of-Changes</th>
<th>No. of Transects Used</th>
<th>Average Erosion Rate (m/year)</th>
<th>Average Accretion Rate (m/year)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>End Point Rate (EPR) for Short-term Period</td>
<td>7266</td>
<td>(-) 0.73</td>
<td>1.22</td>
</tr>
<tr>
<td>2</td>
<td>Weighted Linear Regression (WLR) for Long-term Period</td>
<td>8820</td>
<td>(-) 0.21</td>
<td>0.53</td>
</tr>
</tbody>
</table>

4.7 Erosion and accretion area mapping

Accretion and erosion are two inverse processes. Accretion increases the area of agricultural activity, forest, mangroves, salt pans, sand beach and coastal plains, while erosion causes the loss of land, destroys constructed buildings, houses and bullying human lives. The result of the analysis has shown the erosion areas through time (Table 6).

![Figure 6 Conceptual flowchart for the shoreline change analysis mapping](image-url)
Shore is often eroded robustly in well-aired zones and when it has direction perpendicular to northeast and southeast wind. Most of accretion/erosion sections in 1973-2006 are continued in the next stage in 2006-2014 with the higher and more rapid measure. The result shows that there is spatially significant changes of all shoreline either erosion or accretion trends (Figure 8).

**Figure 7** Shoreline mapping for six different periods from 1973 to 2014

**Figure 8** Erosion and accretion area mapping
Table 6 Erosion and accretion area

<table>
<thead>
<tr>
<th>S. No.</th>
<th>Year</th>
<th>Length of Shoreline (km)</th>
<th>Analysis between</th>
<th>Accretion (sq km)</th>
<th>Erosion (sq km)</th>
<th>Total Growth (sq km)</th>
<th>Growth Rate (sq km/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1973</td>
<td>76.024</td>
<td>1973-1989</td>
<td>1.636</td>
<td>4.045</td>
<td>-2.409</td>
<td>-0.150</td>
</tr>
<tr>
<td>2</td>
<td>1989</td>
<td>80.241</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>3</td>
<td>1999</td>
<td>79.213</td>
<td>1989-1999</td>
<td>1.277</td>
<td>0.886</td>
<td>0.390</td>
<td>0.039</td>
</tr>
<tr>
<td>4</td>
<td>2006</td>
<td>80.290</td>
<td>1999-2006</td>
<td>0.439</td>
<td>1.496</td>
<td>-1.057</td>
<td>-0.151</td>
</tr>
<tr>
<td>5</td>
<td>2009</td>
<td>80.346</td>
<td>2006-2009</td>
<td>0.777</td>
<td>0.224</td>
<td>0.552</td>
<td>0.184</td>
</tr>
<tr>
<td>6</td>
<td>2014</td>
<td>79.916</td>
<td>2009-2014</td>
<td>0.965</td>
<td>0.259</td>
<td>0.705</td>
<td>0.141</td>
</tr>
</tbody>
</table>

|       | 79.338* | 41         | 0.124*          | 0.168*           | -0.044*        | -0.001*             |                        |

* Average shoreline length
* Average accretion and erosion area /year
* Average accretion and erosion growth /year
* Average accretion and erosion growth rate /year

During 1973-1989 periods about 4.04 sq km of land was eroded from the area, but between 1973 and 1989, accretion is about 1.63 km². From 2006 to 2009, the accretion increased to 0.78 sq km from 0.44 sq km and decreasing of erosion was to 0.22 sq km from 1.49 sq km. The average erosion area per year from 1973 to 2014 was 0.168 sq km, while the average accretion area per year for the same period was 0.124 sq km. Average accretion-erosion growth per year was -0.044 and average accretion-erosion growth rate per year was -0.001 as shown in Table 6.

5. Conclusion

The recent systematic tools of remote sensing and GIS are very useful for shoreline change analysis and coastal geomorphological studies. In this study, the coastal processes in Tiswadi taluka of Goa state, the drainage network, soil, geology, geomorphology and shoreline change were analysed using remote sensing and GIS tools. It can be concluded that remote sensing and GIS may be beneficial for qualitative monitoring of shoreline change as demarcation of erosion and accretion area in the absence of ground data. DSAS tool, an extension of ArcGIS software has improved the efficiency for calculation of rate-of-change statistics from multi historic shoreline data. The study notices both erosion and accretion in isolated pockets all along the Tiswadi coast. It is estimated that the erosion from 1973 to 1989 was 4.05 sq km, 1989 to 1999 was 0.88 sq km, 1999 to 2006 was 1.49 sq km, 2006 to 2009 was 0.23 sq km, and 2009 to 2014 was 0.26 sq km respectively, while the accretion from 1973 to 1989 was 1.63 sq km, 1989 to 1999 was 1.27 sq km, 1999 to 2006 was 0.43 sq km, 2006 to 2009 was 0.78 sq km, and 2009 to 2014 was 0.96 sq km respectively. The combined approach using remote sensing and GIS tools evidently sheds lights on both the cause and reasons for shoreline change. The outcomes of this study would be useful for shoreline, coastal zone management and implementation of CRZ act in the study area. We conducted a statistical analysis of the database to identify patterns of erosion and accretion and to make quantitative estimates of the rates of shoreline changes. Such rates provide information that can be used to understand the magnitude and trends of shoreline changes and can also provide a basis for various coastal zone management strategies. The methodology adopted in this study can be adapted to other countries.

6. Acknowledgement

We are profoundly thankful to our Guru Ji Prof. J. L. Jain, who with his unique research competence, selfless devotion, thoughtful guidance, inspirational thoughts, wonderful patience and above all parent like direction and affection motivated us to pursue this work.

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Comparison of an Upflow Anaerobic Sludge Blanket and an Anaerobic Filter for Treating Wheat Straw Washwater

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Abstract: The study compared the performance of upflow anaerobic sludge blanket (UASB) reactors and anaerobic filters (AF) for the treatment of wheat straw washwater (WSW) which has a high concentration of Potassium ions. The trial was conducted at mesophilic temperatures (37 °C). The digesters were started up over a 48-day period using a synthetic wastewater feed and reached an organic loading rate (OLR) of 6 g COD L⁻¹ day⁻¹ with a specific methane production (SMP) of 0.333 L CH₄ g⁻¹ COD. When the feed was switched to WSW, it was not possible to maintain the same loading rate; the SMP in all reactors fell sharply to less than 0.1 L CH₄ g⁻¹ COD, with the AF affected more than the UASB. On reducing the OLR to 3 g COD L⁻¹ day⁻¹ the reactors recovered to produce 0.21 L CH₄ g⁻¹ COD added and exhibited 82% COD removal. A discrepancy between the COD consumed and the methane produced could be accounted for through increased maintenance energy requirement of the microbial community for osmoregulation as K⁺ was found to accumulate in the sludge and in the UASB reached a concentration of 4.5 mg K g⁻¹ wet weight of granules. Overall, anaerobic digestion of WSW was found to be one of the promising techniques in generating clean energy as one tonne of wheat straw would approximately generates 4148 L of CH₄.

Keywords: Anaerobic digestion, osmotic stress, chemical oxygen demand, specific methane production.

1. Introduction

Straw from cereal production has attracted attention for many years as a potential biomass fuel source, and there is interest in using it as a combustible fuel in both large and small-scale plant for heat and power production. The abundance of straw, its short rotation period and intensive culture systems have led to considerable interest from researchers in exploring efficient methods for its utilization as an energy source (He et al., 2008). Wheat straw is a major potential source of waste biomass for renewable energy production, but its high salt content causes problems in combustion. In the UK, according to Defra Statistics: UK Cereal Production Survey (accessed 18 May 2012), of 12.2 million tonnes of straw (wheat, barley, oats and oil seed rape), more than half is from wheat which comprised 7.7 million tonnes. Based on the findings from Biomass Futures project, it is estimated that in the UK more than 5 million tonnes of straw (dry matter), equivalent to approximately 15 million tonnes wet weight, will be available by year 2020 (Elbersen et al., 2012). Nevertheless, the major problem associated with the use of straw in thermal energy production is the low fusion temperature ash which can cause fouling during the combustion process. The high salt content, particularly K and Cl, can also give rise to acidic gases with associated boiler corrosion problems.

Pre-treatment techniques such as washing can improve the quality of the straw for thermal processing by reducing the concentrations of the salts K and Na as well as Cl (Jensen et al., 1997). Sander (1997) reported that the leaching process allows rapid removal of K because it is not associated with the structural components of plants. Although salts can be removed from straw in a relatively simple process, the saline wheat straw washwater (WSW) obtained requires treatment before disposal since in addition to the leached salts, it is also rich in organic matter. Anaerobic digestion (AD) has been identified as an option for treatment of this washwater, since it has a low operating cost, low sludge production, and the methane produced could be used as an energy source. Several studies have looked at the application of anaerobic treatment

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for the removal of organic pollutants in highly saline wastewaters, including the treatment of tannery wastewater, seafood processing effluent, fishery and fish farm wastewater effluent. The presence of excessive amount of salts, particularly Sodium, has been revealed as inhibitory for anaerobic wastewater treatment at concentrations of more than 10 g L⁻¹ due to the effect of osmotic shock (Lefebvre and Moletta, 2006a). The toxic effect of high Na concentrations has been reported by many researchers, but little attention appears to have been given to the effects of high concentrations of K, which is the salt most likely to contribute to salinity in straw washings. Kugelman and McCarty (1965) and Mouneimne et al. (2003) investigated the toxicity of K in an acetate batch assay test, while Fernandez and Foster (1994) demonstrated the threshold of K inhibition using glucose feed substrate in batch study. Habert et al. (1997) demonstrated the treatment of inuline effluent using UASB at 10 g L⁻¹ of salt at OLR of 23-32 kg COD m⁻³ day⁻¹ and achieved 65% - 85% of COD removal rate.

The use of non-halophilic groups in an upflow AF has been demonstrated by Guerrero et al. (1997) during the treatment of seafood processing effluent at 15 g L⁻¹ of salts and revealed 83% of COD removal at COD influent of 34 g L⁻¹ and OLR of 2.8 kg COD m⁻³ day⁻¹. Furthermore, Boardman et al. (1995) also reported on the treatment of seafood processing wastewater using UASB and discovered 77% of COD removal at 7.7 to 26.3 g L⁻¹ of salt. The OLR applied in the study was at 13.6 kg COD m⁻³ day⁻¹ and COD influent of 1.7 g L⁻¹. Anaerobic digestion has not worked well in all such cases; however, and there are reports of saline wastewaters being inhibitory in AD systems when employing non-halophilic methanogens. Mosquera Corral et al. (2001) investigated the efficiency of a Hybrid Anaerobic Sludge Bed Filter in treating seafood processing effluent which contained 1.5 g L⁻¹ of influent COD concentration at OLR of 1.52 kg COD m⁻³ day⁻¹. The findings showed about 70%-90% of COD removal at HRT of 18 hrs. Application of an Upflow Anaerobic Sludge blanket at OLR of 0.5 kg COD m⁻³ day⁻¹ has also been reported by Lefebvre et al. (2006b) during the treatment of tannery wastewater at 71 g L⁻¹ of salts and found 78% of COD removal at COD influent of 2.3 g L⁻¹. Vidal et al. (1997) investigated the performance of AF in treating seafood processing wastewater at salt concentration of 30 g L⁻¹. This study revealed the COD removal efficiencies remained at 70% with an Organic Loading Rate (OLR) ranging from 1 to 15 kg COD m⁻³ day⁻¹.

Idrus et al. (2012) assessed biogas potential from wheat straw washed water using UASB. This study confirmed that the organics washed from wheat straw were to a large extent biodegradable in UASB digesters with a specific methane yield of 0.29 l CH₄ g⁻¹ COD extracted and 84% COD removal. Nevertheless, the achievement was only temporary when both specific methane yield and COD removal dropped to 0.12 l CH₄ g⁻¹ COD extracted and less than 50%, respectively. Studies on the treatment of saline wastewater indicate that salt concentration is not the main indicator in determining the level of inhibition. Apart from pre adapted sludge and population of the bacteria, the presence of other cations and OLR also need to be considered. These statements are supported by Feijoo et al. (1995) who noted that performance in treating saline wastewater in AD system depends on nutrient in the feedstock, previous adaptation of the sludge, antagonistic or synergistic effect (due to the presence of other cations) and lower substrates to biomass ratio used. Numerous studies on saline wastewater have highlighted the role of Na in contributing to inhibition in AD systems. The effect of other salts, particularly K, has been given relatively little attention. Microorganisms accumulate cations, including ionic K, and/or low-molecular-weight organic compounds known as compatible solutes from the surrounding medium to achieve an osmotic equilibrium. Most cells also maintain [K⁺(in)] >> [K⁺(out)] (Schönheit et al., 1984; Sprott et al., 1985). Inhibition or dehydration occurs when K⁺ in the extracellular solute concentration exceeds K⁺ in the cell cytoplasm (Lai and Gunsalus, 1992).

Fernandez and Foster (1994) investigated the threshold of K inhibition using glucose feed substrate (batch study) and observed that low concentrations of K (less than 100 mg L⁻¹) facilitated performance in both the thermophilic and mesophilic conditions, while at higher concentrations (greater than 2500 mg L⁻¹) toxicity of K was significant. Mouneimne et al. (2003) investigated the toxicity of K in an acetate batch assay test and found the toxicity threshold to be 0.43 mol L⁻¹. Kugelman and McCarty (1965) reported that 5.87 g L⁻¹ of K caused 50% inhibition of acetate utilizing methanogen. Most of the K toxicity thresholds which have been reported were conducted in batch studies, which implied that further investigation is required in continuous system. Idrus et al. (2012) reported on potential of biogas production from WSW in a continuous system of UASB, and the aim of this study was therefore to compare the potential of biogas production from UASB and AF in treating WSW at higher OLR and to establish the highest methane production that could be achieved in either of these types of digesters.

2. Method

2.1 Feedstocks

A synthetic wastewater (SW) was prepared as a concentrate (Table 1) and diluted to the required working strength using tap water. The working strength was determined by the required organic loading rate (OLR) based on measured chemical oxygen demand (COD). The COD of the SW concentrate was approximately 50 g L⁻¹. Wheat straw leachate was prepared by chopping up wheat straw into approximately uniform length pieces which were placed inside the mesh drum of a reciprocating-action washing machine. A quantity of water in a known
ratio to the dry weight of the straw was then added to the drum trough and the temperature was raised to 60 °C for 3 hours, afterwards the liquid was drained off.

### Table 1 Composition of synthetic wastewater concentrate

<table>
<thead>
<tr>
<th>Component</th>
<th>Unit</th>
<th>Quantity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trace element solution (mg L⁻¹: Fe 10; Co, Mn, Ni and Zn 1; Cu, B, Al, Mo, and Se 0.1)</td>
<td>mL</td>
<td>1</td>
</tr>
<tr>
<td>Yeast (block baker’s form)</td>
<td>g</td>
<td>23</td>
</tr>
<tr>
<td>Urea</td>
<td>g</td>
<td>2.14</td>
</tr>
<tr>
<td>Full cream milk (UHT sterilised)</td>
<td>mL</td>
<td>144</td>
</tr>
<tr>
<td>Sugar (granulated white)</td>
<td>g</td>
<td>11.5</td>
</tr>
<tr>
<td>Dried blood</td>
<td>g</td>
<td>5.75</td>
</tr>
<tr>
<td>Ammonium phosphate</td>
<td>g</td>
<td>3.4</td>
</tr>
<tr>
<td>Tap water</td>
<td></td>
<td>Make up to 1 L</td>
</tr>
</tbody>
</table>

2.2 Experimental set-up

The experimental work used two anaerobic filters (AF1 and 2) and two UASB reactors (U1 and 2), each with a working volume of 1.5 L and maintained at 37 °C. The anaerobic filters were inoculated using salinity-acclimated digestate from a laboratory-scale digester which had been fed daily on SW supplemented with a mix of Potassium Chloride and Sodium Chloride (KCl + NaCl). Three litres of this digestate was divided evenly between AF1 and AF2 and recirculated through the filter for 7 days using a peristaltic pump. The UASB reactors were each inoculated with 1 kg wet weight of granular sludge, which had been stored at ambient temperature and then brought up to operating temperature and maintained at this for 7 days, whilst the liquid fraction was recirculated through the granular bed. SW was then fed into each of the digesters using a variable speed peristaltic pump (Model 505S, Watson Marlow, UK), Feeding commenced at a low OLR and then gradually increased over a 48-day period reaching a final OLR of 6 g COD L⁻¹. During this start-up phase the hydraulic retention time (HRT) was maintained more or less constant at 18-19 hours by gradually reducing the dilution of the SW. The effluent from the digesters was collected in a sealed container, which was connected to a gas sampling bag, allowing separation of the biogas generated from the liquid effluent stream. Biogas production was determined by measuring the quantity of biogas in the sample bag using a weight-type water displacement gasometer (Walker et al., 2009), and then subtracting the volume of effluent collected in the inter-connected receiver vessel. All gas volumes are reported at a standard temperature and pressure (STP) of 101.325 kPa and 0 °C.

2.3 Analytical methods

Biogas composition (CH₄ and CO₂) was determined using a Varian star 3400 CX gas chromatograph (GC), calibrated with a standard gas of 65.12% CH₄ and 34.88% CO₂ (v/v). Conductivity was measured using a LF330 meter (WTW GmbH, Germany). pH was measured using a Jenway 3010 pH meter (Bibby Scientific Ltd, UK) with a combination glass electrode calibrated in buffers at pH 4, 7 and 9 (Fisher Scientific, UK). Suspended solids were determined according to Standard Method 2540 D (Environment Agency Q1 2007), using glass fiber filter discs (GFC) (Whatman, UK). COD was measured using a closed tube digestion and titration [put author surname here not a number as per JHER style]. Total Organic Carbon (TOC) was measured using a Dohrmann TOC (DC-190) based on Standard Method 5310 (APHA, 2005). Ammonia was determined using a Kjeltech block digestion and steam distillation unit according to the manufacturer’s instructions (Foss Ltd, Warrington, UK). K, Mg, Na and Ca in leachate samples were analyzed by first filtering the sample (Whatman No 1) and then diluting it into 12.5% of nitric acid (HNO₃). The acidified samples were analyzed using a Varian Spectra AA-200 atomic absorption spectrometer (Varian Ltd, UK), according to the manufacturer’s instructions.

3. Results and discussion

3.1 Characterization of wheat straw leachate

The two batches of WSW had the following characteristics (mg L⁻¹): batch 1 - COD 4500; K 638; Na 28; Mg 352; Ca 367; Zn 43; Cl 432; SO₄²⁻ 484; PO₄³⁻ 86; and conductivity 3554 µS cm⁻¹. Batch 2 - COD 2250; K 317; Na 9; Mg 112; Ca 123; Zn 14; Cl 136; SO₄²⁻ 182; PO₄³⁻ 26; and conductivity 1114 µS cm⁻¹. The cation present at the highest concentration in the leachate was thus potassium, in agreement with the results of Jensen et al. (1997).
3.2 Digester start-up

Figure 1 shows the applied OLR for all four digesters. Minor fluctuations were due to slight variations in pumping rates. Specific methane production (SMP) in the AF was initially slightly lower than in the UASB reactors, indicating that the former needed some additional time for acclimatization, but by day 25, all digesters had achieved SMP values of around 0.32 l CH$_4$ g$^{-1}$ COD$_{added}$ (Figure 2), and COD removal rates of 95%.

All four digesters responded well to the rapid increase in OLR up to around 6 g COD L$^{-1}$ day$^{-1}$, although U1, which received a marginally higher loading due to variability in pump flow rates, showed a marginally lower SMP. Between day 25-40 the pH in U1 and 2 dropped below 7, but recovered to around 7.4 by day 48 (Figure 2).

![Figure 1 Applied OLR and SMP in UASB and AF digesters. Vertical dotted lines indicate the change from SW to WSW, and the OLR reduction from 6 to 3 g COD L$^{-1}$ day$^{-1}$](image)

3.3 Wheat straw leachate

On day 48, the feed was switched to WSW at an OLR of 6.0 g COD L$^{-1}$ day$^{-1}$. As the organic strength of the WSW was greater than that of the SW, this involved reducing the volume added each day and increasing the HRT. At this point, the SMP fell sharply in all reactors (Figure 2), with the AF affected more than the UASB, and U2 more than U1. The effluent COD concentration also rose sharply to around 4 g COD L$^{-1}$ in all the digesters, corresponding to a fall in removal rates to around 35% (Figure 2) although there were some signs of recovery in the following days. On day 55, the OLR was therefore reduced to 3.0 g COD L$^{-1}$ day$^{-1}$ by diluting the WSW with tap water to give an influent concentration of around 2.5 g COD L$^{-1}$ and a HRT of around 26 hours: these values were fairly close to those for the SW feed. The SMP in both UASB reactors improved over the next 25 days, with U2 showing an immediate peak in gas production in response to the reduced load. The AF digesters responded more slowly, but by day 90 the SMP in all the digesters had recovered to around 0.21 l CH$_4$ g$^{-1}$ COD$_{added}$ with a biogas methane concentration of around 73%. By day 69, COD removal was close to 75% in all the digesters, and by the end of the run had stabilized at around 83%.

Effluent VFA concentrations (not shown) remained extremely low throughout the experimental run (measured values < 10 mg L$^{-1}$), indicating that any peaks in effluent COD consisted of unhydrolysed material and/or other intermediate products. Effluent suspended solids concentrations also showed little variation throughout the experiment (Figure 2). During the period of feeding on WSW, the pH in the AF was slightly higher and more stable than in the UASB, which operated at just below pH 7 for much of the time. The ammonia concentration in the effluent of all the four digesters stabilised at around 0.1 g N L$^{-1}$ as shown in Figure 2, indicating that little buffering was available from this source.

In terms of performance parameters (SMP, COD and suspended solids removal), the two systems tested thus appeared to perform equally well. During the last 10 days of operation, the SMP was stable at 0.20 l CH$_4$ g$^{-1}$ COD$_{added}$ and COD removal rate was 82% for all the digesters (Table 2). Comparisons between UASB and AF reactors have been carried out in the past, but not using wheat straw washwater. In a study by Hutnan et al. (1999), a synthetic wastewater with a COD of 6 g L$^{-1}$ was used. The loading was successfully increased to 15 g COD L$^{-1}$ day$^{-1}$ and comparable COD removals were seen for the different reactor types. This loading is higher than that applied in this study, but the strength of the wastewater [use surnames] was higher and the study was thus not affected by reductions in HRT which may become critical when dealing with lower strength wastewaters.
Comparison of Upflow Anaerobic Sludge Blanket and an Anaerobic Filter
Idrus et al.

Figure 2 pH, COD and COD removal, suspended solids removal and ammonia concentrations in UASB and AF during experiment. Vertical dotted lines indicate the change from SW to WSW, and the OLR reduction from 6 to 3 g COD L\(^{-1}\) day\(^{-1}\).

Table 2 Average values for performance parameters in last 10 days of experimental run

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>AF1</th>
<th>AF2</th>
<th>U1</th>
<th>U2</th>
</tr>
</thead>
<tbody>
<tr>
<td>SBP</td>
<td>l biogas g (^{-1}) COD added</td>
<td>0.290</td>
<td>0.292</td>
<td>0.288</td>
<td>0.297</td>
</tr>
<tr>
<td>SMP</td>
<td>l CH(_4) g (^{-1}) COD added</td>
<td>0.212</td>
<td>0.212</td>
<td>0.212</td>
<td>0.214</td>
</tr>
<tr>
<td></td>
<td>l CH(_4) g (^{-1}) COD removed</td>
<td>0.256</td>
<td>0.257</td>
<td>0.257</td>
<td>0.260</td>
</tr>
<tr>
<td>COD removal</td>
<td>%</td>
<td>82.5</td>
<td>82.2</td>
<td>82.4</td>
<td>82.2</td>
</tr>
</tbody>
</table>

3.4 Accumulation of K in digesters

Table 3 and Figure 4 show the accumulation of K in the digesters during the experimental period, calculated on a mass balance basis from influent and effluent K concentrations. At an OLR of 6 g COD L\(^{-1}\) day\(^{-1}\), rates of accumulation were around 0.3 g K day\(^{-1}\). When the OLR was reduced by diluting the WSW, this rate fell dramatically to 0.033 and 0.034 g K day\(^{-1}\) in U1 and U2. Accumulation in the AF showed some variation with time, but in the last 35 days of operation, the rates in AF1 and AF2 were 0.024 and 0.031, respectively. The rates in the two reactor types were surprisingly similar, and showed no obvious sign of falling by the end of the run. The final values for K accumulation equated to an additional concentration of 4.5 mg K g\(^{-1}\) wet weight of granular sludge inoculum.

Osmo-regulation in Archaea has been previously described by Roberts (2004) and this group of organisms is known to be able actively to pump K\(^{+}\) across the cell membrane. Under these circumstances, additional energy in the form of ATP is required to maintain osmo-regulation as reported by Mitchell (1973), Kashket (1985), Gober and Kashket (1986) and a small proportion of carbon is converted to meet this need.
reducing slightly the COD that can be metabolised to CH₄. In the current study the average SMP per g COD removed in the UASB reactors at the end of the experimental period was calculated to be 0.259 L CH₄ g⁻¹ COD_removed. This corresponded to around 74% of the theoretical methane yield of the COD removed, or a shortfall of 0.029 L CH₄ g⁻¹ COD_removed.

<table>
<thead>
<tr>
<th></th>
<th>AF1</th>
<th>AF2</th>
<th>U1</th>
<th>U2</th>
</tr>
</thead>
<tbody>
<tr>
<td>K accumulated by day 56 (g)</td>
<td>2.35</td>
<td>2.52</td>
<td>2.73</td>
<td>2.78</td>
</tr>
<tr>
<td>K accumulated by day 115 (g)</td>
<td>4.46</td>
<td>4.74</td>
<td>4.74</td>
<td>4.92</td>
</tr>
<tr>
<td>(mg g⁻¹ WW of original inoculum)</td>
<td>4.46</td>
<td>4.74</td>
<td>4.74</td>
<td>4.92</td>
</tr>
<tr>
<td>Accumulation rate at OLR 6 (g K day⁻¹)</td>
<td>0.281</td>
<td>0.314</td>
<td>0.349</td>
<td>0.336</td>
</tr>
<tr>
<td>Accumulation rate at OLR 3 (g K day⁻¹)</td>
<td>0.024*</td>
<td>0.031*</td>
<td>0.033</td>
<td>0.034</td>
</tr>
</tbody>
</table>

* Average for last 35 days

Figure 3 K accumulation in UASB and AF digesters. Vertical dotted lines indicate the change from SW to WSW, and the OLR reduction from 6 to 3 g COD L⁻¹ day⁻¹.

4. Conclusion

The comparison of a UASB digester with an anaerobic filter for the treatment of WSW showed no difference in the COD removal or the specific methane production for these two process types. An OLR of 6 g COD L⁻¹ day⁻¹ could not be maintained with WSW, and the highest steady loading achieved was 3 g COD L⁻¹ day⁻¹ at an influent concentration of 2.5 g COD L⁻¹, giving a HRT of around 26 hours. When fed on WSW the digesters accumulated K; the rate of accumulation and final amounts accumulated in both the UASB and AF were found to be very similar and had reached an additional concentration of between 4.5-4.9 mg K g⁻¹ wet weight of granular sludge. The findings of this study will be useful in similar studies in other countries as it is also applicable for the other types of straw such as rice straw, barley straw, oat straw and soybean straw.

5. Acknowledgements

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References


Bio-Chemical Separations and Purification of Heavy Metal from Industrial Waste Water: A Review on Adsorption and Precipitations

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Abstract: The discharge of heavy metals into aquatic ecosystems has become a matter of concern over the last few decades. In small quantities, certain heavy metals are nutritionally essential for a healthy life. Heavy metals become toxic when they are not metabolized by the body and accumulate in the soft tissues. Heavy metals may enter the human body through food, water, air, or absorption through the skin when they come in contact with humans in agriculture and in manufacturing, pharmaceutical, industrial and residential settings. To address this situation, this study presents a review on how the reduction of heavy metal from our surrounding can be achieved in particular by adsorption and precipitation methods including the type of adsorbent, characteristics of adsorbent and effectiveness in treatment. It has been found that these methods are quite effective in removing harmful heavy metals from wastewater. These methods generally use natural low cost materials like activated carbon, sawdust chitin and azadirachta indica (neem). These materials can be used effectively in developing countries like Ethiopia as they are quite inexpensive.

Keywords: Adsorption; precipitation, activated carbon, chitin, neem.

1. Introduction

Nowadays, environmental pollution is a global problem; it has been an active area of research and scientists are struggling for a solution. The sources of environmental contaminants include geological weathering, mining effluents, industrial effluents, and uses of chemicals like fertilizers (Atkinson et al., 2008). Generally industrial or municipal wastes, containing different chemicals, are disposed to water bodies such as lakes, oceans and rivers as water is considered as a universal solvent. The chemical contamination of drinking water leads to health problems primarily through chronic exposure, as it may persist for years before detection (Bosnic et al., 2000). In many countries, the major chemical pollutants of surface water come from industrial and municipal sewages as many cities lack sufficient waste treatment facilities (ESID, 2003). Sometimes the corrosion of the urban water supply system also contributes heavy metal contamination of water (Toprak, 1994; Fabiani et al., 1996; Andualem, 2005). The preservation and maintenance of natural water resources is a burning issue. The quality of water resources is deteriorating day by day due to continuous discharge of municipal and industrial effluents to water systems. On the other hand, the demand for safe water is increasing continuously due to the increase in population, living standard and industrialization.

The discharges from many industrial wastes contain various organic and inorganic contaminants including higher level of toxic heavy metals like Pb, Cd, Zn and Cu. Among the various water bodies river water is the most exposed one for pollution due to the direct discharge of municipal and industrial effluents to many rivers. Groundwater in the vicinity of drain systems can be polluted by toxic metals contamination from industrial, domestic, mining and others waste discharges (ESID, 2003). The incorporation of toxic metal ions in human organs through drinking water and various food chains may cause serious health problems.

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Since metals are biologically no degradable they tend to accumulate in various vital organs; therefore, even exposure to trace concentration of various metal ions can lead to long term toxic effect (UNEP 2010). Toxic concentration of trace metals, like zinc, lead, cadmium, copper and others, interfere with the normal metabolic process of fish and other species life in marine environment. These metals can be passed onto humans who consume them and pose adverse effects on the human metabolic process. National and international regulations on food quality have lowered the maximum permissible levels of toxic metals in human food; hence an increasingly important aspect of food quality should be to control the concentration of trace metals in food (Deng et al, 2006). The health and welfare of people, which is intimately connected with the quality of water for drinking and for common household purposes, should control the level of heavy metals in different water supplies (Monser and Adhoum, 2002). These days, there is a great interest to the investigation of toxic heavy metals in environmental samples especially in industrial wastes, as they do have potential for contamination of surface and groundwater supplies (Agarwal et al., 2006).

Among the various metal ions Pb, Cd and Hg are toxic at all concentration levels and have no known functions in animal bodies. The metal ions of Cu, Zn, Co and Fe are required for physiologival and cellular activities, but toxic above a certain level (Bosnic et al., 2000; Toprak, 1994; UNEP 2010; Deng et al 2006). Toxic metals as they are non-degradable and bio-accumulative, cause tissue degradation in nature (Monser and Adhoum, 2002; Agarwal et al., 2006). Mercury (Hg) both elemental and organic mercury are highly toxic to the central nervous system and kidney, especially in children. Brain damage can also be observed in developing fetus if the mother is exposed. It is a highly toxic metal; it has no known beneficial effects in humans (Machel and Gregorious, 1948). This paper presents a review of various adsorption and precipitation processes to remove heavy metals from wastewater. At the beginning, treatment studies by adsorption are discussed, followed by precipitation. A number of case studies are also presented.

2. Treatment by adsorption

Adsorption is a process in which a single or a group of ions/compounds get accumulated on the surface of another solid or liquid. The substance on which the adsorption takes place is known as adsorbent and the substance which gets adsorbed is called adsorbate. Due to ease of operation, the adsorption techniques have been used widely to treat metal ion containing wastewater. Several contender technologies exist to eliminate heavy metals from industrial wastewater, including reduction followed by adsorption on miscellaneous adsorbents, supercritical fluid extraction, ion exchange, ion flotation, electrocoagulation, high-gradient magnetic separation and enhanced ultrafiltration (Murphy and Erkey, 1997). Most methods seem to be almost equally effective. However, further consideration of suitability places a large portion of them outside of industrial scale applications due to the high capital investment and operational costs involved. Specifically speaking, the most effective and versatile technique for heavy metal removal, even at very low concentrations, is adsorption. However, the high price of adsorbents (usually activated carbon) is regarded as the major obstacle for industrial application. From an economic point of view, it is not feasible to utilize activated carbon for industrial wastewater treatment. To that end, the focus of heavy metal adsorption studies has been changed towards natural materials that are available in vast amounts, as well as certain waste products from industrial and agricultural operations (Monser and Adhoum, 2010). Among other processes for removal of Cr (VI) from industrial wastewater, biosorption process is an economically feasible alternative. The major advantages of biosorption over other conventional treatment methods are: (i) low cost and easy availability of adsorbent; (ii) utilization of industrial, biological and domestic wastes as adsorbent; (iii) low operational cost and high efficiency of metal removal from dilute solution; (iv) ease of operation compared to other processes and regeneration and recycling of the biosorbent; (v) ability to remove complex form of metals that is generally not possible by other conventional methods.

Based on the extent of attraction between the adsorbent and adsorbate, the adsorption process can be classified into two types: physical adsorption and Vander Waal’s adsorption and Chemisorption. Adsorption, which can result from the Vander Waal’s force of interaction, is known as physical adsorption or VanderWall’s adsorption. In this type of adsorption, the process heat is of the order of 20-40 kJ/mol. Physical adsorption process is reversible and established rapidly. Physical adsorption can be of two types, i.e., monolayer adsorption and multi-layer adsorption. In chemisorption, the chemical interaction/electrostatic force of attraction occur between the adsorbent surface and adsorbate molecules. In this process, the heat of adsorption usually varies from 40 to 400kJ/mol. It is associated with appreciably high activation energy and therefore termed as activated adsorption. It is relatively a slow process (Sundar, 2010). Physical adsorption is a reversible process that occurs at a temperature lower or close to the critical temperature of an adsorbed substance. On the other hand chemisorption, in general, is an irreversible process because of strong electrostatic force of interaction between the adsorbent and adsorbate molecules. Physical adsorption is very effective, particularly at a temperature close to the critical temperature of a given fluid. Chemisorptions occur usually at temperatures much higher than the critical temperature and by contrast to physical adsorption, is a specific process, which can only take place on some solid surface for a given fluid. Contrary to physical adsorption, chemisorption leads to monolayer adsorption. Under favorable conditions, both the processes
can occur simultaneously or alternately. Physical adsorption is accompanied by a decrease in free energy and entropy of the adsorption system; this process is exothermic in nature (Samuel et al., 1954).

Most of the adsorbents in industrial processes have complex porous structures that consist of pores of different sizes and shapes. The total porosity is usually classified into three groups: micropores (d < 2 nm), mesopores (2 < d < 50 nm) and macropores (d > 50 nm). The significance of pore size in the adsorption process is well known. Because the size of micro pores are comparable to those of adsorbate molecules, all atoms or molecules of the adsorbent can interact with the adsorbate species. This is the fundamental difference between adsorption in micro, meso and macro pores. Consequently, the adsorption in micro pore is essentially a pore filling process in which the pore volume is the main controlling factor.

In case of mesopores whose walls are formed by a large number of adsorbent atoms or molecules, the boundary of the inter-phase has a distinct physical meaning. That means the adsorbent surface area has a physical meaning. The adsorption forces do not occur throughout the void volume in macropores, but at a close distance from their walls. Therefore, the mono and multilayer adsorption take place successively on the surface of mesopores and their final fill proceeds according to the mechanism of capillary adsorbent condensation. The basic parameters characterizing mesopores include specific surface area, pore volume and pore size or pore-volume distribution. The mechanism of adsorption on the surface of macro pores does not differ from that of the flat surfaces. The specific surface area of macro-porous solid is very small; that is why adsorption on this surface is usually neglected. The capillary adsorbate condensation does not occur in macro pores (Dubinin and Stoeckli, 1980). Various forms of chemical adsorbents and materials of biological origin or biosorbent have been shown to be effective metal removers from the industrial wastewater.

2.1 Biosorption of heavy metals

Based upon the metal binding capacities of various biological materials, biosorption can separate heavy metals from wastewater. Lately, biosorption has emerged as a cost-effective and efficient alternative for application to low strength wastewaters. Biosorption, a term used to describe the removal of heavy metals using a passive binding process with nonliving microorganisms including bacteria, fungi, and yeasts, and other biomass types that are capable of efficiently collecting heavy metals (Vilar et al., 2007; Pavasant, 2007; Parvathi and Nagendran, 2007). Obviously, some of the advantages biosorption have over conventional treatment methods include low cost, high efficiency for dilute solutions, a minimal amount of chemical and/or biological sludge, no additional nutrients required and the possibility of biosorbent regeneration and metal recovery (Vilar et al., 2007). The sorption of heavy metals onto these biomaterials is attributed to their constituents, which are mainly proteins, carbohydrates and phenolic compounds, since they contain functional groups such as carboxylates, hydroxyls and amines, which are able to attach to the metal ions (Pavasant, 2007). Heavy metal accumulation in aquatic organisms, which is an active process involving metabolic activity within living organisms, has been studied by several researchers since 1978 (Wong, 1978; Volesky, 1978; Volesky and Holan, 1995; Volesky, 2003; Gutnick and Bach, 2000; Ahluwalia and Goyal, 2007). At present, the biosorption field has been enriched by a vast amount of studies published in different journals. Although in the beginning most researchers focused their efforts upon heavy metal accumulation and concentration within living organisms, upon noticing that dead biomass possesses high metal-sorbing potential, their interest shifted to biosorption. This is mainly due to the abundant availability, adsorption capacity and economic value of agricultural, plant tissue and industrial wastes, which has made them the main subject for biosorption research during the last decade (Shukla, 2006).

Several reviews are available that discuss the use of biosorbent for the treatment of water and wastewater containing heavy metals. The most encountered mechanism is the ion exchange mechanism. Recognition of ion exchange between protons and heavy metals, contribution of surface functional groups in ion exchange processes and ion exchange isotherm development are presented. Reliable sorption equilibria and kinetic data are of substantial value for adsorption separation system design. A majority of biosorption studies have been devoted to this end. Furthermore, a combination of basic equilibrium sorption data, sorbate behavior in solution, sorbate uptake mechanisms, mass transfer characteristics and fluid dynamic characteristics is required for the design of adsorption systems (Volesky, 2003).

Other researchers also provide several reviews that are worthy of mention. For instance, structural modifications of bacteria through alteration of their polysaccharide backbone or side chains or through sugar modification for biosorption purposes have been performed by many research groups. The discussion focus on mainly, cation binding to specific bacterial biopolymers, binding of cations by biopolymers, molecular approaches to biopolymer modification and preparations, and also potential application for metal sequestration. This review presents useful information on polymer engineering applications within the biosorption area. The remark drawn is that the benefits associated with enhanced specificity and cation-binding capacity should be examined to judge the economic competitiveness of polymer engineering for industrial
applications (Gutnick and Bach, 2000). Furthermore, studies about the applications of microbial and plant derived biomass to sequester metal ions from solutions were presented. Some conventional techniques utilized to remove heavy metals from industrial effluents, as well as the advantages and disadvantages of biosorption processes and an outline of the use of different surface and structural characterization methods, such as SEM (scanning electron microscope), and X ray diffraction method (EDAX), to determine biosorption mechanisms, were mentioned (Ahluwalia, and Goyal, 2007).

Sawdust, being cheap and easily available, is used widely either as such or in treated form to remove metal ions from wastewater. Sorption studies were also carried out mostly in batch scale and various adsorption parameters affecting the overall process were studied (Shukla et al., 2002). The role of saw dust for wastewater treatment has been given emphasis by some researchers. The adsorption mechanism, influencing factors and favorable conditions for different sawdust materials were discussed (Shukla et al., 2002; Bailey et al., 1999). A statistical review on the biosorption of Cd (II), Cr(IV), Ni(II), Pb(II) and Zn(II) using different classes of algae was provided. The focus was mainly on the biochemistry of biosorption, which revolved around a detailed description of the macromolecular conformation of the alginate biopolymers, metal binding mechanisms and the role of cellular structure, storage polysaccharides and cell wall and extracellular polysaccharides on metal sequestration (Romera et al., 2006; Davis et al., 2003). The state-of-the-art review in the field of heavy metal biosorption employing Saccharomyces cerevisiae was also compiled. The result showed that yeast-utilized biosorption depends largely on pH, the initial metal ion to biomass concentration ratio, culture conditions, the presence of various ligands and competitive metal ions in solution, and, to a more limited extent, temperature (Wang and Chen, 2006).

2.2 Use of activated carbon

The well known adsorbent which is in application in industrial scales is the activated carbon, which can remove metal ions from wastewater. The activated carbon is made from various raw materials having high carbonaceous materials, including wood, saw dust, coconut shell etc. They can be activated by thermal decomposition in a high-temperature oxidation or lower temperature chemical dehydration reaction. The activated carbon is being used widely to treat wastewater to remove organic or inorganic pollutants because of their large specific surface area, high adsorption capacity and special surface chemical properties. These physical and chemical properties of the activated carbon depend on pore size, pore distribution and number of surface oxygen groups. The pore size and pore volume can be controlled during the activation process such as activation time, activation agent and temperature. The surface oxygen can also be changed by using suitable oxidizing agents and thermal treatment in order to get the surface functional groups such as carboxyl, phenolic and lactonic group attached to carbon. These groups can improve the adsorption capacity and selectivity on a certain adsorbate in the gaseous or liquid phase (Yunhai et al., 2008; Schneider et al., 2007; Lodeiro et al., 2006).

2.3 Roles of living organisms

Waste microorganisms usually in the form of dead cells are also used as an alternative adsorbent for the treatment of heavy metal containing wastewater. In this process biological materials accumulate heavy metal materials from wastewater by either metabolically mediated or purely physico-chemical pathways of uptake. The microorganisms can take up metal ion in numerous pathways. The uptake of heavy metal ions can take place by entrapment in the cellular structure and subsequent sorption on to the binding sites present in the cellular structure (Pavasant, 2007). Mucor hiemalis was also reported to remove Cr(VI) from solutions. The detailed studies were made with regards to its kinetics and mechanism of adsorption. The competitive biosorption of Fe(III) and Cr(VI) on C. vulgaris from binary mixtures was investigated in a single stage batch reactor by varying the solid/liquid ratio at an initial pH of 2.0. The batch adsorption was assumed to be a single stage equilibrium operation (Volesky, 1978).

A number of fresh water macrophytes like aeromonas caviae was reported to treat Cr(VI) contaminated wastewater. The adsorption studies were carried out in a stirred reactor. Equilibrium and kinetic experiments were carried out for various parameters like bulk concentration, biomass load, temperature and ionic background. The isotherm followed monolayer Langmuir model. The adsorption process followed chemical sorption. The Cr(VI) reducing capacity was enhanced by modifying the amino and carboxyl group (Volesky, 2003).

2.4 Roles of agricultural and cellulosic materials

Agricultural by-products are mostly composed of lignin and cellulose, as well as other polar functional group-containing compounds, which include alcohols, aldehydes, ketones, carboxylates phenols and ethers. These groups are able to bind heavy metals through replacement of
hydrogen ions with metal ions in solution or by donation of an electron pair from these groups to form complexes with metal ions in solution (Ahluwalia and Goyal, 2007).

Adsorption studies were also carried out using various agricultural crop wastes. In these studies, adsorption analyses were carried out using different reactor types. Various adsorption parameters were studied to evaluate their effects on Cr(VI) removal efficiency. Among the adsorption parameters, pH was observed to be an important factor in determining the adsorption efficiency. Adsorption kinetics was observed to be reasonably faster and followed dual rate, i.e. initial faster rate followed by slower one. The initial faster and latter slower rates might be due to surface and intra particle diffusion processes, respectively (Lodeiro et al., 2006).

An adsorption study was made on *Tamarindus indica* on the capability of Cr(VI) removal by Agarwal et al., (2006). It was concluded that the use of tamarind seed as an alternative adsorbent is more economical and worthwhile than other conventional methods. The removal of Cr(VI) ions by tamarind seed is significantly reduced with a pH increase, slightly decreased with ionic strength enhancement and enhanced with rising temperature. It is suggested that chemisorption is the most plausible mechanism involved (Agarwal et al., 2006).

Azadirachta indica (neem) is a typical tree in the mahogany family of Meliaceae. Products derived from neem have proven to possess therapeutic value such as anthelmintic, antifungal, antibacterial, antiviral, anti-infertility and sedative activities. Several researchers have investigated the prospects of neem leaf as an alternative adsorbent for Cr(VI), Cd(II) and Pb(II). Azadirachta indica (Neem Leaf Powder) was utilized to extract chromium (VI) from solution. The adsorption was carried out in a batch process using different concentrations of metal ions in aqueous solution in parallel with variation in the adsorbent amount, pH, agitation time and temperature. The suitability criteria of the adsorbent were checked by the Langmuir and Freundlich isotherms, and also by various equilibrium kinetic data. On the basis of the Langmuir model, an adsorption capacity of neem leaf powder for Cr(VI) of 0.0028 mmol/g was obtained. A relatively low amount of *Azadirachta indica* (Neem Leaf Powder) (1.6 g/ dm$^3$) could remove as much as 87% of Cr(VI) from a dilute solution (0.137 mmol/l) during a 300 min period at 300K. The optimum pH range in the study was 4.5–7.5. By modifying the physical structure and surface chemistry of the neem leaf via an activation process, the adsorption capacity could be significantly enhanced (Babu and Gupta, 2008; Sharma and Bhattacharyya, 2004).

Chitin, the waste polymer from fishery industry, was reported to be a good adsorbent for Cr(VI). It is white, hard inelastic material containing nitrogenous polysaccharides derived from the outer skeleton of insects, crabs, shrimps and other marine animals. Chitin is the second most abundant natural polymer i.e. polysaccharide and its estimated annual production is almost equal to cellulose. Chitin is converted to chitosan by alkaline hydrolysis using 50% (W/W) aqueous NaOH solution. Chitosan has many applications due to the presence of reactive –NH$_2$ group and -OH groups. Due to the presence of these functional groups, chitosan is a good chelator and forms complexes with almost all heavy metal ions. Further, due to its cationic nature, it adsorbs various anionic species. The solubility of chitosan in aqueous acids is a limiting factor for many such applications. It is therefore necessary to crosslink chitosan to render it insoluble in acid media. Chitosan is generally cross-linked using chemical reagents such as glutaraldehyde and epichlorohydrin (Schneider et al. 2007).

It was revealed that a major rice bran fraction contains 12%-13% oil and highly un-saponifiable components. In view of its abundant availability and ease of retrieval as an unused by-product, the possibility of using this material as an alternative adsorbent for heavy metal removal from synthetic wastewater was conceived. The study also explored the influence of parameters such as pH, temperature, and particle size and adsorption time on sorption capacity. The results underscore the importance of pH on heavy metal sorption capacity, while to a certain extent; the capacity also relies upon sorbent size (Grimm et al., 2008). With the aim of increasing biosorption capacity, modifications of the biosorbent surface have been carried out. For instance, thiolation of coconut fiber, alteration of the surface properties of pine bark using the Fenton reagent, alteration of the surface of olive pomace by phosphoric acid and hydrogen peroxide (Igwe et al., 2008; Argun and Dursun, 2008).

2.5 Chemical adsorbents

Apart from biomaterials, a number of chemical adsorbents were used to treat Cr(VI) contaminated water by Ghosh and Goswami (2005). Stannous hydroxide was reported to have good Cr(VI) adsorptive capacity. Various adsorption parameters were studied to evaluate their effect. The adsorption process followed first order kinetics (Ghosh and Goswami, 2005). Hydroxalate was also used to treat Cr(VI) contaminated water. Hydroxalate, Mg$_6$Al$_2$(OH)$_{16}$CO$_3$·4H$_2$O (HT), is a double layered mixed metal hydroxide. Due to positive charge of HT, it is a potential adsorbent for various anions. Cr(VI) adsorption studies were carried out by using manganese nodule leach residue. Since the leached residue has a high surface area, it is a potential adsorbent. Its adsorption behavior was studied as a function of contact time, pH, temperature, concentrations of adsorbate and adsorbent dose. The adsorption process was endothermic in nature (Lazaridis, 2003). Metallurgical wastes like...
red mud was also reported to be a good adsorbent for Cr(VI). Activated red mud was prepared by acid dissolution followed by ammonia precipitation and drying at 110 °C. The process was optimized by varying the adsorption parameters (Pradhan et al., 1999). Various other chemical adsorbents were used to treat Cr(VI) contaminated water such as clay, zeolites, feldspar, hydrated zirconium oxide, hydrous titanium oxide and ion exchange resins (Marchioretto et al., 2009).

Apart from process development in small scale, adsorption studies were also carried out in different reactors, which helped in scaling up the laboratory/bench scale data to pilot or industrial scale. The main requirement of a large scale sorption process is that the sorbent should be used in a suitable reactor configuration such as packed or fluidized bed or stirred tank reactor. Such large scale adsorption in reactors requires suitable adsorbent in granulated or pellet form. The size, density and shape of the particles should prevent clogging or large pressure drops across the adsorbent bed (in case of packed bed column) and at the same time, should permit optimum flow. Continuous adsorption of metal ions from solution can be accomplished by employing different types of reactor configurations. The reactors commonly employed for wastewater treatment include packed bed, fluidized bed, expanded bed, stirred tank reactor and moving column reactor etc. (Marchioretto et al., 2009).

3. Precipitation methods to remove heavy metals

Precipitation is the formation of a solid in a solution during a chemical reaction or by diffusion in a solid. When the reaction occurs in a liquid, the solid formed is called the precipitate, and the liquid remaining above the solid is called the supernatant. Natural methods of precipitation include settling or sedimentation, where a solid form over a period of time due to ambient forces like gravity or centrifugation. During chemical reactions, precipitation may also occur particularly if an insoluble substance is introduced into a solution and the density happens to be greater (otherwise the precipitate would float or form a suspension). With soluble substances, precipitation is accelerated once the solution becomes supersaturated. An important stage of the precipitation process is the onset of nucleation. The creation of a hypothetical solid particle includes the formation of an interface, which requires some energy based on the relative surface energy of the solid and the solution. If this energy is not available, and no suitable nucleation surface is available, supersaturation occurs (Veeken et al., 2012).

When the precipitate has low solubility products, supersaturation is favored and consequently the nucleation rate increases, as the particle sizes decrease. The particle size of precipitates depends on the competition between nucleation and crystal growth. When the nucleation rate dominates the crystal growth rate, colloidal particles (<0.05 mm) will result. If the initial concentration of the reactant is high, consequently increasing the concentration of the precipitate, which has a low solubility, the resulting high supersaturation leads to high rates of a primary nucleation. In fact, after filtration, the liquid was a bit turbid and this is an indication of the high nucleation rate. Common reagents used include alkalis, such as lime, magnesia, NaOH, NaHCO₃, Na₂CO₃. Heavy metals like Cr, Cu, and Zn do not precipitate at pH below 7, permitting some possibility for separation from ferric iron and aluminum which precipitate at pH below 6.5. The association of heavy metal ions with a ferric or aluminum hydroxide precipitate is controlled by adsorption or co-precipitation. Co-precipitation implies the simultaneous removal of a metal ion during the formation of the primary metal precipitates. In addition to common alkalis, sulfides such as Na₂S, H₂S, NaHS, or FeS can also be used to precipitate metals and have been applied frequently for metal removal from waste effluents to achieve discharge standard limitations.

The lower solubility of metal sulfides in the acid region below pH 7 permits reduction of metal solubility to values that are orders of magnitude lower than are attainable by hydroxide precipitation (Marchioretto et al., 2009). Hydroxide precipitation is the most frequently applied treatment technology for removing heavy metals from industrial wastewaters and polluted ground waters. Disadvantages of hydroxide precipitation include the relatively high residual effluent metal concentrations (0.5 - 2 mg/ L), the interference of chelating compounds, and unsuitability of metal sludge reuse in case more metals are present. A well-known alternative is sulfide precipitation. Sulfide precipitation is superior to hydroxide precipitation for removal of heavy metals from wastewaters as it results in lower effluent concentrations (< 0.01 mg/ L) and less interference from chelating agents. However, sulfide precipitation is not widely applied in practice because the dosing of sulfide cannot adequately be controlled, and excess sulfide in the effluent is toxic and corrosive. Moreover, precipitation by sulfide results in colloidal metal sulfide precipitates that are poorly separable from the water phase by sedimentation or filtration (Veeken et al., 2012).

Nowadays, a combination of hydroxide and sulfide precipitation for optimal metals removal is being considered. A common configuration is a two-stage process in which hydroxide precipitation is followed by sulfide precipitation with each stage followed by a separate solids removal step. This could produce the high quality effluent similar to that obtained with the sulfide precipitation process. Moreover, it will reduce the consumption of sulfide reagents, which are usually more expensive than hydroxide reagents, and will result in a smaller amount of sludge compared to the hydroxide precipitation process alone. The drawbacks of the chemical precipitation process are related to the amount of
of the stable oxidation and removal of hexavalent chromium was also enhanced compared to Cr(VI). The two ethyl dithiocarbamate (SDDC) immobilised at their surface has been some evidence that organic compounds can decrease its sorption. The effect of addition of negatively-charged biosurfactants (rhamnolipids) on chromium contaminated kaolinite was studied. Results showed that the rhamnolipids have the capability of extracting 25% percent of the stable form of chromium, Cr(III), from the kaolinite, under optimal conditions. The removal of hexavalent chromium was also enhanced compared to water by a factor of 2 using a solution of rhamnolipids. Results from the sequential extraction procedure showed that rhamnolipids remove Cr(III) mainly from the carbonate and oxide/hydroxide portions of the kaolinite. The rhamnolipids had also the capability of reducing close to 100% of the extracted Cr(VI) to Cr(III) over a period of 24 days. This study indicated that rhamnolipids could be beneficial for the removal or long-term conversion of chromium Cr(VI) to Cr(III) (Hafez et al., 2007).

### Modified Activated Carbon for the Removal of Copper, Zinc, Chromium and Cyanide from Wastewater

Modified activated carbon is carbonaceous adsorbents which have tetrabutyl ammonium iodide (TBAI) and sodium diethyl dithiocarbamate (SDDC) immobilised at their surface. Monser and Adhoum (2002) investigated the adsorption of toxic ions, copper, zinc, chromium and cyanide on these adsorbents that have undergone surface modification with tetrabutyl ammonium (TBA) and SDDC in wastewater applications. The modification technique enhance the removal capacity of carbon and therefore decreases cost-effective removal of Cu(II), Zn(II), Cr(VI) and CN− from electroplating unit wastewater. Two separate fixed bed modified activated carbon columns were used; TBA-carbon column for cyanide removal and SDDC-carbon column for multi-species metal ions (Cu, Zn, Cr) removal. Wastewater from electroplating unit containing 37 mg/l Cu, 27 mg/l Zn, 9.5 mg/l Cr and 40 mg/l CN− was treated through the modified columns. A total CN− removal was achieved when using the TBA-carbon column with a removal capacity of 29.2 mg/g carbon. The TBA-carbon adsorbent was found to have an effective removal capacity of approximately five times that of plain carbon. Using SDDC-carbon column, Cu, Zn and Cr metal ions were eliminated with a removal capacity of 38, 9.9 and 6.84 mg/g, respectively. The SDDC-carbon column has an effective removal capacity for Cu (four times), Zn (four times) and Cr (two times) greater than plain carbon (Monser and Adhoum, 2002).

### Removal of Chromium (VI) from Wastewater by Combined Electrocoagulation—Electroflotation without a Filter

A combined electrocoagulation and electroflotation process was designed by Gao et al. (2005) to reduce Cr6+ to Cr3+ and then to remove the total Cr from wastewater to a value below 0.5 mg/l. Acidic condition was employed in the reduction of Cr6+ and neutral conditions were found to be beneficial for the coagulation of the precipitates of Cr(OH)3 and Fe(OH)2. The formation of Fe(OH)3 was ensured by sparging compressed air in the coagulation unit through a draft tube. The air not only oxidizes Fe2+ produced electrically, but also helps to mix the water for a better coagulation of the particles. The two-stage electroflotation arrangement can separate the solids from the wastewater to a value of less than 3 mg/L with total Cr less than 0.5 mg/L. The residence time required is about 1.2 h. The optimal conditions for the treatment are: charge loading of about 2.5 Faradays/m3 water, pH value in the coagulation unit is 5–8. The power consumption is less than 1 kwh/m3 water at the conductivity of 1.5 mS/cm (milli simens per cemetimeter). When aluminum ions are either added or produced in situ in the coagulation unit, the treated wastewater can be discharged without any filtration. (Gao et al., 2005).

### Effects of Rhamnolipids on Chromium-Contaminated Kaolinite

Hexavalent chromium Cr(VI) is a common environmental pollutant that is treated by its reduction to the trivalent form Cr(III). The latter can be re-oxidized to the toxic form, Cr(VI), under specific conditions. A study was conducted by Hafez et al. (2007) on the removal of Cr(III) to eliminate the hazards imposed by its presence in soil as there has been some evidence that organic compounds can decrease its sorption. The effect of addition of negatively-charged biosurfactants (rhamnolipids) on chromium contaminated kaolinite was studied. Results showed that the rhamnolipids have the capability of extracting 25% percent of the stable form of chromium, Cr(III), from the kaolinite, under optimal conditions. The removal of hexavalent chromium was also enhanced compared to water by a factor of 2 using a solution of rhamnolipids. Results from the sequential extraction procedure showed that rhamnolipids remove Cr(III) mainly from the carbonate and oxide/hydroxide portions of the kaolinite. The rhamnolipids had also the capability of reducing close to 100% of the extracted Cr(VI) to Cr(III) over a period of 24 days. This study indicated that rhamnolipids could be beneficial for the removal or long-term conversion of chromium Cr(VI) to Cr(III) (Hafez et al., 2007).
Trivalent Chromium Removal from Wastewater Using Low Cost Activated Carbon Derived from Agricultural Waste Material and Activated Carbon Fabric Cloth: A low cost activated carbon (ATFAC) was prepared by Mohan et al. (2006) from coconut shell fibers (an agricultural waste), characterized and utilized for Cr(III) removal from water/wastewater. A commercially available activated carbon fabric cloth (ACF) was also studied for comparative evaluation. All the equilibrium and kinetic studies were conducted at different temperatures, particle size, pH, and adsorbent doses in batch mode. The Langmuir and Freundlich isotherm models were applied. The Langmuir model best fit the equilibrium isotherm data. The maximum adsorption capacities of ATFAC and ACF at 25 °C are 12.2 and 39.56 mg/g, respectively. Cr(III) adsorption increased with an increase in temperature (10 °C: ATFAC 10.97 mg/g, ACF 36.05 mg/g; 40 °C: ATFAC 16.10 mg/g, ACF 40.29 mg/g). The kinetic studies were conducted to delineate the effect of temperature, initial adsorbate concentration, particle size of the adsorbent, and solid to liquid ratio. The adsorption of Cr(III) follows the pseudo-second-order rate kinetics. From kinetic studies various rate and thermodynamic parameters such as effective diffusion coefficient, activation energy and entropy of activation were evaluated (Mohan et al., 2006).

Removal of Lead and Chromium from Wastewater Using Bagasse Fly Ash - A Sugar Industry Waste: An inexpensive and effective adsorbent was developed by Gupta and Ali (2004) from bagasse fly ash, obtained from a sugar industry, for the dynamic uptake of lead and chromium. Lead and chromium are sorbed by the developed adsorbent up to 96–98%. The removal of these two metal ions up to 95–96% was achieved by column experiments at a flow rate of 0.5 ml/min. The adsorption was found to be exothermic in nature. The adsorbent was successfully tried for the removal of lead and chromium from wastewater in the laboratory scale. The developed system for the removal of two ions was found to be useful, economic, rapid, and reproducible (Gupta and Ali, 2004).

4. Conclusion

Presence of heavy metals in the environment is of great concern as these affect human health negatively. Wastewater contains many harmful heavy metals, which can be bio-accumulated and enter into the food chain. This paper presents a review on the removal of heavy metals from wastewater by adsorption and precipitation methods. It has been found that these methods are quite effective in removing harmful heavy metals from wastewater. These methods generally use natural low cost materials like activated carbon, sawdust chitin and azadirachta indica (neem). These materials can be used effectively in developing countries like Ethiopia as they are inexpensive. Further research is recommended to enhance the knowledge of heavy metal treatment by cheaper materials and means to achieve a sustainable environment.

References


Potential Migration of Leachate from an Active Landfill: Spatial Analysis of Groundwater Quality in Communities Surrounding the Dompoase Landfill, Kumasi, Ghana

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Abstract: This study applied physico-chemical and spatial techniques to investigate potential transport of leachate from the Dompoase Landfill site in Ghana. Groundwater contamination gradient from the Landfill site provided evidence of leachate transport and potential impact on adjoining communities. Static water level (SWL), pH and electrical conductivity (EC) of well water, as well as nitrate (NO3-), phosphate (PO43-) and lead (Pb) concentrations showed negative correlation with distance from the Landfill. The negative gradient of SWL suggests relatively deeper water table in areas proximate to the Landfill. For the other parameters, the negative gradient provided evidence that the contamination potentially emanated from the Landfill, but appeared to attenuate naturally with distance. Nickel (Ni) and mercury (Hg) however behaved differently spatially. While the concentration of Ni in groundwater did not change with distance from the Landfill, Hg showed a positive gradient. Presumably, there were other sources of contaminations from Hg other than the Landfill. A sharp contrast existed in groundwater quality between communities located at the east and west ends of the Landfill. The groundwater quality was suitable for drinking at the eastern end, but relatively poor at the western end. Potentially, the groundwater flow and contaminant transport was influenced by a shallow aquifer, which maintained a contamination gradient that affected communities located west of the Landfill.

Keywords: Landfill, leachate, spatial interpolation, pollution.

1. Introduction

Landfills remain an integral part of any solid waste management system. Although rightly consigned to the tail end of the solid waste management hierarchy, it is relevant for the fact that eventually components of the waste stream that cannot be recycled, as well as residues from other waste treatment processes must all be landfilled. Three types of landfills can be identified – open dumps, semi-controlled landfills and the sanitary landfills (Remigios, 2010). Open dumps in the strict sense are not landfills, but land depressions where solid wastes are dumped. They abound in developing countries and have huge environmental related issues. The semi-controlled landfills are also a common feature in some developing countries that lack engineered landfills. The semi-controlled landfills may provide certain improvement over the open dumps, such as compaction of dumped refuse and daily covering with topsoil to prevent nuisance. Nevertheless, because it is not engineered, there is leachate discharge and emissions of landfill gases. The sanitary landfills on the other hand are properly engineered with facilities to intercept and treat leachates and also control landfill gases.

Despite its perceived environmental integrity, threats of groundwater contamination from sanitary landfills have been widely reported (Mikac et al., 1998; Abu-Rukah and Al-Kofahi, 2001; Ahmed and Sulaiman, 2001; Srivastava and Ramanathan, 2008; Han et al., 2013). It is apparent from

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these studies that sanitary landfills may yield groundwater contamination problems, especially of shallow aquifers. The risk of contamination is potentially high in the direction of groundwater flow, and contamination could stretch to distances of about 6 km from the landfill (Abu-Rukah and Al-Kofahi, 2001).

Presently, there are only two engineered landfills in Ghana, one at Tamale in the Northern Region of Ghana and the other (Dompoase Landfill) in Kumasi, the second largest city in Ghana. The capital city, Accra, lacks an engineered landfill because of “not in my backyard” (NIMBY) syndrome. In 2007, an attempt to build an engineered landfill in Kwabenya, a suburb of Accra was met with stiff resistance from residents, such that the plan was abolished. This was against the backdrop of poor management of waste disposal sites in Ghana. Even with those landfills that were built, there are clear operational issues. For instance, the operation of the Dompoase Landfill has come with very overwhelming challenges as people who reside close to it and the general public has put up stern resistance in an effort to protect their right to live in a healthy environment (Owusu-Sekyere et al., 2013).

One of the operational difficulties of sanitary landfills in Ghana concerns the fact that the various fractions of the waste stream, from biodegradable organic materials to plastics, metals and bottles are all indiscriminately landfilled. This is precipitated by the lack of source separation mechanisms and poor culture of recycling. In the absence of source separation, minimal amount of wastes is captured for recycling, and virtually everything discarded is eventually landfilled. Based on an estimated population of 25 million and an average daily waste generation per capita of 0.45 kg, Ghana currently generates about 4.5 million tons of solid waste annually (Ofori-Boateng et al., 2013). About 65% of this tonnage is made up of decomposable organic materials, which could be recycled, but unfortunately are mostly landfilled. Landfilling such huge volumes of decomposable materials tend to speed up the rate at which landfills may fill up. Secondly, decomposable organic wastes, if not diverted from landfills would generate a lot of leachate and greenhouse gas problems.

Percolation of leachate through soil can affect underlying aquifer and surface water surrounding the landfill. Contamination of surface and ground water by landfill leachate raises serious public health and environmental concerns. It is therefore of grave concern in Ghana that estates are springing up proximal to landfills and waste dumpsites. Many of the households located around the Dompoase Landfill in Ghana access groundwater for drinking purposes, via individual boreholes and hand-dug wells and risk sourcing water that is contaminated by leachate from the landfill (Owusu-Sekyere et al., 2013). A recent study, applying geophysical techniques, suggested potential subsurface migration of leachates from the Dompoase Landfill Facility (Boateng et al., 2013). The aim of this study was to investigate the potential of leachate contaminant transport from the Dompoase Landfill site and its impact on the quality of groundwater in surrounding communities, applying physico-chemical and spatial techniques. Specifically, the study (i) evaluated water quality parameters (such as pH, electrical conductivity, nitrate, nitrite, phosphate, lead, nickel and mercury) of hand-dug wells in communities surrounding the Dompoase Landfill site in Ghana, and (ii) spatially resolved the extent of contamination in groundwater as a function of distance from the Landfill site to help establish safe and unsafe drinking water zones.

2. Materials and Methods

2.1 Description of the Dompoase Landfill site

The Dompoase Landfill is an engineered sanitary landfill belonging to the Kumasi Metropolitan Assembly (KMA). It is a 100 acre facility located on the Oti Stool Lands and near the Dompoase community, about 10 km from the central business district of Kumasi (Figure 1). It serves for solid waste disposal and treatment and has a septage treatment plant attached. The Landfill commenced operation in 2004 and receives industrial, institutional, commercial and domestic waste without separation. It is anticipated to have a life span of 15 years. It receives all forms of solid waste from various transfer stations in the metropolis. In 2009, the Dompoase Landfill received 72 percent of the total volume of waste generated in the metropolis as compared to the previous year’s figure of 86 percent (KMA-WMD, 2009). This drop reflects the inability of the sanitary landfill to handle the increasing volume of solid waste generated annually in the Kumasi Metropolis.

Each day, solid waste dumped at the site is leveled off using a bulldozer, and then covered with sand and finally compacted using a compacting machine. As is the case in many developing countries, informal recycling is carried out by people who scavenge through the waste looking for scrap metals, plastic bottles or anything they can salvage to sell. There is an extensive liner system that serves as a barrier between the waste and the environment. Leachate collection pipes and drainage channels conduct leachate to a treatment facility, which has ten sedimentation ponds that receive municipal septage for treatment. Landfill leachate is mixed with the septage and after treatment the effluent is discharged into a nearby river (Oda River).
2.2 Sample collection

Groundwater samples were collected from three nearby communities (classified as zones) – Aprahon (DA), Dompoase-Kuwait (DK) and Sokoban (SB), with 10 sampling points in each zone, noting the specific distance of each sampling point from the Landfill (Figure 2). Untreated leachate samples were also collected from the Landfill leachate treatment facility. The samples were collected into plastic bottles that have been acid washed and rinsed with de-ionized water. Samples for heavy metals analysis were preserved using 2 ml conc. HNO₃ acid. All the samples were preserved at 4 °C and transported to the laboratory for analysis. The coordinates of the sampling points as well as their distances from the boundary of the Landfill were determined using a hand-held GPS. The static water level of each of the sampled wells was also determined with the aid of a nylon thread tied to a metallic load and suspended float. The thread was lowered into the water until the float suspended on the surface of the groundwater. The water elevation from the ground surface was then measured using a tape measure.

2.3 Analysis of water sample

The temperature, pH, and electrical conductivity of the collected samples were determined on-site using waterproof multiparameter PCStestrTM 35 (Oakton Instruments, USA). Spectrophotometric analysis was used to determine the nitrate, nitrite and phosphate contents of the samples using DR/2400 Spectrophotometer (Hach Company, USA). Buck Scientific 210 VGP model (flame atomic absorption spectroscopy) was employed to determine the heavy metals with the exception of mercury which was determined by the wet digestion/reduction/cold vapor atomic absorption spectrometry (CVAAS) method using Model Hg-201 semi-automated mercury analyzer. Pb, Ni, Hg, Cd and Cr were first determined in leachate samples. It emerged that Cd and Cr were below detection (0.01 mg/l). Subsequently, only Pb, Hg and Ni were monitored in the groundwater samples. The detection limits of Pb and Hg were 0.001 mg/l and Ni 0.01 mg/l.

2.4 Analysis of data

The data were subjected to descriptive statistics and analysis of variance applying Microsoft Excel 2010 software. Variations corresponding to p-values < 0.05 were considered significant. ArcGIS Version 10.0 was applied to map the spatial variation in groundwater quality around the Dompoase Landfill site using inverse distance weighting interpolation method.
3. Result and Discussion

3.1 Characteristics of leachate

Temperature values of the leachate ranged from 33.90 - 34.10 °C. Although temperature has no direct health implications, it is considered important because of its effects on chemical and biological reactions. The temperature range recorded in this study is considered favourable for bacteria activity in the process of leachate formation and other reactions (Tchobanoglous et al., 1993). The pH values of the leachate samples were almost neutral and ranged from 7.36 to 7.44. The Landfill can thus be said to be in the methanogenic decomposition stage (Environmental Agency of UK, 2002). Similar levels of pH were recorded by Longe and Enekwechi (2007) in an investigation on potential groundwater impacts and influence of local hydrogeology on natural attenuation of leachate at a municipal landfill in Lagos, Nigeria.

The very high EC values recorded for the leachate indicate the presence of large quantities of inorganic materials in the leachate samples since conductivity is a measure of the ions present. The mean EC of leachate in this study (8293.33 ± 23.33 µs/cm) (Table 1) was greater than the reported average value at the Oblogo dumpsite in Accra (Denutsui et al., 2012). Nitrate and nitrite contents in the leachate were very high at average concentrations of 211.67 ± 10.93 and 1.20 ± 0.076 mg/l respectively (Table 1). This could have resulted from oxidation of nitrogenous waste products, largely of plant and animal origin, deposited at the Landfill. Comparatively, the Oblogo dumpsite in Accra, Ghana, recorded lower nitrate concentrations in the range of 5.57 – 6.19 mg/l (Denutsui et al., 2012). Nitrite values were relatively smaller because it is not usually present in significant concentrations except in an oxygen stressed environment, as nitrate is the more stable oxidation state (WHO, 2011).

High amounts of heavy metals (lead, nickel and mercury) were recorded in the leachate. Similar results were obtained by Narrey et al. (2012) when leachate samples from five dumpsites in Accra were analyzed. The presence of lead, for instance, may be from the disposal of plastics and rubber remnants (because of previous application of lead as plasticizers), lead-acid batteries, lead foils such as bottle closures, discarded electronic gadgets and used motor oils unto the Landfill (Woodbury, 1992). Presence of mercury in municipal solid wastes is from sources such as cathodes for production of chlorine and caustic soda, mercury vapour lamps, mirror coatings and boilers. According to Tchobanoglous et al. (1992), the presence of nickel emanates from waste products of stainless steel and nickel alloys.
Table 1 Physico-chemical characteristics of leachate and groundwater samples

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Leachate (sampled from landfill) (n=3)</th>
<th>Groundwater (from communities near landfill)</th>
<th>WHO drinking water guidelines</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Who (n=10)</td>
<td>Dompoase-Kuwait (n=10)</td>
<td>Sokoban (n=10)</td>
</tr>
<tr>
<td>Temperature (°C)</td>
<td>34.00 ± 0.06</td>
<td>28.37 ± 0.33</td>
<td>28.65 ± 0.23</td>
</tr>
<tr>
<td>pH</td>
<td>7.40 ± 0.02</td>
<td>6.09 ± 0.15</td>
<td>5.68 ± 0.18</td>
</tr>
<tr>
<td>EC (µS/cm)</td>
<td>8293.33 ± 23.33</td>
<td>203.31 ± 29.55</td>
<td>687.13 ± 478.91</td>
</tr>
<tr>
<td>Nitrate (mg/l)</td>
<td>211.67 ± 10.93</td>
<td>5.81 ± 0.40</td>
<td>6.54 ± 0.75</td>
</tr>
<tr>
<td>Nitrite (mg/l)</td>
<td>1.20 ± 0.08</td>
<td>0.01 ± 0.001</td>
<td>0.01 ± 0.002</td>
</tr>
<tr>
<td>Phosphate (mg/l)</td>
<td>117.33 ± 36.73</td>
<td>3.39 ± 0.43</td>
<td>3.95 ± 0.81</td>
</tr>
<tr>
<td>Lead (mg/l)</td>
<td>0.13 ± 0.004</td>
<td>0.05 ± 0.01</td>
<td>0.05 ± 0.01</td>
</tr>
<tr>
<td>Nickel (mg/l)</td>
<td>2.16 ± 0.07</td>
<td>1.76 ± 0.11</td>
<td>1.55 ± 0.06</td>
</tr>
<tr>
<td>Mercury (mg/l)</td>
<td>0.342 ± 0.21</td>
<td>0.002 ± 0.0004</td>
<td>0.012 ± 0.01</td>
</tr>
<tr>
<td>Cadmium (mg/l)</td>
<td>&lt; 0.01</td>
<td>N/A</td>
<td>N/A</td>
</tr>
<tr>
<td>Chromium (mg/l)</td>
<td>&lt; 0.01</td>
<td>N/A</td>
<td>N/A</td>
</tr>
</tbody>
</table>

N/A means not analyzed

3.2 Static water levels of experimental wells

Distances of the experimental wells from the leachate sampling point ranged from 527.5 to 2028.4 m. The SWLs of the wells also ranged from 0.37 m to 36.58 m with a mean value of 5.31 m. These are generally hand-dug wells that tapped into shallow aquifers. There was significant difference in SWLs among the three communities (p = 0.002). A weak negative correlation existed between the SWLs of the experimental wells and distance from the Landfill site (Figure 3). The decrease in SWL with distance from the Landfill suggests deeply situated water table in areas closest to the Landfill. For instance, the sampling point which was very close to the Landfill site recorded comparatively very high SWL value (36.58 m). Landfills are generally accommodated at sites where the water table is sufficiently low to prevent contamination of groundwater. This is presumably the reason why areas in close proximity to the Landfill site have relatively high SWL. According to the general landfill design considerations by Ghana Landfill Guidelines (2002), the bottom of landfill should be at least 2 m above the seasonally high groundwater table. Understandably, people living close to the Landfill had to dig deeper to tap groundwater.

3.3 Physico-chemical parameters of groundwater samples

3.3.1 Temperature, pH and EC

The temperature of the groundwater samples ranged from 26.60 to 31.00 °C, with a mean value of 28.15 °C. According to WHO drinking water guideline (2011), temperature impacts on the reaction of most inorganic constituents and chemical contaminants that may affect the quality of the water. For instance, high water temperature enhances the growth of microorganisms. The recorded pH values ranged from acidic to near neutral with mean values of 6.09, 5.68 and 5.66 for Aprabon, Dompoase-Kuwait and Sokoban respectively. The difference in pH among the three communities was not statistically significant (p = 0.14). All the groundwater samples were weakly acidic, which is consistent with studies conducted by Longe and Enekwechi (2007) in groundwater around the Olusosun Landfill base in Lagos, Nigeria. The WHO (2011) recommends a pH range of 6.5 – 8.5; either sides of this range may be too corrosive for both potable use and, particularly, for equipment.
especially in water supply. Although the pH of water usually has no direct impact on consumers, it is one of the most important operational water quality parameters (Denutu et al., 2012).

Generally, the electrical conductivity values of groundwater were relatively low (26.00 – 391 µs/cm) and within the recommended WHO guideline value for drinking water of 1000 µs/cm. There was however two outlying EC values of 999.00 and 4990.00 µs/cm, recorded at sampling points SB02 and DK09 respectively. Sample DK09 which recorded an extremely high EC value was taken from a shallow well at an establishment where cow hides were processed for meat. The hides were preserved applying large quantities of salt, which possibly impacted on groundwater quality at that point.

### 3.3.2 Nitrate, nitrite and phosphate

The concentration of nitrate in the groundwater ranged from 2.20 – 12.50 mg/l. There was a negative correlation between the concentrations of nitrate in groundwater and distance from the Landfill site (Figure 3). The nitrate content in the groundwater was generally low, mostly below the WHO drinking water guideline value of 11.0 mg/l (WHO, 2011). Studies at the Olusosun Engineered Landfill in Lagos, Nigeria, have also recorded low nitrate content (0.12 – 0.43 mg/l) in nearby groundwater sources (Longe and Enekwechi, 2007). Concentration of nitrate decreased with distance from the Landfill site, with a significant variation (p = 0.014) among the zones. We presumed that the source of nitrate contamination originated from the Dompoase Landfill, such that the level of contamination decreased down gradient by attenuation in the direction of groundwater flow. An experimental well at Dompoase Kuwait close to the northern boundary of the Landfill recorded a value of 12.5 mg/l, which was greater than the guideline value. Potentially, nitrate might have migrated from the Landfill into adjoining wells.

Very low nitrite values were recorded in the groundwater with no significant difference among the three zones of the study area (p = 0.267). There was a weak positive correlation between nitrite content of groundwater and the distance from the Landfill site (Figure 3). Very low values of nitrite were recorded because nitrite is not usually present in significant concentrations except in a relatively reducing environment, as nitrate is the more stable form of nitrogen under oxidized conditions (WHO, 2011). Nitrite levels expectedly increased as nitrate decreased with distance from the Landfill site (Figure 3). The content of phosphate in the groundwater was generally low and decreased down gradient from the site (Figure 3). The variation in phosphate concentration among the three zones was statistically significant (p = 0.043), preempted by relatively low concentrations of phosphate in zone 3 (the Sokoban area). Water from the experimental well DK02, which recorded very high nitrate concentration, also recorded a high content of phosphate (10.97 mg/l), far above the WHO drinking water guideline value of 5.00 mg/l.
This could be due to its proximity to the Landfill. In Ghana, large fractions of food, garden and other forms of wastes are often deposited into landfills. The degradation of such wastes would usually liberate nutrients such as phosphates, which might contaminate groundwater (Osei et al., 2011).

### 3.3.3 Heavy metals

Comparatively, lead and nickel contents in the groundwater were generally very high, while mercury was low. Groundwater contamination of heavy metals can result from natural occurrences or because of numerous types of human activities; residential, municipal, commercial, industrial, and agricultural activities can all affect groundwater quality (Balakrishnan et al., 2011). The concentration of lead in the groundwater samples ranged from <0.01 to 0.09 mg/l with mean values of 0.05, 0.05 and 0.02 mg/l for Aprabon, Dompoase-Kuwait and Sokoban, respectively. The difference in mean values between Aprabon and Sokoban, as well as between Dompoase-Kuwait and Sokoban was each statistically significant (p = 0.02). High lead values were observed in the groundwater near the Dompoase Landfill (which could be described as the source of the contaminant), but the concentrations decreased with distance from the Landfill site (Figure 4). Almost the entire area of Aprabon and Dompoase-Kuwait had lead values greater than the WHO drinking water guideline of 0.01 mg/l (Figure 4). Hence, the use of groundwater for drinking purpose in this zone poses a health risk. This is a deviation from other studies elsewhere in Africa, e.g. Olusosun Landfill in Nigeria and Casablanca Landfill in Morocco, in which recorded Pb values of nearby groundwater were below the WHO guideline (Longe and Enekwechi, 2007; Smahi et al., 2012). Intake of high lead concentration is associated with neurodevelopmental effects, mortality (mainly due to cardiovascular diseases), impaired renal function, hypertension, impaired fertility and adverse pregnancy outcomes (WHO, 2011).

Nickel values ranged from 1.34 to 2.57 mg/l and were all greater than the WHO drinking water guideline value of 0.007 mg/l. There was no significant difference in Ni concentrations among the three zones (p = 0.114). Nickel content of the groundwater did not vary with distance from the Landfill site (Figure 4). The fact that nickel content of the groundwater did not show much variation with distance from the Landfill site and virtually all the sites recorded higher values than the WHO drinking water guideline could be an indication of non-point source of contamination, which may have resulted from the geological setting of the study area or buffering effect of the soil (Boateng et al., 2013). Concentration of nickel in drinking-water is normally less than 0.02 mg/l. It is only in special cases of release from industrial or natural nickel deposits in the ground that concentrations in drinking-water may be higher (WHO, 2011).

![Figure 4](image-url)  
Figure 4 Concentration of heavy metals in groundwater with distance from landfill site
The content of mercury in all the samples ranged from <0.001 to 0.41 mg/l. Aprabon and some parts of Dompooase-Kuwait near the Landfill boundary recorded low levels of mercury which were all within the WHO drinking water guideline value of 0.006 mg/l. Other parts of Kuwait and Sokoban recorded comparatively high levels of mercury. Significant difference existed in the concentration of Hg in groundwater between Aprabon and Sokoban (p = 0.0004). Mercury may occur naturally in groundwater, usually at low concentrations (about 2 μg/l) from geological deposits. Elevated mercury in groundwater may result from releases from chemical spills or improper disposal of materials that contain mercury.
Potential Migration of Leachate from an Active Landfill

Sulemana et al.

Journal of Hydrology and Environment Research

56

dry-cell batteries, fluorescent light bulbs, thermostats, wood preservatives and paints (US State Water Resources Control Board, 2009). The mercury content of the groundwater showed a weak positive correlation with distance from the Dompoase Landfill site (Figure 4). That is, areas closer to the Landfill rather had relatively reduced concentrations of mercury, such that significant difference existed in the concentration of Hg in groundwater between Sokoban (relatively remote) and Aprabon (proximate) to the Landfill site. The main source of mercury contamination could therefore not be attributed to the Landfill alone. Industrial activities at Sokoban might also be implicated. The Sokoban area houses the “Wood Village” of the Kumasi Metropolitan Assembly where various furniture production, saw milling, transportation and other industrial activities are prevalent. The use of wood preservatives, especially, might be a possible factor of Hg contamination in this area.

3.4 Spatial analysis of groundwater contamination

Groundwater from communities located at Aprabon (Zone 1) recorded largely near neutral pH; while groundwater collected from wells located at Dompoase-Kuwait (Zone 2) to Sokoban (Zone 3) were relatively acidic (Figure 5A). The spatial distribution of electrical conductivity showed that very large portion of Aprabon have values ranging from 152 to 271 µs/cm with intermittent portions of relatively high EC. Spatially, EC increased gradually away from the west end of the Landfill (Dompoase-Kuwait), but appeared to decrease as one traversed further to Sokoban (Figure 5B). Large portions of the study area recorded comparatively high values of nitrate concentration (Figure 5C). Nitrate concentrations were comparatively very high at sections of the Landfill boundary, but the concentrations generally decreased with distance away from the boundary in the western direction. Almost the entire area of the Aprabon recorded comparatively low concentration of nitrite, with few portions having comparatively high values (Figure 5D). Some portions at Dompoase-Kuwait near the Landfill boundary recorded very low values of nitrite, but the concentrations increased with distance towards Sokoban. Sections of the north-eastern parts of Dompoase-Kuwait and Aprabon recorded low values of phosphate, but concentrations increased with distance subsequently (Figure 5E). In contrast to the north-eastern parts, the western portions of the study area recorded very high values of phosphate, which decreased with distance from the Landfill boundary.

Large portions of the study area recorded comparatively high concentrations of lead, with intermittent spots of relatively elevated concentrations (Figure 5F). Locations distant to the Landfill site had reduced Pb concentration. The spatial distribution of nickel was such that concentrations were relatively greater in the Aprabon district than Dompoase-Kuwait (Figure 5G). Thus, generally, the space with relatively increased nickel concentration was much broader at communities bordering the eastern end of the Landfill. Areas bordering the north and eastern side of the Landfill (Dompoase-Kuwait and Aprabon) recorded relatively low concentration of mercury, with decreased concentration further east to the Landfill. On the contrary, the concentration of mercury appeared to increase spatially to the west of the Landfill (Figure 5H).

An overlay map which considered spatially the combined effect of the various water quality parameters is shown in Figure 6. It was assumed that the suitability of the groundwater as source of drinking water was dependent on the WHO drinking water guideline values of the analyzed parameters. Groundwater quality in the Aprabon district was suited for several purposes including drinking (Figure 6). The water quality characteristics were generally within acceptable range according to the WHO guideline values for drinking. Potentially, leachate migration was not in the direction of the Aprabon community, taken into consideration the topography of the area and SWLs, which suggested a groundwater flow from east to west of the study area. The spatial impression of overlay of the physicochemical parameters suggested that groundwater quality in areas around Dompoase-Kuwait to Sokoban were potentially unsuited for drinking, but may be applied for other domestic purposes such as cleaning and washing.

Figure 6 Overlay of groundwater quality parameters based on WHO guideline values
4. Conclusion

The Dompoase Landfill in Kumasi, Ghana, handles the enormous waste generated in the Kumasi Metropolis. Despite its benefits, the operation of the Landfill raises serious public health and environmental concerns, particularly, the impact of leachate on the quality of groundwater in surrounding communities. Leachate from the Dompoase Landfill showed elevated levels of EC, nitrate, nitrite, phosphate and specific heavy metals (lead, nickel and mercury). The static water levels of the experimental wells showed weak negative correlation with distance from the Landfill down gradient in the direction of sub-surface water flow. Similar trend was observed for pH, NO₃⁻, PO₄³⁻ and Pb. It was concluded that natural attenuation might be taking place as contaminants migrated in groundwater away from the Landfill. On the other hand, Hg showed a positive correlation with distance from the Landfill, which suggested other sources of contamination for this metal. The spatial analysis of groundwater contamination revealed clear patterns of groundwater suitability for drinking. Groundwater at Aprabon was suited for drinking. At Dompoase–Kuwait and Sokoban however, groundwater was of relatively poor quality and unsuitable for drinking without treatment, but may be used for other domestic, agricultural and industrial purposes. With the exception of mercury, it was concluded that leachate from the Dompoase Landfill was potentially the main source of contamination in nearby groundwater. As most of the contaminants in the groundwater appeared to attenuate with distance from the Landfill, it is important to create a considerable spatial buffer zone around landfills in Ghana to protect nearby communities. The findings from this study provide important empirical data for management of the Dompoase Landfill as well as policy makers to devise pragmatic policies aimed at reducing the adverse impacts of the landfill on groundwater quality in the surrounding communities.

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Impact of Sea Level Rise in Bangladesh: A Socio Engineering Overview

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Abstract: Sea Level Rise (SLR) is currently a burning issue for the coastal zone of Bangladesh. The main cause of the SLR is the changes in climate conditions. Approximately 28% of the total populations of this country are under threat of being ‘climate refugee’ due to SLR. The projection of SLR has raised the concern of permanent inundation, land erosion, salinity intrusion, fresh water and food security along with existence of ecosystems in the coastal zone. This study emphasizes the present and future scenario of the SLR projections rather identifying the root causes of SLR as well as global warming. As an integral part of SLR, the impacts of sea surface water temperature rise and the possible threat on the coastal ecology of the country have been pointed out using secondary data sources. Intensive information on salinity-intrusion and possible coastal storms are also studied. Furthermore, how the poverty and the dependency on the foreign aid are linked with the climate change and SLR are demonstrated in the study from the socio engineering aspect. The critical combinations of location, geography and climate of Bangladesh coupled with its high population density are not only influencing the country to be a disaster prone one, but also placing difficulty in adopting protection measures. The proposed study is expected to provide useful information regarding socio-economic impacts of SLR, so that sustainable adaptation measures can be identified to cope with SLR in the coastal region of Bangladesh as well as other parts of the world.

Keywords: Coastal region, climate change, SLR, salinity intrusion, socio engineering, Bangladesh.

1. Introduction

Sea Level Rise (SLR) is one of the climate change issues that first drew global attention of climate scientists. The greenhouse gas emissions are resulting global warming which in turn causing glacier melting and sea-level rise (SLR). According to the Intergovernmental Panel on Climate Change (IPCC, 2007a), Bangladesh has been described as one of the most vulnerable countries in the world facing SLR and storm surges from the Bay of Bengal (IPCC, 2007b). The geographical location, geomorphology and local factors like subsidence and sediments play a catalytic role for turning it to be a disaster prone country (Khan et al., 2011; Bhuian and Dutta, 2012; Bose, 2013; Saroar and Routray, 2010). In recent time, the coastal zone of Bangladesh was hit by a series of tropical cyclone, SIDR, AILA and NARGIS, which have exposed the vulnerability of the people living in the zone (Khan et al., 2011). About 28% of the population lives in the coastal zone (Karim and Mimura, 2008), so a minimum increase in SLR could spell disaster for the whole country (Nicholls, 2002). It is estimated that the projected rise in the sea level would inundate 16% of the populated land in the 21st century which would results a displacement of around 10-30 million people and may reduce the GDP of the country by 10% (Houghton et al., 2009). However, the projected figures of the impacts on the land and GDP due to the SLR were termed not realistic enough in a recent study by Brammer (2014). Coastal zone does not only have an enormous conservation value but is also one of the most productive and diverse ecosystems in the world (Loucks et al., 2009). Thus, the predicted sea surface temperature rise combined with the sea level rise in the coastal zone would be a potential threat to the overall socio-economic development of the country. This study revealed a pragmatic and comprehensive understanding about the socio-economic impacts of SLR and will add value to the existing body of knowledge related to flood plain management authorities and SLR refuges.

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2. Overview of the coastal zone of Bangladesh

Bangladesh, stretching from the Himalayan at north to the Bay of Bengal at south, is one of the largest geosynclinals in the world. With an area of 147,570 km², the country is bounded between 20°34’–26°38’N and 88°01’–92°41’E (Figure 1). Most of the land, except the north-eastern and south-eastern regions, lies in the flat and low-lying floodplains of three major rivers - the Ganges, the Brahmaputra and the Meghna (Brammer, 2014). This tropical country is mainly undergoing sea level rise along its 710 km long coastline as a result of the global warming and the subsidence of the Bengal Delta over the last 11,000 years (Alam, 1996; Haque, 1997). All these scenarios make the country susceptible to the natural catastrophic disasters (Ali and Chowdhury, 1997).

![Figure 1 Coastal region of Bangladesh (Karim and Mimura, 2008)](image)

About 28% of the country’s population occupying with the 32% of the land area live in the coastal zone under tidal influence (Karim and Mimura, 2008). Moreover, during the fishing season a significant number of migratory fishermen come to the coastal zone from all over the country. The south-western zone, which is the home to around 14 million people, is considered as the exposed coast for its extreme vulnerability from water logging, salinity intrusion and storm surges (BBS, 2005).

3. Future climate change scenarios and the SLR projections

The global temperature has increased by about 0.6 °C and global sea levels have risen by around 10–20 cm during the last 20th century (Houghton et al., 2001). The apparent rise of the sea level of the twentieth century is predicted to continue or get worsen in the twenty first century (Nicholls et al., 2011, Ahmed and Alam, 1999). Bangladesh has already become warmer by 0.5° C over the past 100 years (Karim and Mimura, 2008). The monsoon rainfall (long term mean 166 ± 12 % cm) has been observed to increase abruptly since 1950 in Bangladesh (Bhuiyan, 2005). The Organization of Economic Co-operation and Development (OECD) conducted a study which found a steady increase in temperature in future in comparison to the current climate pattern of Bangladesh (Agarwala et al., 2003). The meteorological research council of the South Asian Association for Regional Cooperation in a research found relative sea level rise at three significant parts of coastal zone. They found that the SLR has risen by 4.0, 6.0 and 7.8 mm/year based on the sea level data of 22 years during the period of 1980-2002 (SMRC, 2003). This rise in sea level in Bangladesh was found to be much higher than that of global average (1-2 mm/year in the last century) (Karim and Mimura, 2008).
According to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC), the global mean sea level is estimated to rise in the range of 18–59 cm from 1990 to 2100 (IPCC, 2007a). The predictions were found to be higher in the Second and Third Assessment Report, which were in between 23 and 96 cm, and 9 and 88 cm, respectively (Church et al., 2001; Warrick et al., 1996). In 2005, the National Adaptation Program of Action (NAPA), based on the result of the available existing studies and reports, came up with the following result (Table 1) for future Bangladesh where it was projected that Bangladesh may experience an SLR of 14, 32 and 88 cm by the year 2030, 2050 and 2100, respectively.

### Table 1: Future climate change scenarios for Bangladesh, revealed by NAPA, 2005 (IWM, 2005)

<table>
<thead>
<tr>
<th>Year</th>
<th>Temperature change (°C) mean</th>
<th>Precipitation change (%) mean</th>
<th>Sea level rise(cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Monsoon Season</td>
<td>Dry Season</td>
<td>Monsoon Season</td>
</tr>
<tr>
<td>2030</td>
<td>0.8</td>
<td>1.1</td>
<td>+6.0</td>
</tr>
<tr>
<td>2050</td>
<td>1.1</td>
<td>1.6</td>
<td>+8.0</td>
</tr>
<tr>
<td>2100</td>
<td>1.9</td>
<td>2.7</td>
<td>+12.0</td>
</tr>
</tbody>
</table>

### 4. Impacts of the SLR

Among the adverse impacts of climate change conditions, sea level rise is one of the major issues that first triggered the concern globally. Several assessment reports of the Intergovernmental Panel on Climate Change (IPCC) have revealed that the SLR may have unprecedented impacts on low-lying deltaic coasts, where one-fifth of the global population live (IPCC 1996, 2001, 2007). Hence, it is one of the major challenges for the coastal habitants as well as the whole nation of Bangladesh to reduce and manage the impact of SLR. Bangladesh is prone to various detrimental hydro-meteorological calamities, such as, flood, cyclone, tidal surge and river bank erosion; the estimated SLR may increase the intensity and magnitude of the disaster. The overview of the possible impacts of SLR of 0.10 m, 0.25 m and 1.00 m for Bangladesh had been assessed by the World Bank in its report, titled as “Bangladesh: Climate change and Sustainable Development report in 2000” (World Bank, 2000). The report unveils that a 1.0 m rise in sea level would affect 17.5% land area of Bangladesh. It would result in devastating storm surge along with significant inundation and salinity intrusion in surface and groundwater which would eventually affect the agriculture, livelihood and ecosystem of the country (Table 2). Based on the study of (Bijlsma, 1996; Singh et al., 2001; Nicholls, 2002; Cannon, 2002; Saroar and Routray, 2010) the four major impacts of SLR in the coastal zones can be identified as shown in Figure 2.

**Figure 2** Major impacts of the Sea Level Rise in the coastal zone
### Table 2 Possible impact of SLR in Bangladesh (World Bank, 2000; Sarwar and Khan, 2007)

<table>
<thead>
<tr>
<th>SLR (m)</th>
<th>Affected Land Area (m²)</th>
<th>Storm surge</th>
<th>Flooding</th>
<th>Agriculture</th>
<th>Ecosystem</th>
<th>Salinity</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.1</td>
<td>2,500 (2%)</td>
<td>Storm surges with a 10% increase in intensity from the cyclone of 1991, wind speed increases from 225 to 248 km/h; Storm surges goes from 7.1 to 8.6 m with 0.3 m SLR.</td>
<td>Inundate 0.2 Mmt.of production &lt;1%of current total.</td>
<td>Inundates 15% area of the Sundarbans</td>
<td>Increase</td>
</tr>
<tr>
<td></td>
<td>0.25</td>
<td>6,300 (4%)</td>
<td>Storm surge goes from 7.4 to 9.1 m with 1 m SLR.</td>
<td>0.3m SLR inundate 0.5 Mmt.of production; 2% Of current total.</td>
<td>Inundates 40% area of the Sundarbans.</td>
<td>Increase</td>
</tr>
<tr>
<td></td>
<td>1.0 (high end estimate)</td>
<td>25,000 (17.5%)</td>
<td>Both inundation area and flood intensity will increase significantly.</td>
<td>Devastating flood may cause crop failure for a year.</td>
<td>The Sundarbans would be lost. Loss of the Sundarbans and other coastal wetlands would destroy the ecosystems.</td>
<td>Increase</td>
</tr>
</tbody>
</table>

### 4.1 Coastal Inundation

Permanent inundation is the most apparent consequence of the SLR in certain parts of the coastal region of Bangladesh. The country is situated at the lowest riparian area of the world’s second largest catchment Ganges-Brahmaputra-Meghna (GBM). The combined flow and the sediment yield of the total catchment from the melting of Himalayan glaciers and ice sheets with an annual runoff of 1200 km³ will raise the sea level; inundate deltas, coastal lowlands and coral islands; erode lands and worsen coastal flooding (Mirza, 2002). The monsoon rainfall along with air temperature and extreme climatic events are expected to increase more in these geographical regions in future.

A mathematical model was developed by the Institute of Water Modelling (IWM), Bangladesh in 2005 to estimate the impacts of SLR. Through analysing the existing and estimated SLR scenarios, the model predicted that 11% of the total land area would be permanently inundated within the century. The Sundarbans, the world’s largest mangrove ecosystem, declared as the Ramsar Heritage Site in 1997, will be destroyed by the year 2100 as a result of the projected SLR by IWM (2005). As in other Asian countries, the numbers of coastal flooding events in Bangladesh have tripled with financial losses and human casualties rising by more than five folds in the last 30 years (Bhuiyan, 2005).

### 4.2 Salinity Intrusion

The effect of salinization and its consequences to the surface water and ground water resource is one of the pressing issues of SLR. The salinity intrusion will ultimately affect the fresh water quality (Bashar and Hossain, 2006) and in turn will affect the coastal habitats, agricultures, fisheries, industries, aquatic plants and animals and the ecosystems adversely (Zhang et al., 2011). A recent study showed that the amount of salinity and the extent of salinity intrusion will increase due to sea level rise in Bangladesh (Bhuiyan and Dutta, 2012). The study developed an
Impact of Sea Level Rise in Bangladesh: A Socio Engineering Overview

Ali and Kader

Journal of Hydrology and Environment Research

63

integrated model to simulate salinity intrusion in the exposed coast (south-western part shown in Fig. 1) of Bangladesh. The upper range of the SLR value (59 cm), predicted by the fourth assessment report of IPCC, was applied in the model for simulating the worst possible impact (Bhuiyan and Dutta, 2012).

Table 3 Results of increase in salinity showed by the model from the two different stations in the coastal zone of Bangladesh (Bhuiyan and Dutta, 2012)

<table>
<thead>
<tr>
<th>Station</th>
<th>Without SLR(ppt)</th>
<th>With 59 cm SLR(ppt)</th>
<th>Increase of Salinity(ppt)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1st station</td>
<td>14.8</td>
<td>15.7</td>
<td>0.9</td>
</tr>
<tr>
<td>2nd station</td>
<td>17.3</td>
<td>18.0</td>
<td>0.7</td>
</tr>
</tbody>
</table>

The results from the two different stations demonstrate the increase in salinity due to SLR at the range of 0.7-0.9 ppt. Furthermore, National Adaptation Programme of Action (NAPA), in 2005, represented an indication of increase of salt water intrusion through an example of a power station situated in the coastal zone of Bangladesh. The study found that, the fresh water (free from salt impurities) that the power station needs has been collected from noticeably further upstream since the last decade (NAPA, 2005). The largest continuous mangrove ecosystem, Sundarban, has been affected already by the salt water intrusion and as an impact, its historical natural heritage, Sundari Tree is ceasing to exist (Rahman et al., 2011).

4.3 Cyclone Storm Surge

IPCC (2007) has predicted that Bangladesh is likely to be affected by more extreme catastrophic weather events like cyclone and storms in the upcoming days due to climate change and SLR (IPCC, 2007a). However, the country’s overall capacity to face the natural disasters has improved over the last decade as a result of better flood forecasting, warning and evacuation systems. For example, the cyclones of 1970 and 1991 washed away around 500,000 and 138,000 lives respectively in Bangladesh, whereas Sidr a cyclone of similar destructive magnitude caused 500 people in 2007 (Karim and Mimura, 2008).

An analysis was conducted in 2008 using a model calibrated with present climate condition to assess the impact of SLR due to cyclonic storm surge floods in Bangladesh by Karim and Mimura (2008). It was observed that the 2º C rise of sea surface temperature and 0.3m rise of sea level could lead to increase in flood risk area by 15.3% from the present conditions. This will worsen the depth of flooding by 22.7%, up to 20km from the borderline of the coastal zone. Consequently, the high risk zone areas are predicted to be 1.26 times larger than the existing one.

5. Socio- engineering aspects of SLR

The impacts of SLR (inundation, salinity intrusion and cyclone or storm) will ultimately affect the basic needs of the millions of people and will raise the issue of country’s water and food security problem. Particularly the agricultural production like rice production, and fisheries sector like shrimp farming in the coastal zone will be impacted by SLR in Bangladesh. For example, a flood in the south-western part of Bangladesh caused damages to crops, fish farms (especially shrimp farms), property and infrastructures worth about at least $500 million in the year of 2000 (Basher, 2000). The physical geography of the Bangladesh’s coastal area was stated as neither uniform nor static; it was rather identified as diverse and dynamic (Brammer, 2010), so are the people of Bangladesh. The casual loop diagram in Figure 4 (Haraldson, 2004) was developed to show the impact of SLR on the agriculture and fisheries sector of the coastal zone of Bangladesh.

From the above loop diagram, how the poverty and the dependency on the foreign aid are boosting up from the climate change and sea level rise are depicted. Here, the diagram is centralized with respect to the agricultural and fisheries sector, as because these two are the main driving factors of the coastal economy as well as national economy. Coastal inundation or flood, salinity intrusion, cyclone frequency, and damage and cyclone storm surge increases are shown in the figure as the three major impacts of SLR. All these three factors affect the coastal agriculture and fisheries adversely, so represented as inversely proportional. However, about 60-80% of animal protein intake by the people of Bangladesh comes from fish consumption (Alam and Thomson, 2001). So, diminishing coastal fisheries would cause protein deficiency and eventually cause...
health hazards. Moreover, agricultural and fisheries industries (both frozen and fresh) are one of the largest foreign exchange earning sectors of Bangladesh. Hence, reduction of foreign exchange earnings would raise the poverty as well as dependency on foreign aid due to climate change conditions and SLR.

**Figure 3** Map representing risk prone areas in the coastal zone of Bangladesh resulting from 2°C temperature rise and 0.3m sea level rise (Karim and Mimura, 2008)

**Figure 4** Casual Loop Diagram of Sea Level Rise Impacts on Coastal Agriculture and Fisheries Sector of Bangladesh (Sarwar and Khan, 2007)
6. Conclusion

The detrimental impacts of SLR are emerging even more pronouncedly in the future and this issue alone can let down the Millennium Developments Goal and can cause significant amount of ecological refugees. This paper identifies the key impacts of SLR in coastal Bangladesh that fall under the socio engineering overview. Using secondary data sources, intensive information on coastal inundation, salinity intrusion and possible cyclone storm surges are studied and showed how the poverty and the dependency on the foreign aid are boosting up with SLR. The investigation will provide opportunities to the responsible authorities and policy makers to better comprehend the coastal and flood management plan. The Government of People’s Republic of Bangladesh has already taken some steps and articulated Coastal Zone Policy in 2005 (CZPo, 2005), still there are additional opportunities emphasising holistic, integrated and sustainable measures to combat SLR impacts. In addition, the study will facilitate the environmental inventors and researchers to develop methodologies for assessing the impacts of climate changes, so that they can establish and identify pragmatic, site specific mitigation measures based on the socio economic conditions. Nevertheless, the ability to develop, adopt and cope with climate change impact is a function of capital, scientific and technical understanding, skills, information, strategy and managing bodies and equity. This type of study is essential for the global communities as well as it is a global issue and their collective efforts are indispensable to apprehend the extent at which these climate elements will change and to realize the consequent impacts at the coast.

References


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Abstract: Correlation of the Priestley-Taylor coefficient value, $\alpha$, with air humidity and wind speed for obtaining evapotranspiration (ET) values similar to those by the Penman-Monteith equation maybe justified from a strictly utilitarian viewpoint in cases when wind measurements are lacking. However, by doing so, a conflict of scale arises, which leaves ample room for incorrect generalization about the behavior of the Priestley-Taylor $\alpha$ value at the original, regional scale, it was designed for. Such improper generalizations are misleading and obstructive to further ET research, as this author experienced more than once (and thus prompting the recent comment) when he was pointed to the work of Cristea et al (2013) on how the PTE $\alpha$ is expected to behave.

Keywords: Priestley-Taylor equation, Priestley-Taylor coefficient, Penman-Monteith equation, wet-environment evaporation, spatial scale, oasis effect.

Technical Note

The authors calibrate the single parameter, $\alpha$, of the Priestley-Taylor (1972) equation (PTE)

$$ET = \alpha \frac{\Delta}{\Delta + \gamma} R_n$$

(1)

with the help of the FAO-56 version of the Penman-Monteith (PM) equation (Allen et al., 1998) in order to estimate daily reference evapotranspiration. Here ET (mm day$^{-1}$) is the evapotranspiration rate of an extensive wet surface, $R_n$ is the net radiation expressed in water depth equivalent of mm day$^{-1}$, $\gamma$ is the psychrometric constant (hPa K$^{-1}$), and $\Delta$ (hPa K$^{-1}$) the slope of the saturation vapor pressure curve to be evaluated at the air temperature ($T_w$) above the wet surface (Priestley and Taylor, 1972).

According to the authors, the rationale of calibrating $\alpha$ is that the PTE does not require wind velocity ($u$) measurements unlike the PM equation, therefore, after calibration, it may become a more flexible tool than the latter since wind velocities are not measured as widely as more basic meteorological variables, such as air temperature ($T_a$), humidity, and perhaps solar radiation ($R_s$). Then the authors regress the obtained $\alpha$ values against air humidity [i.e., relative humidity (RH) and vapor pressure deficit (VPD)] and wind speed. Finally, they create a map of $\alpha$ (as well as RH and $u$) values across the conterminous U.S. by spatial interpolation.

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The core idea of the paper is a practical one: the application of a simpler equation for performing the job of a more complex and data intensive one. However, two problems with this approach exist. First, the complete overlook of scale, the second which derives from the first, is the room left for incorrect generalizations.

The FAO-56 version of the PM equation was meant to be used in agriculture (hence the Food and Agricultural Organization acronym, FAO), therefore the scale the equation is valid for is the size of the typical agricultural plot, i.e., in the order of a couple of hundred meters. In contrast, the scale the PTE is valid for is at least a magnitude larger, a couple of kilometers to tens or even hundreds of kilometers, depending on how fast the local advection term (i.e., the second term that contains the vapor pressure deficit) of the PM equation approaches zero as one moves along the wet and vegetated land surface and the air becomes closer and closer to saturation due to evaporation from the wet surface. A quote from Priestley and Taylor’s original (1972) article: “We shall consider the minimum requirements to be the daily rates … with a spatial resolution... on the order of several hundreds of kilometers.”

Typically, the more humid the climate, the less is the difference in air temperature and humidity between the land and a permanently wet vegetated surface, therefore, the less significant this local advection of hotter and drier air becomes. Then the classical PTE (with its typically employed constant $\alpha$ value of 1.26) yields ET rates close to the PM-calibrated PTE (i.e., $\alpha$ varies by location), as illustrated in Figure 1. Note that with only one single exception, the PM-calibrated PTE ET rates are larger than the corresponding classical PTE ET values.

**Figure 1** ET rates (mm d$^{-1}$) of the PTE when a) $\alpha$ was calibrated via RH and $u$ in the FAO-56 PM equation; b) $\alpha$ = 1.26 and $A$ is evaluated at $T_w$; c) $\alpha$ = 1.26 and $A$ is evaluated at $T_{aw}$ provided $T_{aw} < T_r$. $T_{aw}$ is the Szilagyi and Jozsa (2008) estimated wet surface temperature. $R_n$ is in mm d$^{-1}$, $u$ in m s$^{-1}$, and VPD in kPa. The product of $u$ and VPD is proportional to the advection term of both the FAO-56 PM and Penman (1948) equations.

Figure 1 employs the annual mean values of the Appendix Table of Cristea et al. (2013) in the same station order. $R_n$ was obtained by using the regression equation they provide for Makkink-Hansen $C$ and $\alpha$, via the solar radiation values of the Table. As the PTE was parameterized under actual wet environmental conditions (Priestley and Taylor, 1972), it expects the wet environment air temperature, $T_{aw}$ for $A$. However, the PTE is typically employed under drying conditions (as in Figure 1), resulting in that $A$ is evaluated at $T_a$ and not at the required $T_{aw}$. Since the latter is unknown under drying conditions, it can be approximated by the wet surface temperature, $T_{aw}$ which is easier to estimate than $T_{aw}$ (Szilagyi and Jozsa, 2008; Szilagyi et al., 2009). In more humid climates where the difference in $T_r$ and $T_{aw}$ is typically small, $T_{aw}$ can be larger than $T_r$. Therefore, in such cases it is better to use the cooler actual air temperature as an estimate of $T_{aw}$ (which is always smaller than $T_{aw}$ under equilibrium-flux conditions). This is the case at 21 out of the 22 stations of Figure 1 (and so the $T_r$ and $T_{aw}$ ET values overlap), except at station #9, in Oasis, California, where $T_{aw} < T_r$. Note that on a daily basis (rather than on an annual mean basis of Figure 1) and especially in the summer when RH is typically smaller, $T_{aw}$ would be more frequently below $T_r$ and so the two ET rates would differ more.
What is important, however, at the Oasis station is that the PM-calibrated $\alpha$ value yields an ET rate significantly larger than the energy ($R_a$) available for sensible and latent heat fluxes at the surface. Note that over an extended wet surface the PTE is valid for, equilibrium fluxes develop (Priestley and Taylor, 1972; Brutsaert, 1982), meaning that both latent and sensible heat fluxes are constant with elevation from the ground in the lowest couple of tens of meters at least. This is only possible if both specific humidity and the potential temperature profiles are monotonically decreasing with height above the ground, and both of the resulting fluxes thus are directed from the surface toward the air. From this it follows that the equilibrium latent heat flux must be smaller than the available energy at the surface. What we see at Oasis is that the PM-calibrated $\alpha$ value yields a latent heat flux much larger than the available energy. How is it possible?

It is possible exactly because of the scale the FAO-56 PM equation is valid at and was parameterized for. Namely, in strong local advective conditions, such as those that no doubt exist at Oasis, CA (the name is telling), the ET rate of a wet plot (may the plot be ‘extensive’ in agricultural term, but still not reaching the scale required by the PTE) is boosted significantly by the advected energy of the hot and drier air of the region. So at Oasis, the PM-calibrated PTE gives a plot-sized ET rate (via an artificially boosted $\alpha$ value) larger than the available energy and thus much larger than the ET rate of a truly expansive wet surface (with the standard $\alpha$ value of 1.26) over which these local advection effects are negligible. Therefore, by tweaking the $\alpha$ value of the PTE, it is possible to estimate plot-sized wet surface ET, but then it must be acknowledged that the PTE is not used as it was intended to be used, and most importantly, one must not generalize the results as if the inflated $\alpha$ values were the norm and that is how the PTE $\alpha$ behaves. What Cristea et al. (2013) do is simply this tweaking of the $\alpha$ value to artificially account for advection effects, the PTE was never meant to be applied for, in fact, it was deliberately derived to exclude such effects. From a strictly utilitarian and practical point of view what they do may be justifiable – since this way one does not need wind velocity values (once $\alpha$ is calibrated for the study region), as one does in the PM equation. But then one should fully recognize and acknowledge what is done in the paper.

From Figure 1 it is evident that whenever energy advection (proportional to $u$ times VPD, since the larger VPD the larger the $T_a - T_w$ difference) peaks, so does the PM-calibrated $\alpha$ value, and also the larger the resulting ET rate becomes in comparison to the constant $\alpha$ case. It is also interesting to note why it is that at Puyallup, WA, close to the cold water of Puget Sound, the opposite is taking place, i.e., the PM-calibrated $\alpha$ value is much smaller than 1.26 (close to unity). It is because here relative humidity is the largest of the stations, and the humid and cool air coming from the sound by the prevalent wind exerts a) a possible negative advection effect by cooling the warmer wet surface and thus suppressing ET, or b) advection effect is indeed negligible but the FAO-56 PM equation may have been calibrated in predominantly drier and warmer climates (for irrigation purposes) where local energy advection at the plot scale it was designed for is always significant, thus when it is supplied by a significantly depressed VPD value, it undershoots.

The mean ET rate of Figure 1 with the PM-calibrated values is 2.89 mm day$^{-1}$; by a replacement of the Oasis value with $R_a$ it becomes 2.85 mm day$^{-1}$, which is still 10% larger than the constant $\alpha = 1.26$ case of 2.57 mm day$^{-1}$. So on average, local energy advection boosted ET rates by about 10% at the stations of Figure 1. But this boost is not even – at dry and hot stations it becomes much larger.

As a consequence, the PM-calibrated $\alpha$ value becomes correlated with air humidity and wind speed [as Cristea et al. (2013) conclude], as it indeed should at the plot scale, for the above reasons. Unfortunately, Cristea et al. (2013) never mention this scale conflict in their work, which leaves ample room for incorrect generalization by other workers about the behavior of the PTE $\alpha$ value at the proper, more regional scale, it was designed for. Such generalizations are misleading and obstructive to further ET research, as this author experienced more than once (and thus prompting the recent comment) when he was pointed to the work of Cristea et al (2013) on how the PTE $\alpha$ is expected to behave.

It is true, however, that the larger than unity $\alpha$ value of the PTE involves certain advection effects, but it is regarded by most workers as largely the result of the entrainment of drier air from the free troposphere into the daytime developing convective boundary layer (Brutsaert, 1982; de Bruin, 1983; Culf, 1994; Lhomme, 1997; Heerwaarden et al., 2009). Local energy advection at the plot scale however should not be washed together with the latter. The former takes place even in the middle of the oceans, while the latter, local advection, mainly over (or near the shore, see sea-land breeze systems) the land along soil moisture gradients.

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