CHAPTER 2

LITERATURE REVIEW

2.1 INTRODUCTION

This chapter reviews the literature related to the river water pollution and ground water pollution. Also, literature related to nitrate and phosphate pollution is also discussed. Adsorption and related works that already have been carried out are mentioned elaborately in this study.

2.2 SURFACE WATER POLLUTION IN INDIA

2.2.1 River Water Pollution

There is no systematic study available to provide a comprehensive picture of the magnitude and spatial dimension of water pollution in India. However, studies carried on by India’s Central Public Health Engineering Institute at Nagpur and the Central Inland Fisheries Research Institute at Calcutta, Allahabad; highlight the seriousness of the water pollution problem. The average chemical composition of sewage in some major cities of India is given was studied and reported by Mishra (1994), Dakshni & Soni (1979), Bharati et al. (1979) and NEERI (1994). With the increase in human and livestock populations, the wastes produced in India have increased considerably. Since most of the cities situated along the banks of the streams do not have any sewage treatment facilities, the release of sewage into rivers
can pollute the water (Subbarao & Vachaspati 1978; Subbarao & Sathyanarayana 1988).

Even in Delhi, which has a sewage treatment plant, over 100 million gallons of untreated sewage is discharged into the Yamuna River (Dakshini & Soni 1979; Sarin and Krishnaswami, 1984). In fact, according to the latest health statistics (1982) around 11 million people in India suffered from various waterborne diseases and 11,234 died. The diseases include typhoid, infectious hepatitis, dysentery, and gastroenteritis. Apart from pollution from animal wastes, referred to above, industries also release waste effluents either directly into streams or into drains/channels that eventually join the streams, resulting in degradation of water quality. Only 42% of the major industries in India have effluent treatment plants. In addition to these, there are numerous small industries that discharge their untreated wastes into the streams and lakes.

2.2.2 Stream Pollution

It is estimated that 70% of India’s inland waters are of “doubtful quality.” Unsatisfactory methods for disposal and treatment of sewage and industrial wastes are among the major causes of water pollution. Waste materials are commonly discharged without treatment into inland or coastal waters. For example, raw sewage is dumped into the Ganga at Varnasi, where water is used for drinking purposes by thousands of pilgrims each day (Mathur 1979, Tripathi et al. 1991).

In Kolkata, Mumbai, and Chennai, part of the untreated sewage goes directly to the river or sea (Ghosh 1969, Somasundaran et al. 1993). Information collected from officials of 20 major Indian cities on status of sewage disposal revealed that nine cities (Kanpur, Bangalore, Poona, Nagpur, Lucknow, Agra, Varnasi, Madurai, and Allahabad) had no sewage treatment
plant; six cities (Amritsar, Hyderabad, Indore, Jaipur, Sholapur, and Patna) had secondary treatment plants; only two cities (Ahmadabad and Delhi) had modern sewage treatment plants.

The untreated sewage going into the river or sea creates serious odour problems in some sections of these cities. In addition, the overburdened sewers of these three metropolitan centres lead into the water mains laid under them, polluting the drinking water supply. In India there are no legal standards of maximum allowable pollution and no data exist on the capacities of receiving streams to absorb pollution loads without adverse environmental impact.

Water pollution resulting from industrial waste disposal into streams or other bodies of water is localized in industrial centres such as Bombay, Calcutta, Delhi (Agrawal et al. 1986), Kanpur (Garg et al. 1992) and the Damodar Valley. Destruction of fish from lack of dissolved oxygen or from the toxic effect of industrial wastes has been reported in rivers such as the Damodar (Gopalkrishnan et al. 1966, NEERI 1994), Son, Mahanadi, Jamuna, Ganges (Tripati et al. 1991), Cauvery, Krishna, Mahi near Baroda, and Kalu near Bombay. A large fish kill was reported in the Jamuna River near Delhi due to the untreated waste of an insecticide factory located at Najafgarh, north of the city. In March 1970, the plant that filters water from the Jamuna for over half a million residents of South Delhi was shut down for three days because it was clogged with dead fish resulting from industrial waste. During this water crisis, municipal tankers distributed water to residents of South Delhi. However, hepatitis due to use of “contaminated” water was revealed in a survey conducted by the All-India Institute of Medical Sciences in Kailash, which is located south of Delhi and received drinking water from the plant.
The concentrations of total DDT residues ranged from 0.04 μg/L to 1.42 μg/L in the water and, from 0.007 to 5.63 mg/kg in bottom sediments were detected at Delhi. The total DDT concentration was higher at the downstream Wazirabad site where there is a mixing of river water with the discharge from the Najafgarh drain, which also carries effluents of a DDT factory along with that of other industries (Agarwal et al. 1986).

The Damodar River is perhaps the most polluted river in India (Gopalkrishna et al. 1966, NEERI 1994). It receives wastes from various industries on its banks—steel mills, coke-oven plants, coal-based chemical industries, and fertilizer factory wastes such as alkalis, ammonia, cyanide, and phenols. About 160,000 m$^3$ of waste water having a BOD of over 43,000 kg from industries is discharged daily into the Damodar River. Untreated wastes from chemical and rayon industries between Bombay and Kalyan flow into the creeks, sewers, or streams which find their way to the Kalu River. As a result of the efforts by the State Water Pollution Investigation Centre in Bombay, the industries are being persuaded to limit their effluents to improve the water quality in the Kalu and Ulhas estuaries.

Industrial effluents flowing into the Cauvery River, the major source of drinking water in Tamil Nadu, contain tannery, distillery, and acidic sugar mill wastes with high alkalinity content and dissolved solids. The fish catch dwindled in the Bhadra River near Bhadravati, Mysore, because of wastes from paper and steel mills. The Godavari in Andhra Pradesh and the Chaliyar in Kerala, the Tapti, and the Chambal in Madhya Pradesh are among the other major Indian rivers contaminated by industrial wastes at various locaations. The Ganga River, with a length of 2500 km, is one of the largest rivers on the Indian sub-continent. There are 48 large cities and 66 small towns situated on its banks, and none of them has sewage treatment plant; the major part of the sewage is discharged into the river. Discharge of untreated
industrial waste effluents and animal and human excreta along its banks are the other major sources of pollution. There are several stretches along its course where the river is grossly polluted, viz., near Kannauj, Kanpur, Allahabad, Varanasi, Ballia, Patna, and Calcutta.

2.3 GROUND WATER POLLUTION IN INDIA

2.3.1 Iron

In northeast India, West Bengal (Handa 1975; CGWB 1976) and in some other parts of India, ground waters with relatively high iron concentrations have been observed. In northeast India, for example, ground waters having as much as 20 mg L⁻¹ Fe have been observed. High iron ground waters containing up to 2–3 mg L⁻¹ Fe are also found in some parts of UP, Pondicherry, etc. The occurrence of high iron ground waters is associated with a humid climate, occurrence of pisolitic nodules containing iron (e.g., in West Bengal), and lowering of redox potential due to anaerobic conditions.

2.3.2 Fluoride

In many parts of the semi-arid to arid parts of India (Handa 1984), e.g., in the western parts of Sirohi District, Eastern Nagaur, south western Ganga nagar, northwest and east of Jodhpur (all in Rajasthan), Prakasam (Raman & Rao 1983), Nalgonda, Anantapur, Chittor, Medak, Nandigama taluk in the Krishna District, Vishakhapatnam (Sarma & Swamy 1981, 1983, Prakash et al. 1989), Hazurabad taluk in Karimnagar (all in AP) and in some parts of Haryana, Tamil Nadu, Karnataka, and Punjab, ground water with high concentrations of fluoride are found (Raju & Goud 1990; Ramesh & Subramanian 1988). The highest values for fluoride were recorded in Bankapatti, where concentrations up to 17–20 mg L⁻¹ have been found (Deolankar 1990).
2.3.3 Manganese

In Unnao District (UP) and in some other parts of India (Handa 1984), ground water with relatively high concentrations of manganese is encountered. Apparently this water owes their origin to the occurrence of anaerobic conditions that resulted in the mobilization of manganese present in the strata through which the ground water was circulating.

2.3.4 Nitrate and Potassium

In many parts of India, waters with high K and/or high NO\textsubscript{3} content occur, with K and/or nitrate ions exceeding 1000 mg L\textsuperscript{-1}. It has been suggested that this phenomenon is due to contamination from nutrient-enriched return irrigation flows (Jacks & Sharma 1983, Pawar & Shaikh 1995). In some cases anomalous values of phosphates have also been found.

2.3.5 Landfill Leachate Pollution

Landfills have been identified as one of the major threats to groundwater resources (Fatta et al. 1999) not only in India but throughout the world (US EPA 1984). More than 90% of the Municipal Solid Waste (MSW) generated in India is directly dumped on land in an unsatisfactory manner (Chatterjee 1967). The solid waste placed in landfills or open dumps are subjected to either groundwater underflow or infiltration from precipitation or any other possibility of infiltration of water. During rainfall, the dumped solid wastes receivers’ water and the by-products of its decomposition move into the water through the waste deposition. The liquid containing innumerable organic and inorganic compounds is called 'leach ate'.

This leach ate accumulates at the bottom of the landfill and percolates through the soil and reaches the groundwater (Mor et al. 2006).
Areas near landfills have a greater possibility of groundwater contamination because of the potential pollution source of leachate originating from the nearby dumping site. Such contamination of groundwater results in a substantial risk to local groundwater resource user and to the natural environment. The impact of landfill leachate on the surface and groundwater has given rise to a number of studies in recent years and gained major importance due to drastic increase in population (Saarela 2003). There are many approaches that can be used to assess the groundwater and surface water contamination. It can be assessed either by the experimental determination of the impurities or their estimation through mathematical modelling (Moo-Young et al. 2004).

2.3.6 Sanitary Quality Assessment of the Surface Waters

The microbiological quality of water is measured by the analysis and enumeration of indicator coli forms (Briancesco 2005). Coli forms or indicator microorganisms are present in the intestinal tracts of warm blooded animals, including humans (Leclerc 2001) and therefore can be excreted in the feces of these animals, although there have been some associations between high levels of indicator bacteria and disease outbreaks (Chou 2004). Indicator organisms are commonly used to assess the microbiological quality of surface waters and fecal coli forms (FC) are the most commonly used bacterial indicator of fecal pollution in India. They are found in water that is contaminated with fecal wastes of human and animal origin. Total coli forms (TC) comprise bacterial species of fecal origin as well as other bacterial groups. The coli forms are indicative of the general hygienic quality of the water and potential risk of infectious diseases from water. High FC and TC counts in water are usually manifested in the form of diarrheal and sometimes by fever and other secondary complications.
2.4 NITRATE POLLUTION

In the past century, the intensification of agricultural activities and the application of artificial fertilizers have caused a severely disturbed nitrogen balance in agricultural areas as the input of nitrogen by far exceeds the yield in these systems (e.g. Goodchild 1998; Iversen et al. 1998; Kirchmann et al. 2002). Under aerobic soil conditions ammonia and ammonium ions are microbially transformed (nitrified) to nitrate. The adverse effects of high atmospheric nitrogen deposition in Western Europe (e.g. Bobbink et al. 1998) have resulted in governmental measures to decrease the loss of volatile ammonia from agricultural soils. Low emission application techniques (including manure injection) substantially reduced the ammonia emission but as a result, the leaching of nitrate to the groundwater further increased (Iversen et al. 1998). Next, non-agricultural sources, such as leaking sewage systems and home gardens (Wakida & Lerner 2005) and forests (e.g. Rothe & Mellert 2004; Gundersen et al. 2006), which catch atmospheric nitrogen deposition, may also significantly contribute to increased groundwater nitrate concentrations. In most parts of Europe, anthropogenic nitrogen deposition has led to increased nitrate leaching from forests. Pine forests have a high leaf area which is also maintained throughout the year, which leads to higher rates of N deposition from the atmosphere to the forest floor and to higher concentrations of nitrate in drainage water compared with broadleaved forests (Rothe & Mellert 2004).

World-wide, nitrate has become one of the common groundwater contaminants (e.g. Kool 2002; Postma et al. 1991; Nolan et al. 1997; Goodchild 1998; Iversen et al. 1998; Kirchmann et al. 2002; Senn & Hemond 2002; Thorburn et al. 2003; Broers et al. 2004; Surridge et al. 2007). Obviously nitrate pollution plays an important role in surface water eutrophication (Iversen et al. 1998; Kirchmann et al. 2002). However, it also
poises a serious risk for drinking water quality as infants under six months of age are susceptible to nitrate poisoning (Comly 1945; Fan & Steinberg 1996). The European standard for the maximum concentrations of nitrate in potable water is 50 mg L$^{-1}$ (806 mol/L). At present, this concentration is often exceeded in potable waters extracted from surface water or aquifers recharged by water from areas of agricultural activity. In the Netherlands, for instance, the phreatic and shallow groundwater below infiltrating agricultural lands nowadays typically contains median nitrate concentrations as high as 2000 mol/L, with minimum and maximum values of 900 and 3300 mol/L (Broers et al. 2004). The increase in deeper aquifers, however, is generally much less pronounced than would be expected from model studies. This phenomenon has been attributed to the consumption of nitrate due to microbial mediated nitrate reduction processes in the sub-soil.

2.5 NITRATE REDUCTION

The reduction capacity of the aquifer soils will determine the extent to which reduction of nitrate may take place. Sedimentary organic matter (SOM) is a relevant electron donor in aquifers (e.g. Hill et al. 2005; Hartog et al. 2002). The remaining reactivity of SOM in ground water systems may differ greatly, but the residual SOM tends to become more stable (less degradable) during oxidation (Cowie & Hedges 1994; Hartog et al. 2002). Sediment oxygen exposure time is probably a key factor determining the remaining reactivity of the SOM in aquifer sediments. Generally, in deeper aquifers SOM is present in low concentrations (0.01–0.2 wt. %).

Next, it is also characterized by the absence of more labile compounds such as cellulose, indicating that SOM has degraded to a considerable extent from its biomass precursors and has become relatively inert (Hartog et al. 2004). Being a powerful electron acceptor, nitrate can also be reduced by ferrous iron and may therefore importantly influence iron
cycling in natural systems. Ferrous iron (Straub et al. 1996) and ferrous iron bearing carbonates (such as siderite) and sulphides (such as pyrite) (Postma et al. 1991; Aravena & Robertson 1998; Pauwels et al. 1998; Moncaster et al. 2000; Lucassen et al. 2004; Haaijer et al. 2006, 2007; Burgin & Hamilton 2008) turn out to be important electron donors for denitrification in aquifer soils. These reactions are all catalyzed by microorganisms which, withstanding a wide variety of environmental conditions, thrive in shallow and deep aquifers (e.g. Pauwels et al. 1998; Haaijer et al. 2006, 2007; Burgin & Hamilton 2008).

As long as groundwater nitrate concentrations are high, ferrous iron levels tend to remain low because nitrate acts as a redox buffer, preventing the reduction of iron(III)oxides (Lucassen et al. 2004; Smolders et al. 2006a). Additionally, ferrous iron may be microbially oxidized by nitrate (Straub et al. 1996; Senn & Hemond 2002). High nitrate loads therefore strongly decrease groundwater iron concentrations. An advantage of this process might be that phosphorous entering aquifers along with nitrate, can be bound to oxidized iron in aquifers, potentially reducing the supply of phosphorous to wetlands via groundwater.

The chemoautolithotrophic oxidation of iron sulphide deposits by nitrate, frequently leads to a concomitant increase of groundwater sulphate concentrations (Postma et al. 1991; Aravena & Robertson 1998; Moncaster et al. 2000; Broers et al. 2004; Vanbeek et al. 2006). Although the reduction of nitrate by organic matter is thermodynamically favourable, iron sulphide oxidation appears to be kinetically favoured in deeper aquifers, probably because of the different microbiological availability of iron sulphides (including amorphic pyrite) and organic matter (Postma et al. 1991; Wriedt & Rode 2006). As a result of nitrate leaching, groundwater chemistry has changed profoundly in many parts of the world and is nowadays characterized
by lower iron and much higher sulphate concentrations (Postma et al. 1991; Aravena & Robertson 1998; Hoffmann et al. 1998; Pauwels et al. 1998; Moncaster et al. 2000; Molenat et al. 2004). In the Pleistocene regions of the Netherlands, for instance, increased agricultural activities have led to a strongly increased leaching of nitrate and to a concomitant increase of the groundwater sulphate concentrations. Near the pumping station of Vierlingsbeek (in the province of Limburg, the Netherlands), sulphate concentrations have increased from 400 to 1450 mol/L between the 1960s and 2000, of which atmospheric sulphur deposition only contributes 200 mol/L. In the same time span groundwater nitrate concentrations have increased from hardly detectable to 500 mol/L (Vanbeek et al. 2006). Estimations reveal that at present up to 70% of the sulphate present in groundwater in the Netherlands is derived from nitrate induced pyrite oxidation resulting in mean groundwater sulphate concentrations of 833 mol/L (Vanbeek et al. 2006). Before the 1950’s, groundwater sulphate concentrations in these regions were typically lower than 200 mol/L (STOM 1983).

2.6 INTERNAL EUTROPHICATION

Under anaerobic conditions, the availability of alternative electron acceptors strongly affects the breakdown of organic matter. In addition to the availability of electron acceptors, (anaerobic) decomposition is obviously strongly regulated by the availability of degradable organic matter (Drever 1997). In wetlands the availability of reactive SOM will be much higher compared to (deeper) aquifers due to the constant input of plant litter. This means that nitrate and sulphate, supplied by groundwater enriched with these compounds, can strongly stimulate the decomposition of organic matter (e.g. Roden & Edmonds 1997; Holmer & Storkholm 2001; Smolders et al. 2006a).
Next, the increased input of sulphate and decreased input of iron in groundwater fed semi-aquatic ecosystems may also cause phosphate release due to the fact that sulphide (produced by sulphate reduction) interferes with the iron-phosphorus cycle. The main product of sulphate reduction is dissolved sulphide. Sulphide may react with dissolved reduced iron and particulate iron (hydr) oxides. The products of the reaction between sulphide and iron are highly insoluble iron sulphide minerals while phosphate adsorbed to the iron (hydr) oxides or present as iron phosphates becomes mobilized (Sperber 1958; Caraco et al. 1989; Sondergaard et al. 1993; Roden & Edmonds 1997; Holmer & Storkholm 2001; Smolders et al. 2001; Lamers et al. 2002; Smolders et al. 2006a).

Prolonged high sulphate loads to such sediments will ultimately result in sediments in which up to 80% of the iron can be bound to reduced sulphur (FeSx) (Smolders et al. 1995). As a consequence, the capacity of the sediment to retain P will greatly decrease (FeSx has far fewer sorption sites for P than have iron (hydr) oxides, which may result in a high mobility of P in the soils and thus in a strong eutrophication of such systems (Caraco et al. 1989; Sondergaard et al. 1993; Roden & Edmonds 1997; Lamers et al. 1998; Wetzel 2001; Lamers et al. 2002; Smolders et al. 2006a). This may be enhanced by the decreased input of ferrous iron via the groundwater due to nitrate induced immobilization of iron. If sulphate reduction continues, toxic concentrations of hydrogen sulphide may accumulate in the sediment pore water and cause serious problems for rooted aquatic macrophytes by inducing sulphide toxicity and iron deficiency (Smolders & Roelofs 1996; Lamers et al. 2002; Smolders et al. 2006a). In extremely dry years desiccation of such sediments can lead to a strong acidification and mobilization of toxic metals, due to the oxidation of accumulated reduced sulphur to sulphuric acid (Lucassen et al. 2002; Smolders et al. 2006b).
In general, the classic iron cycle can explain the actual release of P from the sediment. In the oxygenated boundary layer between sediment and water layer, dissolved iron becomes oxidized and phosphate is effectively bound by iron (III) (hydr)oxides (Wetzel 2001; Smolders et al. 2006a). This mechanism explains the frequently found positive relation between the phosphate release to the water layer and the dissolved-P: dissolved-Fe ratios in sediment pore water (Caraco et al. 1989; Wetzel 2001; Smolders et al. 2001). Under sulphur-rich reducing conditions, however, these mechanisms will no longer function as all dissolved ferrous iron is precipitated with sulphide as FeSx, (Caraco et al. 1989; Smolders & Roelofs 1996; Wetzel 2001; Smolders et al. 2001). Furthermore, dissolved sulphides (and other reduced compounds) consume oxygen in the top sediment layer, thus decreasing the thickness of the oxidized boundary layer. This may greatly boost the release of dissolved phosphate from the sediment. Increased methane production rates under highly reductive conditions may further stimulate P release by ebullition, causing mixing of anaerobic phosphate rich sediment pore water with surface water, especially in organic sediments (Sondergaard et al. 1993; Wetzel 2001).

Geurts et al. (2008) recorded the aquatic vegetation and collected surface water, sediment pore water and sediment samples in 145 fen waters in the Netherlands, Ireland and Poland. In halve of these waters they found a decreased biodiversity and increased phosphate concentrations in the water layer which appeared to be SO$_4$ induced and especially occurred below certain threshold values for total sediment Fe:P (10 mol mol$^{-1}$). This suggests that especially sediments with relatively low iron content and/or high phosphorus content are prone to sulphate induced eutrophication.
2.7 WATER TREATMENT METHODS

Although very little investment has been made in the past on water treatment facilities, typically water supply and treatment often received more priority than wastewater collection and treatment. However, due to the trends in urban development along with rapid population increase, wastewater treatment deserves greater emphasis. Several research studies showed that, treated wastewater, if appropriately managed, is viewed as a major component of the water resources supply to meet the needs of a growing economy. The greatest challenge in implementing this strategy is the adoption of low cost wastewater treatment technologies that will maximize the efficiency of utilizing limited water resources and ensuring compliance with all health and safety standards regarding reuse of treated wastewater effluents.

Treatment options which are typically considered for the removal of emerging contaminants from drinking water as well as wastewater include adsorption, Advanced Oxidation Processes (AOPs), Nanofiltration (NF), and Reverse Osmosis (RO) membranes. However, the shortcomings of most of these methods are high investment and maintenance costs, secondary pollution (generation of toxic sludge, etc.) and complicated procedure involved in the treatment. On the other hand physicochemical treatments such as coagulation/ flocculation processes were generally found to be unable to remove Endocrine Disrupting Compounds (EDCs) and Pharmaceuticals and Personal Care Products (PPCPs). Although AOPs can be effective for the removal of emerging compounds, these processes can lead to the formation of oxidation intermediates that are mostly unknown at this point.
Conversely adsorption processes do not add undesirable by-products and have been found to be superior to other techniques for wastewater treatment in terms of simplicity of design and operation, and insensitivity of toxic substances. Among several materials used as adsorbents, Activated Carbons (ACs) have been used for the removal of different types of emerging compounds in general but their use is sometimes restricted due to high cost. Furthermore when AC has been exhausted, it can be regenerated for further use but regeneration process results in a loss of carbon and the regenerated product may have a slightly lower adsorption capacity in comparison with the virgin-activated carbon. This has resulted in attempts by various workers to prepare low cost alternative adsorbents which may replace activated carbons in pollution control through adsorption process and to overcome their economic disadvantages (Mohan & Gandhimathi 2010).

Recently natural materials that are available in large quantities from agricultural operations have been evaluated as low cost adsorbents and environmental friendly. Moreover the utilization of these waste materials as such directly or after some minor treatment as adsorbents is becoming vital concern because they represent unused resources and cause serious disposal problems. A growing number of studies have been carried out in recent years to evaluate the behaviour of emerging adsorbents such as agricultural products and by-product for emerging contaminants removal. On the other hand industrial wastes, such as, fly ash, blast furnace slag and sludge, black liquor lignin, red mud, and waste slurry are currently being investigated as potential adsorbents for the removal of the emerging contaminants from wastewater.
2.8 WATER QUALITY INDEX (WQI)

The quality of water generally is defined in terms of its physical, chemical and biological parameters (Ketata Mouna et al. 2011) and measured as Water Quality Index (WQI) to assess whether water is potable or not. WQI provides a single number that expresses overall water quality at certain location, based on several water quality parameters (Yogendra & Puttaiah, 2008). Hence, for any city, a groundwater quality map becomes a valuable parameter for evaluating potability and also as a precautionary indication of potential environmental health problems (Challerjee et al. 2009). The water quality index is a very powerful and useful tool to summarize and to report on the monitoring data to the decision makers in order to be able to understand the status of the groundwater quality and to have opportunity for better use in future as well.

2.9 GEOLOGICAL INFORMATION SYSTEMS (GIS)

Groundwater resources are dynamic in nature and affected by such factors as the expansion of irrigation activities, industrialization and urbanization, hence monitoring and conserving this important resource is essential. GIS and remote sensing has been used extensively used to assess the water quality all over the world (Shomar et al. 2010; Asadi et al. 2007; Ketata Mouna et al. 2011; Mitra et al. 1998; Liu et al. 2007). GIS can be a powerful tool for developing solutions for water resources problems, assessing water quality, determining water availability, preventing flooding, understanding the natural environment and for managing water resources on a local or regional scale (Ferry et al. 2003). GIS shows the same approach can be used for determining, monitoring and managing ground water quality and its
pollution for wide areas. The database formed can be very useful for future research and reference (Tikle et al. 2012).

Thus, extensive literature review in this area gave an idea of approaching to the problem in Kelavarapalli reservoir.