Role of Geographic Information System for Water Quality Evaluation

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ABSTRACT

Water quality evaluation is an overall process of evaluating physical, chemical and biological nature of water in relation to natural quality, human effects and intended uses particularly uses which may affect human health and the health of the ecosystem itself. Interpretation of enormous water quality data in a convenient manner for visual inspection is an important but often underestimated or omitted step in a water quality evaluation program. Recently, need of modern approaches and tools for interpreting water quality is emphasized for efficient water quality management. Geographic Information System (GIS), with an ability of capturing, storing, analyzing, manipulating, retrieving and displaying spatial data, has emerged as a powerful tool for decision-making in several areas including environmental field. This chapter aims at highlighting the role of GIS in synthesising, compiling, presenting and interpreting chemical data of both surface and ground waters. Firstly, few relevant fundamental terms and process of water quality evaluation are defined and/or described. Thereafter, the chapter contains theoretical procedure for applying GIS to assess spatial change or variability in water quality by characterizing extent and patterns of contamination. In general, a water quality monitoring network consists of a group of point locations with known chemical attributes of water. GIS helps converting the point values into areal information through spatial interpolation. Hence, an overview of spatial interpolation techniques is provided, together with the methodologies for employing geostatistical modelling (kriging) and inverse distance weighting techniques and for computing spatial statistics (mean, median, standard deviation and coefficient of variation). The major application of GIS in past groundwater studies has been for assessing groundwater vulnerability. Therefore, the concept of groundwater vulnerability along with its historical perspective is described and different GIS-based overlay and index methods used for groundwater vulnerability assessment are summarized. Methodologies for applying different GIS methods in evaluating the groundwater vulnerability are illustrated through flowcharts. The major tools for describing groundwater vulnerability in GIS framework include DRASTIC, modified DRASTIC, DRAMIC, GOD, AVI, SINTACS, EPIK, GLA, PI and COP. Furthermore, the development of GIS-based water quality index for evaluating water quality is discussed. Finally, combined use of GIS and multivariate statistical analysis techniques in delineating water quality zones is discussed. It is concluded that GIS is a promising geospatial tool which offers efficient framework for sustainable management of freshwater resources.

1. INTRODUCTION

Water quality is governed by a set of complex factors and there is large choice of variables use to describe water quality status in quantitative terms. Hence, it is difficult to provide a simple definition for water quality. Water quality of the aquatic environment is defined by (a) set of concentrations, speciation, and physical partitions of inorganic or organic substances, (b) composition and state of aquatic biota in the waterbody, and (c) description of temporal and spatial variations due to factors internal and external to the waterbody (Meybeck and Helmer, 1992). Water quality is also defined as a consequence of natural physical and chemical state of water (surface or subsurface) as well as alterations caused by human activities (Fetter, 1994). The quality of water is a measure of its suitability as a water supply source for domestic and agricultural consumption as well as for irrigation, industrial and other purposes; the suitability of water is decided based on criteria for various uses and water quality standards. The definition of water quality is therefore not objective; rather it is socially defined depending on the desired use of water. Different water uses require different standards of water quality and water quality criteria define desirable characteristics and acceptable levels of constituents for water of various intended uses (Freeze and Cherry, 1979; Todd, 1980; McCutcheon et al., 1993; Fetter, 1994). To establish quality criteria, the measures of *physical*, *chemical*, and *biological* constituents must be specified, together with standard methods for comparing results of water quality analyses (Todd, 1980; McCutcheon et al., 1993).

The pollution of the aquatic environment can be defined as introduction of substances or energy by man, directly or indirectly, which result in such deleterious effects as harm to living resources, hazards to human health, hindrance to aquatic activities including fishing, impairment of water quality with respect to its use in agricultural, industrial and often economic activities (Meybeck and Helmer, 1992), and reduction of amenities (GESAMP, 1988). The term *pollution* refers to changes caused by humans and their actions that result in water-quality conditions that negatively impact the integrity of the water for beneficial purposes, including natural ecosystem integrity (Johnson, 2009). Determining the extent of pollution is difficult, given the wide range of constituent measures that characterize water quality (e.g., dissolved and suspended solids, organics, bacteria, toxics, and metals). Evaluation of water quality, assessment of spatial and temporal variations and its vulnerability mapping are among the important tasks in order to manage quality of the useful water resources. There are many tools and techniques for evaluating the water quality and geographic information system is one of them, which is gaining a wide popularity now-a-days because of several advantages of the technique.

Geographic Information System (GIS) has emerged as a powerful tool for capturing, storing, analyzing, manipulating, retrieving and displaying spatial data and using these data for decision making in several areas including engineering and environmental fields (e.g., Stafford, 1991; Goodchild et al., 1993; Burrough and McDonnell, 1998; Lo and Yeung, 2003). It allows for swift organization, quantification and interpretation of a large volume of spatial data with a computer accuracy and minimal risk of human errors. GIS is an effective tool for analyzing spatial and temporal data of water quality (Burrough and McDonnell, 1998; Gurnell and Montgomery, 2000; Chang, 2002; Chen et al., 2004). Information on spatial and temporal variability/trends of water quality is very helpful in the decision-making process (Freeze and Cherry, 1979; Todd, 1980; Fetter, 1994). In addition, water quality mapping is essential for monitoring, pollution hazard assessment, modeling and environmental change detection (Goodchild et al., 1993; Skidmore et al., 1997; Chen et al., 2004; Jha et al., 2007). In a GIS framework, point estimates of water quality parameters can be spatially interpolated by spatial interpolation techniques such as kriging, inverse distance weighting, etc. to develop parameter concentration maps at different time scales or other related maps. GIS presents spatial information in the form of maps where different features are located by symbols, and is integrated with databases containing multiple attributes' data of the mapped features. A map helps providing knowledge of where and what things are, and how they are related. The GIS database containing spatial and point attributes can then be used to generate interactive reports and maps, which in-turn can support decision-making about the best design alternatives and their impacts. Furthermore, GIS-based maps serve as powerful communication medium in presenting information in such a way that the people involved in the planning and management of water quality can better understand and get more involved.

This chapter deals with various methods, i.e., statistical properties, vulnerability mapping, water quality indices, etc. for water quality evaluation using integration of the GIS technique. In all the methods, central role of GIS technique in water quality evaluation is highlighted.

2. POINT AND NON-POINT SOURCES OF POLLUTION

GIS plays a central role in water quality management practice and augments efforts to monitor water quality changes in surface waterbodies or aquifers, to calculate pollutant concentrations and loads to a surface waterbody or groundwater, and model water quality of aquatic systems (Johnson, 2009). Water quality protection and management require quantity of the waste-assimilative capacity of receiving waters to be known, which is determined using the concept of *'total mass daily loading'* (TMDL). A TMDL is assessed taking account of all sources of a pollutant, from both point and nonpoint sources, and the waste assimilative capacity of the receiving water body (USEPA, 1991). Water quality evaluations require a broad range of environmental and administrative data and one of the major categories of data include pollutant sources. Pollution may result from point sources or non-point sources (diffuse sources). Point sources are clearly identified at a single or multiple locations such as wastewater flow in conduits from municipalities and industries. However, nonpoint sources are diffuse and may not be defined by certain point locations for pollution such as urban runoff, erosion from agricultural and deforested lands. In other words, non-point sources include everything else that is not a point source. Sometimes, it is difficult to distinguish between both the point and non-point sources of pollution because a diffuse source on a regional or local scale may result from a large number of individual point sources, such as automobile exhausts. An important difference between a point and a diffuse source is that a point source is amenable to control through collection and treatment processes while a non-point source is difficult to control with engineered facilities, e.g. collection and treatment, because of diffuse character of this source. A diffuse pollution source consisting of several point sources may also be controlled provided all point sources can exactly be identified. Most common point and non-point sources of pollution are listed in Table 1.

3. WATER QUALITY EVALUATION

Water quality evaluation is an overall process of evaluating physical, chemical and biological nature of water in relation to natural quality, human effects and intended uses particularly uses which may affect human health and the health of the aquatic system itself (Bartram and Ballance, 1996). Water quality evaluation includes the use of monitoring data to define the condition of water, to provide a basis for detecting trends and to provide information enabling the establishment of cause-effect relationships. Thus, important aspects of water quality assessment are: *interpretation* of water quality data, *reporting* of results, and *recommendations* for future actions. Three important components of water quality evaluation in a logical sequence are monitoring, followed by assessment, followed by management (Meybeck et al., 1992).

The process of water quality evaluation involves many complex operations, which are linked together forming a chain of about twelve links where every link is important as its failure will weaken the entire evaluation. Elements of various water quality evaluation programmes may differ depending upon the objectives of the programme. However, there are certain standard elements, which are common to almost all type of water quality evaluation programmes. A generalized structure of water quality evaluation programme consisting of twelve elements is shown in Fig. 1. Prior to designing a water quality evaluation programme, clear-cut objectives should be set on the basis of environmental conditions (pollution sources), water uses (present and future), and water legislation. Once the programme objectives are set, monitoring design is determined based on review of existing water quality data, which is supported by preliminary survey. In next step, various monitoring operations are performed to collect water samples from selected sites in the field, and then, the collected samples are analysed in laboratory. The last step, which is important but often underestimated or omitted in a program, is synthesis, compilation, presentation and interpretation of enormous chemical data in a convenient manner for visual inspection (Freeze and Cherry, 1979; Sara and Gibbons, 1991). On completion of the programme, recommendations should be communicated to relevant water authorities for water management, water pollution control, and eventually the adjustment or modification of monitoring activities.

Table 1. Summary of point and non-point sources of pollution (after Johnson, 2009)

Point Sources	**Non-Point Sources**
1. Municipal and industrial wastewater effluents	1. Return flow from irrigated agriculture and orchards
2. Runoff and leachate from solid-waste disposal sites	2. Runoff from crops, pasture, and rangelands
3. Runoff and drainage from animal feedlots	3. Runoff from logging operations, including logging roads and all-terrain vehicles
4. Runoff from industrial sites	4.Urban runoff from small communities and unsewered settlements
5. Storm sewer outfalls from urban centers	5. Drainage from failing septic tank systems
6. Combined sewer overflows and treatment plant bypasses	6. Wet and dry atmospheric fall-out or deposition over waterbodies (e.g., acid rain)
7. Mine drainage and runoff (also oil fields)	7. Flow from abandoned mines and mining roads
8. Discharges from storage tanks, chemical waste piles, and ships	8. Runoff and snowmelt from roads outside urban areas
9. Runoff from construction sites	9. Wetland drainage
10. Airport snowmelt and runoff from deicing operations	10.Mass outdoor recreation and gatherings
	11. Military training, manoeuvres, shooting ranges

4. TOOLS FOR WATER QUALITY ANALYSIS

Several conventional tools for the graphical analysis of water quality are described in standard textbooks on groundwater hydrology or hydrogeology (Freeze and Cherry, 1979; Karanth, 1987; Sara and Gibbons, 1991). Recently, the need for application of modern approaches and tools such as multivariate statistical techniques (e.g., principal component analysis, hierarchical cluster analysis, discriminant analysis and correspondence analysis), and remote sensing and GIS techniques have been emphasized for the efficient analysis of water quality (e.g., Jha et al., 2007; Steube et al., 2009). The state-of-the-art review of tools and techniques for the interpretation of water quality can be found in Machiwal and Jha

(2010) wherein available tools and techniques (conventional as well as modern) for analyzing water quality are classified into four major groups: (i) graphical, (ii) statistical, (iii) remote sensing (RS), geographic information system (GIS) and geostatistical, and (iv) modelling techniques.

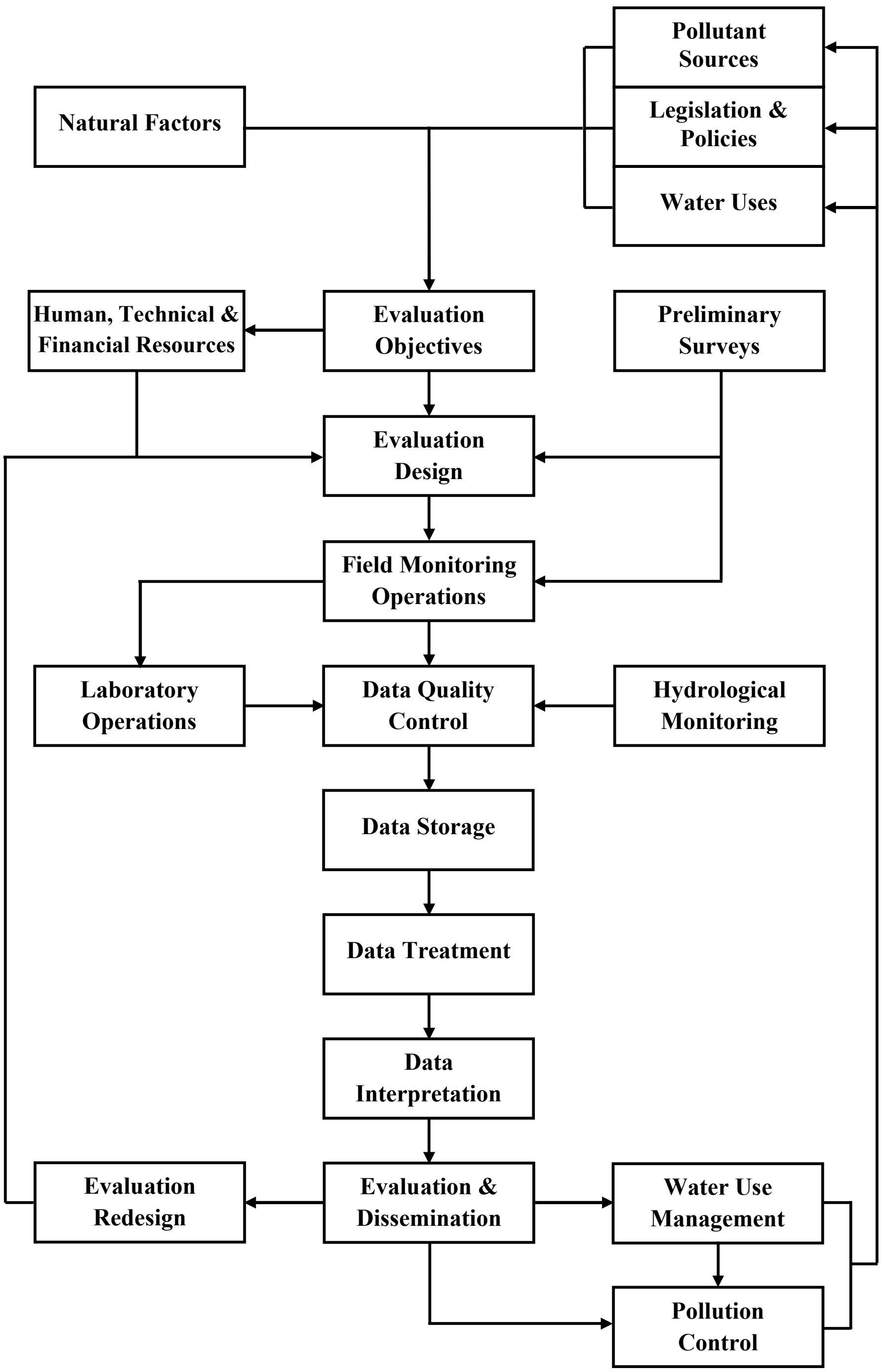

Fig. 1. Generalized framework of water quality evaluation program showing standard aspects (modified from Meybeck and Helmer, 1992).

5. GIS-BASED ASSESSMENT OF WATER QUALITY VARIABILITY

Extensive literature search made by the authors of this chapter revealed that most studies dealing with GIS applications for evaluating water quality are focused on subsurface water compared to that on surface water. This is most likely due to the relatively easy availability of large number of point groundwater samples through wells (hand pump, open well, tubewell, etc.). However, sampling of surface waterbodies requires some mechanism (e.g., boat) to reach different points in the waterbody. Another possible factor for the vast GIS application in groundwater quality studies is the significant variations in the water quality over a short distance within an aquifer. This is, in general, not the case of surface waterbody where water at point is free to move and mix with water at other points and this causes relatively less spatial variability especially in stagnant water of small ponds and reservoirs. Also, the water stored by the surface waterbodies is mostly the rainwater containing less concentration of the major ions as the water on surface has the least chances to come across different geological terrains comprising of certain minerals and other substances. On the other side, groundwater passing and moving through the subsurface formation meets different kind of salts, minerals, etc. which are easily dissolved with the flowing water. Thus, chances of increased concentration of the major ions and other metal contents are relatively higher for the groundwater as compared to surface water. This, perhaps, may be one of the causes that groundwater quality studies cite large application of the GIS techniques.

Spatial and temporal variability of the water quality is one of main features of different types of surface and subsurface waterbodies. Water quality variations over space and time are largely determined by hydrodynamic characteristics of the waterbody. Water quality of a waterbody varies over a space in all three dimensions, which are further altered by flow direction, discharge and time (Meybeck and Helmer, 1992). Thus, one location measurements in a waterbody may not be appropriately represents the water quality of entire waterbody. Instead, one network or grid of sampling sites would be needed to present spatial variations of the water quality. Generally, one-dimensional samples are collected on a longitudinal profile in case of river and on a vertical profile in case of pond/reservoir/lake as illustrated in Figs. 2(a,b). Two-dimensional profile sampling is appropriate for observing plumes of pollution from a source and this is most-suitable for groundwater quality of aquifers (Fig. 2c). Temporal variability of chemical water quality can be defined into five categories based on time scale as listed in Table 2.

5.1 Characterizing Extent and Patterns of Contamination

In water quality studies, activities begin with field data collection where it is a common practice to obtain the data from multiple locations and sources. All the collected data need to be collated and converted into common format of the water quality database. GIS provides excellent and powerful functions to capture and collate the water quality data. It is also seen that water quality studies involving repetitive, archival and historic use of the data requires the data be stored in a formal database that can be used for exploratory purposes. Water quality database to be utilized for GIS applications requires spatial coordinates, i.e. latitude and longitude (or x- and y- coordinates) to be attached with the data. The water quality data is further characterized by the depth at which the sample is taken (vertical z-coordinate). Monitored data must also be characterized with regard to time t at which sample is taken. Thus, concentration (c) of any physical, chemical and biological parameter can be defined by the following function:

$$c = f(x, y, z, t) \tag{1}$$

In surface waterbodies such as rivers where discharge (Q) is a significant quantity, the flux determination and data interpretation also require knowledge of water discharge, and thus the concentration should also be a function of Q as shown below.

$$c = f(x, y, z, t, Q) \tag{2}$$

Firstly, the sampling locations based on their spatial coordinates are located within GIS environment. Then, spatial locations of the sampling points are attached with related attribute tables where different attributes of all individual sites/points are stored such as concentration of all major ions, calcium, magnesium, chloride, carbonate, etc. for one spatial point is stored. Finally, the concentrations of a water quality parameter can be displayed over the entire space through spatial interpolation. There has been evolved a lot of spatial interpolation techniques over the time. An overview of the spatial interpolation techniques is provided in subsequent section.

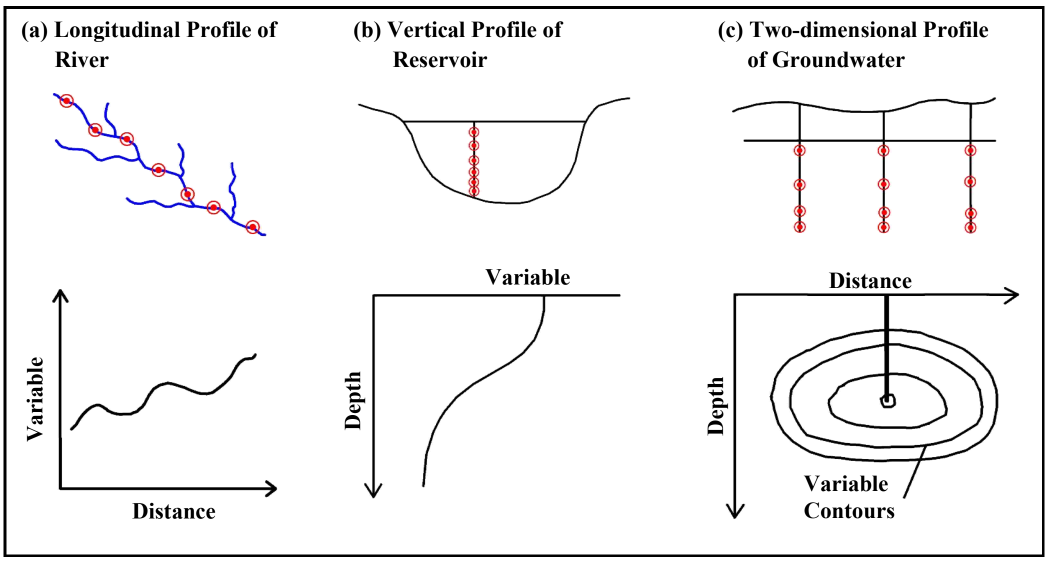

Fig. 2. Sampling strategies for exploring spatial variations of water quality in (a) river, (b) reservoir and (c) groundwater aquifer (modified from Meybeck and Helmer, 1992).

5.2 Overview of Spatial Interpolation Techniques

In numerical analysis, spatial interpolation or multivariate interpolation is interpolation on multivariable functions. The spatial interpolation consists of interpolating the multivariable function, known at given points, to yield values at arbitrary points. Most hydrogeologic applications of spatial interpolation involve quantities that vary in space but the methods may also apply to quantities that vary in time (Kitanidis, 1999). If function values are known on non-uniform grid, then available methods are nearest neighbor interpolation, natural neighbor, inverse distance weighting, kriging (one of the geostatistical techniques), and radial basis function (e.g., Gotway et al., 1996; Robinson and Metternicht, 2006; Namgial and Jha, 2009). Past studies dealing with GIS applications in water quality have mostly used geostatistical modeling and inverse distance weighting techniques for spatial interpolation. It is also revealed from the literature that geostatistical modeling tool was originally developed to deal with subsurface studies, and is widely-used for hydrogeologic studies.

Table 2. Scale of temporal water quality variability and causing factors (after Bartram and Ballance, 1996)

Scale of Temporal Variability	Causing Factor
Minute-to-minute to day-to-day variability	water mixing, fluctuations in inputs, etc., mostly linked to meteorological conditions and water body size (e.g. variations during river floods)
Dual variability (24-hour variations)	biological cycles, light/dark cycles etc. (e.g. O2, nutrients, pH), and to cycles in pollution inputs (e.g. domestic wastes).
Days-to-months variability	climatic factors (river regime, lake overturn, etc.) and to pollution sources (e.g. industrial wastewaters, run-off from agricultural land).
seasonal hydrological and biological cycles	mostly in connection with climatic factors
Year-to-year trends	human influences

5.2.1 Overview of Geostatistical Modeling Technique

Geostatistical modelling is a set of statistical estimation techniques involving quantities which vary in space (i.e., spatial variables). Geostatistical techniques for describing and interpolating spatially correlated data take advantage of the general observation that, on average, values closer together in space will be more similar than those farther from each other. The steps in applying these techniques include developing 'theoretical semi-variogram models' that describe the spatial variation between pairs of spatially or temporally related samples and then using these models to estimate sample parameters and their error variances at unknown locations. Although geostatistical modelling techniques were originally used in geological sciences (Journel and Huijbregts, 1978), they have also been frequently applied in hydrological, agricultural and ecological sciences to evaluate spatial dependence of surface/subsurface properties and ecological communities, or to interpolate these parameters (e.g., Goovaerts, 1999; Castrignanò et al., 2000; Mouser et al., 2005; Schaefer and Mayor, 2007).

The process of applying GIS and geostatistical modelling techniques for developing a spatial distribution map of a water quality variable is illustrated in Fig. 3.

(a) Spatial Estimation by Kriging Technique

In geostatistics, if $Z(x)$ represents any random function for concentration of any water quality variable measured at n locations in space $z(x_i)$, $i = 1, 2, \ldots n$ and if the water quality of the function Z has to be estimated at the point x_0, which has not been measured, the kriging estimate is defined as (Journel and Hujibregts, 1978; Kitanidis, 1997):

$$Z^*(x_0) = \sum_{i=1}^{n} \lambda_i \, z(x_i) \tag{3}$$

Where, $Z^*(x_0)$ = estimation of function $Z(x)$ at point x_0 and λ_i = weighting factors that minimize the variance of the estimation error (ordinary kriging weights).

Now two conditions are imposed to Eqn. (3), i.e., the unbiased condition and the condition of optimality. The unbiased condition means that the expected value of estimation error or the mean difference between the estimated $z^*(x_0)$ and the true (unknown) $z(x_0)$ value of the concentration of water quality variable should be zero. The condition of optimality means the variance of the estimation error should be minimum.

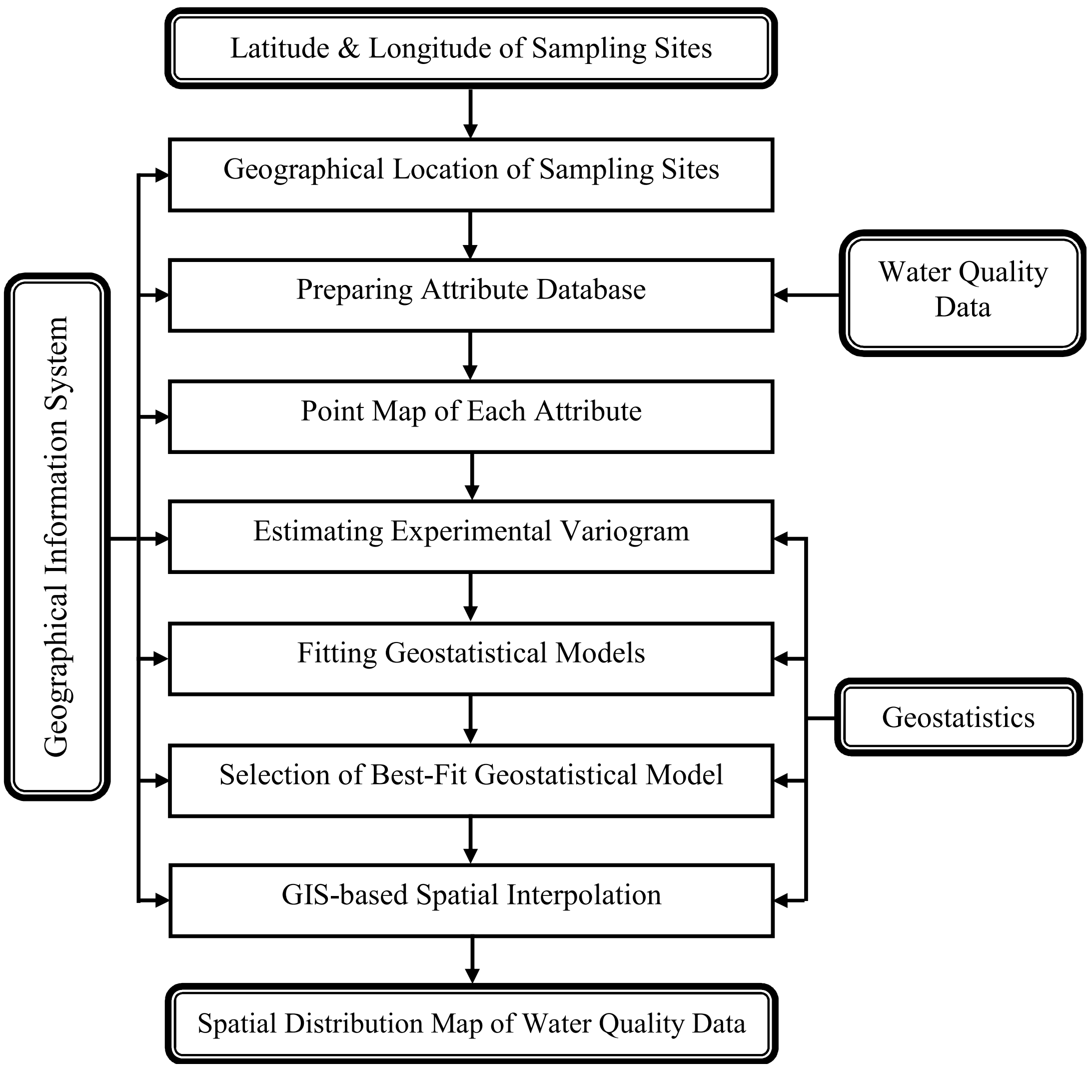

Fig. 3. Flowchart showing step-by-step methodology for applying GIS and geostatistical techniques for generating maps of water quality data.

The spatial structure defined by theoretical variogram, a kriging system of linear equations combining neighbouring information can be defined as

$$\sum_{j=1}^{n} \lambda_j C(x_i, x_j) - \mu = C(x_i, x_0) \quad i = 1, 2, \ldots n \tag{4}$$

subjected to the constraint on weights:

$$\sum_{j=1}^{n} \lambda_j = 1 \tag{5}$$

Where, μ = Lagrangian multiplier and $C(x_i, x_j)$ = value of covariance between two points x_i and x_j.

When we deal with an intrinsic case, i.e., working with variogram, the kriging Eqns. (4) and (5) are simply modified as follows (Marsily, 1986; Ahmed, 2006):

$$C(x_i, x_j) = C(0) - \gamma(x_i, x_j) \tag{6}$$

$$C(x_i, x_0) = C(0) - \gamma(x_i, x_0) \tag{7}$$

Eqns. (6) and (7) hold good only when both the covariance and the variogram exist, i.e., variables are stationary.

(b) Geostatistical or Variogram Models

Experimental geostatistical or variogram model is the function of separation vector between two points i and j. The values of separation vectors, e.g., h_1, h_2 etc. are decided first such that

$$h = |x_i - x_j| \tag{8}$$

Depending upon the value of h, the data are grouped into pairs and some function as defined below is averaged to obtain a variogram (γ_{ij}) (Goovaerts, 1997):

$$\gamma(h) = \frac{1}{2N_h} \sum_{i=1}^{N_h} \{z(x_i) - z(x_i + h)\}^2 \tag{9}$$

Where, N_h = number of pairs for a given lag distance h.

A theoretical geostatistical or variogram model (Fig. 4) can be defined essentially by 'sill' and 'range'. 'Sill' is the constant value on the y-axis around which a variogram stabilizes after a large distance and 'range' is the value at x-axis at which the variogram becomes constant or nearly constant. The sill value is usually very close to the variance of the variable (Matheron, 1965; Ahmed, 2006). In addition, the sudden apparent jump near the origin that occurs in some cases is known as 'nugget' effect. The shape of the variogram between origin and the point of stabilization is different for different variables, which entirely depends on its nature of variability (Matheron, 1965). In order to understand spatial structure, experimental water quality data are classified into lag distances with approximately the same number of data and semi-variogram values are calculated for each class (denoted by individual points shown in Fig. 4) using geostatistics or GIS software packages such as MathWorks, GSLib, GSTAT, GeoPack, ILWIS, ArcGIS, IDRISI, etc.

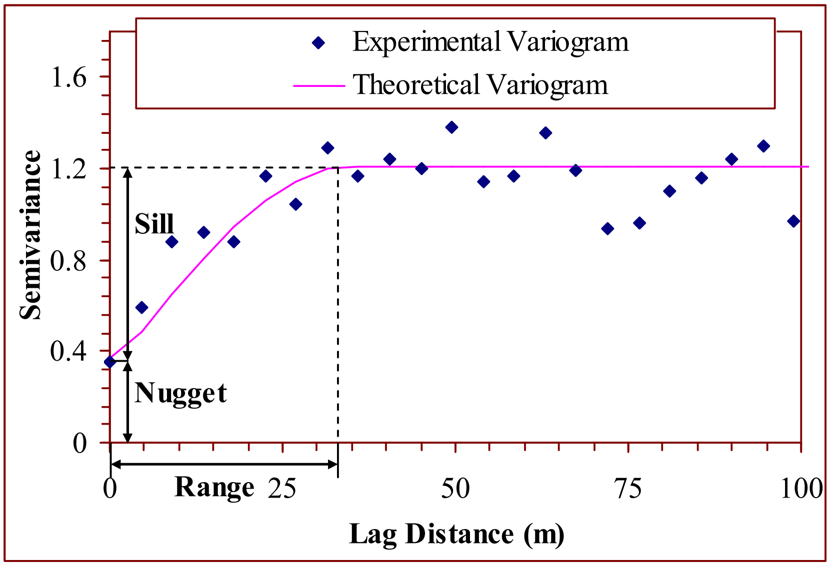

Fig. 4. Fitting of the theoretical variogram to an experimental variogram.

(c) Fitting of Theoretical and Experimental Variograms

The experimental variogram calculated from the observed water quality data using Eqn. (9) is usually an erratic curve (Kitanidis, 1997, 1999). It is not possible to use this experimental variogram in the estimation purpose due to its inconsistent nature. Therefore, the curve of the experimental variogram is approximated by another theoretical curve with a defined mathematical expression. This smooth curve fitted to the experimental variogram is known as 'theoretical variogram' as shown in Fig. 4. This fitting or modeling is performed in several ways mostly visual or using some form of difference between the two variograms but on a trial and error basis. Sometimes an automatic modeling is proposed but is not proved to be very useful. The commonly used variogram models are: spherical, circular, Gaussian, and exponential (Issaks and Srivastava, 1989; Kitanidis, 1997). The mathematical expressions for these theoretical variogram models are given below.

(i) Spherical Model:

$$\gamma(h) = C_0 + C\left(\frac{3h}{2a} - \frac{h^3}{2a^3}\right) \qquad \text{for } 0 < h \le a \qquad (10)$$

$$\gamma(h) = C_0 + C \qquad \text{for } h > a \qquad (11)$$

(ii) Circular Model:

$$\gamma(h) = C_0 + C\left[1 - \frac{2}{\pi}\, arc\cos(h/a) + \frac{2h}{\pi a}\sqrt{1-(h/a)^2}\right] \qquad \text{for } 0<h\leq a \qquad (12)$$

$$\gamma(h) = C_0 + C \qquad \text{for } h>a \qquad (13)$$

(iii) Gaussian Model:

$$\gamma(h) = C_0 + C\left[1 - e^{-(h/a)^2}\right] \qquad (14)$$

(iv) Exponential Model:

$$\gamma(h) = C_0 + C\left(1 - e^{-h/a}\right) \qquad (15)$$

Where, C_0+C is the sill, a is the range, and h is the separation vector or lag distance.

(d) Selection of the Best-Fit Model

Once the fitting of experimental and theoretical variograms is over, the best-fit geostatistical model can be selected based on a set of goodness-of-fit criteria viz., mean error (ME), root mean squared error (RMSE), correlation coefficient (r), mean standard error (MSE), mean reduced error (MRE), reduced variance ($S^2_{R_e}$), and coefficient of determination (r^2). The details about these goodness-of-fit criteria can be found in Table 3.

5.2.2 Inverse Distance Weighting Technique

Inverse distance weighting (IDW) technique is one of the moving average methods for spatial interpolation. Moving average method performs a weighted averaging on point values and returns a spatial map as output based on a specified weight function and a limiting distance (Webster and Oliver, 2001). While applying the IDW technique for interpolating values of any water quality variable for an output point, the distances of all points (where the water quality parameter is known) towards the output point are calculated to determine weight factors for the points. The weight factors for the points are then calculated according to the specified weight function. Two weight functions are available (Burrough and McDonnell, 1998): inverse distance and linear decrease. Weight for the inverse distance function is expressed below:

$$\text{Weight} = (1/\text{d}^{\text{n}}) - 1 \qquad (16)$$

Where, d = D/D_0 = relative distance of a known water quality point to output point, D = Euclidean distance of known water quality point to output point, D_0 = limiting distance, and n = weight exponent.

The weights vary according to the relative distance of any known water quality point to output point and the weight exponent (Fig. 5).

Thereafter, for each output pixel, value of particular water quality variable is calculated as the sum of the products of calculated weight values and point values divided by the sum of weights. That is,

$$WQ = \sum_{i=1}^{p}(w_i \times v_i) / \sum_{i=1}^{p} w_i \tag{17}$$

Where, WQ = value of concerned water quality variable, w_i = weight value for the i^{th} point, v_i = point value of the i^{th} point, and p = total number of points within the limiting distance.

Table 3. Summary of the goodness-of-fit criteria

S. No.	Goodness-of-fit Criteria	Equation
1	Mean Error (ME)	$ME = \frac{1}{n}\sum_{i=1}^{n}[z(x_i) - z^*(x_i)]$ Where, $z(x_i)$ and $z^*(x_i)$ = observed and estimated values of variable z at the location x_i, and n = number of data points.
2	Root Mean Squared Error (RMSE)	$RMSE = \sqrt{\frac{\sum_{i=1}^{n}[z(x_i) - z^*(x_i)]^2}{n}}$
3	Correlation Coefficient (r) (Rodgers and Nicewander, 1988)	$r = \frac{n\sum_{i=1}^{n}[z(x_i) \cdot z^*(x_i)] - \left[\sum_{i=1}^{n} z(x_i) \cdot \sum_{i=1}^{n} z^*(x_i)\right]}{\left[\sqrt{n\left\{\sum_{i=1}^{n}[z(x_i)^2]\right\} - \{z(x_i)\}^2}\right] \cdot \left[\sqrt{n\left\{\sum_{i=1}^{n}[z^*(x_i)^2]\right\} - \{z^*(x_i)\}^2}\right]}$
4	Mean Standard Error (MSE)	$MSE = \frac{1}{n}\sigma_k(x_i)$ Where, $\sigma_k(x_i)$ = estimation variance at the location x_i.
5	Mean Reduced Error (MRE) (Vauclin et al., 1983)	$MRE = \frac{1}{n}\sum_{i=1}^{n}[z(x_i) - z^*(x_i)] / \sigma_k(x_i)$
6	Reduced Variance ($S^2_{R_e}$) (Vauclin et al., 1983)	$S^2_{R_e} = \frac{1}{n}\sum_{i=1}^{n}\left[\{z(x_i) - z^*(x_i)\} / \sigma_k(x_i)\right]^2$
7	Coefficient of Determination (r^2) (Draper and Smith, 1998)	$r^2 = 1 - \frac{SSE}{(SSR + SSE)}$; Where, $SSE = \sum_{i=1}^{n}[z(x_i) - z^*(x_i)]^2$ and $SSR + SSE = \sum_{i=1}^{n}[z(x_i) - \bar{z}(x_i)]^2$ Where, $\bar{z}(x_i)$ = mean of $z(x_i)$.

5.3 Statistical Measures in Spatial Context

Availability of multiple observations of water quality attributes both at spatial and temporal scales provides an opportunity for exploring spatial and temporal variations of the water quality in an area. Spatial statistics of the water quality database (mean, median, standard deviation, coefficient of variation, etc.) can be easily computed in GIS. For example, if multiyear and multi-site water quality parameters are available for a given area. Then annual concentration maps for individual parameters and years can be prepared through spatial interpolation techniques (described earlier) in GIS framework. Subsequently, mean (C_{mean}) annual concentration map for any of the water quality parameter can be created by using following equations (Machiwal et al., 2011):

$$C_{mean,i} = \frac{\sum_{n=1}^{N} C_{n,i}}{N} \quad (18)$$

Where, $C_{mean,i}$ = mean annual concentration map of i^{th} water quality parameter, $C_{n,i}$ = annual concentration map of the i^{th} parameter in n^{th} year, and N = total number of years of data availability.

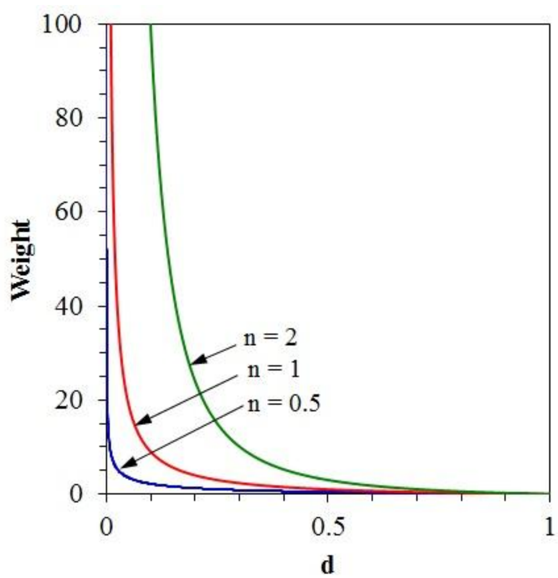

Fig. 5. Inverse distance weights for relative distance of point to output point.

In order to compute the median, first rank the annual observations from the smallest (C'_1) observation to the largest (C'_N) observation and then use one of the following equations depending on the number of observations (N):

$$C_{median,i} = C'_{(N+1)/2}\text{, when N is odd} \quad (19)$$

$$C_{median,i} = \frac{1}{2}\left\{C'_{(N/2)} + C'_{(N/2)+1}\right\}\text{, when N is even} \quad (20)$$

The spatial standard deviation map can be prepared from the following expression (Machiwal et al., 2011):

$$C_{sd,i} = \sqrt{\frac{\sum_{n=1}^{N}\left(C_{n,i} - C_{mean,i}\right)}{N-1}} \tag{21}$$

Where, $C_{sd,i}$ = spatially-distributed standard deviation map of the i^{th} parameter.

Thereafter, the coefficient of variation maps for different parameters can be developed using the following equation (Machiwal et al., 2011):

$$C_{cv,i}\left(\%\right) = \frac{C_{sd,i}}{C_{mean,i}} \times 100 \tag{22}$$

Hydrologic variables with larger CV values are more variable than those with smaller values. Wilding (1985) suggested a classification scheme for identifying the extent of variability for soil properties based on their CV values, where CV values of 0-15, 16-35 and >36 indicate little, moderate and high variability, respectively. Typical ranges of CV values of salient soil properties are reported in the literature (Jury, 1986; Jury et al., 1987; Beven et al., 1993; Wollenhaupt et al., 1997).

6. GIS FRAMEWORK FOR GROUNDWATER VULNERABILITY MAPPING

6.1 Groundwater Vulnerability Concept

The groundwater vulnerability concept, evolved during end of the 1960s in France, aimed at creating awareness of groundwater contamination (Margat, 1968; Albinet and Margat, 1970). The vulnerability concept in hydrogeology began to be widely used in the 1980s (Haertle, 1983; Aller et al., 1987). It was defined as the possibility of percolation and diffusion of contaminants from the ground surface into the groundwater system. Groundwater vulnerability deals only with the hydrogeological setting and does not include pollutant attenuation. Initially, the term 'vulnerability' was meant as relative susceptibility of aquifers to anthropogenic pollution without any formal definition. Later on, the concept began to mean different things to different people. Margat (1968) used the term 'vulnerability' to mean the degree of protection that the natural environment provides against the ingress of pollutants to groundwater. Thereafter, several definitions of vulnerability have been proposed. Foster (1987) defined aquifer pollution vulnerability as the intrinsic character of the strata separating the saturated aquifer from the immediately overlying land surface which determines its sensitivity to being adversely affected by a surface applied (anthropogenic) contaminated load. National Research Council (1993) defined groundwater 'vulnerability' to contamination as the tendency or likelihood for contaminants to reach a specified position in the groundwater system after introduction at some location above the uppermost aquifer. Vrba and Zaporocec (1994) defined 'vulnerability' as an intrinsic property of groundwater, depending on its susceptibility to natural and/or human impact. The groundwater vulnerability is a specific characteristic of the underlying groundwater system and cannot be practically measured in the field.

In general, the status of groundwater contamination is determined by the natural attenuation processes occurring within the zone between the pollution source and the aquifer. Mainly two natural factors, i.e. physical processes and chemical reactions occurring within the soil, unsaturated zone and saturated zone are responsible for alteration in physical states and chemical forms of contaminants, which ultimately leads to attenuation of contaminants. There may be a single or multiple chemical reactions to work with other processes resulting in a varying degree of attenuation. These reactions depend on the specific soil and aquifer characteristics and particular geochemical properties of each contaminant. Thus, groundwater vulnerability is a function of geology and hydrogeology of the unsaturated and saturated zones and physico-chemical properties of the contaminants. All factors affecting groundwater vulnerability may vary from one place to another. The groundwater vulnerability may be classified in two ways: intrinsic vulnerability and specific vulnerability. The term 'intrinsic vulnerability' refers to the vulnerability of groundwater to contaminants generated by anthropogenic or human activities taking into account the inherent geological, hydrological and hydrogeological characteristics of an area but being independent of the nature of the contaminants. On the other side, term 'specific vulnerability' is used to define the vulnerability of groundwater to particular contaminants or a group of contaminants taking into account the contaminant properties and their relationship with the various components of intrinsic vulnerability (Doerfliger et al., 1999; Gogu and Dassargues, 2000).

6.2 GIS-Based Methods to Evaluate Groundwater Vulnerability

Geographic Information System (GIS) technique has fundamentally changed our thoughts and ways to manage natural resources in general and water resources in particular (Jha et al., 2007). GIS is designed to collect diverse spatial data to represent spatially variable phenomena by applying a series of overlay analysis of data layers that are in spatial register (Bonham-Carter, 1996). Vulnerability assessment is a basis for initiating protective measures for important groundwater resources and will normally be the first step in groundwater pollution assessment (Foster et al., 2002). The GIS technique is of great significance in assessing the pollution vulnerability of the aquifers over a large area.

Many approach such as process-based methods, statistical methods, and overlay and index methods have been developed to evaluate aquifer vulnerability (Tesoriero et al., 1998). Variability of land vulnerability to groundwater contamination leads to mapping of groundwater vulnerability (Piscopo, 2001). In the process-based methods, simulation models are used to estimate the movement of contaminant in groundwater. The major drawback in using process-based methods is data shortage and computational difficulties (Barbash and Resek, 1996). In statistical methods, statistical terms are used to determine relations between spatial variables and actual occurrence of pollutants in the groundwater. Their major limitations include absence of sufficient water quality observations, data accuracy, and careful selection of spatial variables (Babiker et al., 2004). Overlay and index methods resulting in vulnerability indices mainly depend upon factors, which control the pollutant movement from the ground surface into the saturated zone. Their main advantage is that vulnerability assessments can be made at regional scale as some of the factors such as rainfall, soil type and groundwater depth are easily available over large areas, which makes them suitable to be used with geographic information system (Thapinta and Hudak, 2003). In general, overlay and index methods and statistical methods are used for contamination assessments at map scales smaller than 1:50,000 (i.e., a large study area), while process-based simulation models are at larger map scales (i.e., a small study area) (Rao and Alley, 1993). Overlay and index methods and statistical methods are used to assess intrinsic vulnerability, while methods based on simulation models are used to assess specific vulnerability.

6.3 Groundwater Vulnerability Mapping by GIS-Driven Overlay and Index Methods

The most common approach to quantify aquifer vulnerability at present is the overlay and index method, whereby the protective effect of the overlying layers is expressed in a semi-quantitative way (Frind et al., 2006). Overlay and index methods efficiently determine groundwater vulnerability. These methods deal with overlaying and aggregation of multiple spatial maps and these spatial analyses of a group of maps can easily be performed in geographic information system. Thus, the overlay and index methods are particularly suitable for use with geographic information systems (Tilahun and Merkel, 2010). An overlay and index method, being a multicriteria model, aggregates different hydrological/hydrogeological factors that control the movement of pollutants from the ground surface to underlying aquifer. A GIS-based overlay and index method combines factors controlling pollutant migration according to certain multi-criterion rule and computes resulting value of vulnerability index for different spatial locations. A general methodology for applying groundwater vulnerability methods in GIS framework is shown in Fig. 6.

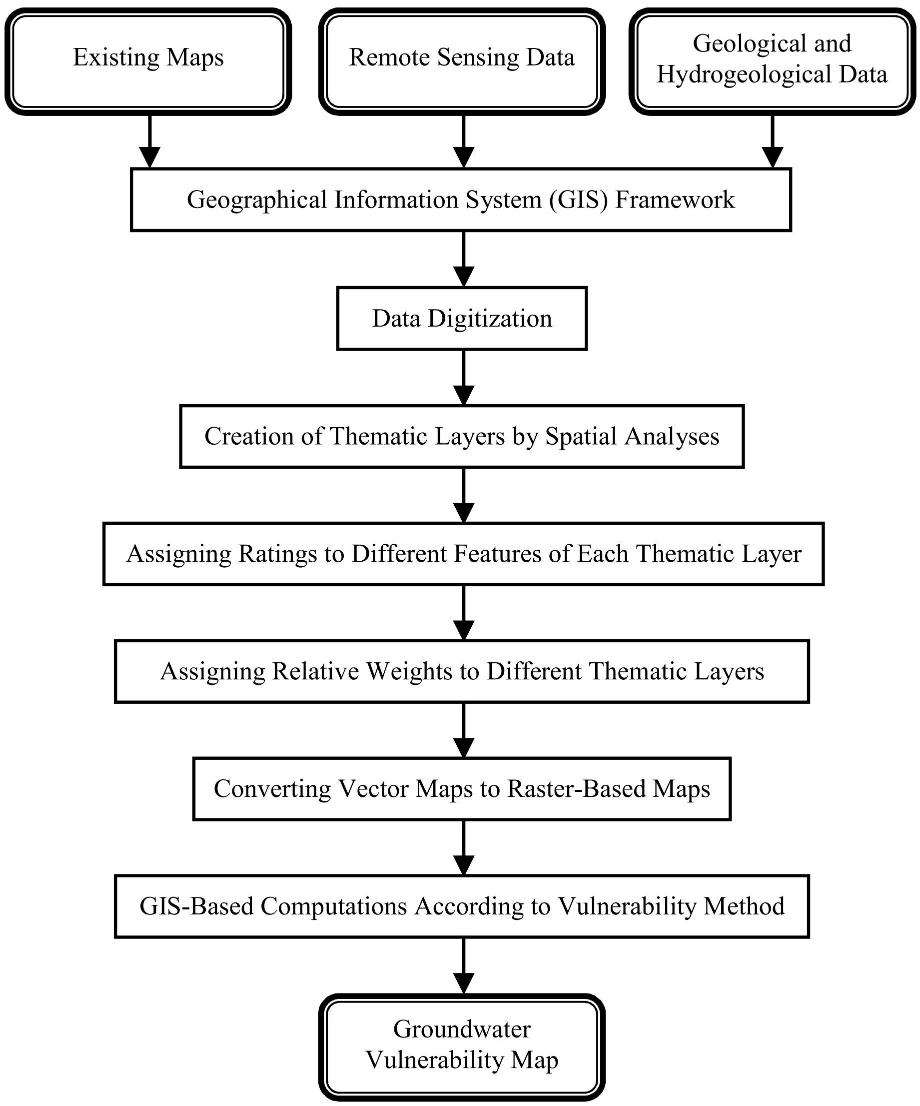

Fig. 6. General GIS-based methodology for groundwater vulnerability study.

Before 1980s, there had been several attempts to formulate and establish a methodology to assess the vulnerability in order to present it in a map. However, the successful results could be obtained during the mid 1980s when two of the pioneer indices called DRASTIC (Aller et al., 1987) and GOD (Foster, 1987) were reported. There are many kinds of vulnerability identified by different methods associated with a wide range of index values and labelled qualitatively. The categorization of vulnerability into different classes depends upon the index values and appropriate number of categories decided by a person. The groundwater vulnerability assessment has rapidly developed over the past 20 years; many new tools and techniques are introduced for the groundwater vulnerability assessment along with specific applications being thoroughly analyzed and tested for different environments (Cramer and Vrba, 1987; Meinardi et al., 1995; Secunda et al., 1998; Lasserre et al., 1999; Al-Adamat et al., 2003; Lake et al., 2003; Rodriguez et al., 2003; Thapinta and Hudak, 2003; GEAM, 2005; Allen and Milenic, 2007; Zhou et al., 2010). Moreover, many studies have used different scales and sources of information for the application of these techniques (Secunda et al., 1998; Foster et al., 2002; Civita and De Maio, 2004; Wang et al., 2007). The widely-used models of index methods include DRASTIC (Aller et al., 1987), GOD (Foster, 1987), AVI rating system (Stempvoort et al., 1993), SINTACS (Gogu and Dassargues, 2000) and EPIK (Doerfliger et al., 1999).

Conventional methods, e.g., DRASTIC, GOD, AVI, SINTACS, etc. do not take into account the peculiar features of karstic (or carbonate) geological formations. Thus, to address pollution vulnerability assessment in karstic aquifers, few specific methods, e.g., EPIK (Doerfliger and Zwahlen, 1998; Doerfliger et al., 1999), PI (Goldscheider et al., 2000) and COP (Vias et al., 2006) have been developed. Available methods (conventional as well as non-conventional) for groundwater vulnerability mapping can be classified into two groups as shown in Table 4.

Three major limitations of overlay and index methods are: (i) defining groundwater vulnerability in qualitative terms, which is opposed by quantitative terms (Gogu et al., 2003; Frind et al., 2006; Popescu et al., 2008), (ii) finding it difficult to quantify exact amount of uncertainty involved in vulnerability assessments in order to handle inaccuracies incurred in analysis (Gogu and Dassargues, 2000), and (iii) strong homogeneous results observed over large areas in many parts of the world, which restricts for discrimination and delimitation of areas of different vulnerability to pollution. These problems are addressed by the study reported by Massone et al. (2010), where different units with different categories of vulnerability in geological homogenous environments are discriminated. Also, use of qualitative adjectives such as ‘low’ or ‘moderate’ is avoided because of their subjective meaning.

6.3.1 DRASTIC Method

DRASTIC is one of the most widely used standard groundwater vulnerability methods, which was developed by the United States Environmental Protection Agency (USEPA) as a method for assessing groundwater pollution potential (Aller et al., 1987). Seven most important mappable factors that control groundwater pollution were determined after a complete evaluation of many characteristics and the mappability of the data; the parameters are as follows:

Table 4. Conventional and modern methods for groundwater vulnerability mapping

S. No.	Method	Parameters	Source
Methods for Porous Aquifers			
1	DRASTIC and Pesticide DRASTIC	D – Depth to water R – (net) Recharge A – Aquifer media S – Soil media T – Topography (slope) I – Impact of vadose zone C – (hydraulic) Conductivity of the aquifer	Aller et al. (1987)
2	DRAMIC	D – Depth to water R – (net) Recharge A – Aquifer media M – Aquifer thickness I – Impact of vadose zone C – impact of Contaminant	Wang et al. (2007)
3	GOD	G – Groundwater occurrence including recharge O – Overlying lithology D – Depth to groundwater	Foster (1987)
4	AVI	c – Hydraulic resistance	Stempvoort et al. (1992)
5	SINTACS	S – Depth to groundwater I – Recharge action N – Attenuation potential of the vadose zone T – Attenuation potential of the soil A – Hydrogeologic characteristics of the aquifer C – Hydraulic conductivity S – Topographic slope	Civita (1994)
Methods for Karstic (Carbonate) Aquifers			
6	EPIK	E – development of Epikarst P – Protective cover I – Infiltration condition K – Karst network development	Doerfliger and Zwahlen (1995)
7	GLA	S – effective field capacity of the soil (rating for FCe in mm down to 1 m depth) W – percolation rate R – rock type T – thickness of soil and rock cover above the aquifer Q – bonus points for perched aquifer systems HP – bonus points for hydraulic pressure conditions (artesian conditions)	Hoelting et al. (1995)
8	PI	P – Protective cover I – Infiltration conditions	Goldscheider et al. (2000)
9	COP	C – flow Concentration O – Overlying layers P – Precipitation	Daly et al. (2002)

D – Depth to Water
R – (Net) Recharge
A – Aquifer Media
S – Soil Media
T – Topography (Slope)
I – Impact of Vadose Zone
C – (Hydraulic) Conductivity of the Aquifer

These seven parameters are briefly described in Table 5. The DRASTIC index model can be used to identify areas that are more vulnerable to contamination than others, or to give priorities to areas that need more groundwater quality monitoring. It is a vulnerability index model designed to calculate vulnerability scores (numerical values) for different locations by combining seven thematic layers/factors. Before combining the factors, ratings and weights are assigned to the seven model parameters. The classes or features of each parameter represent the ranges, which are rated on the 1-10 scale based on their relative effect on the groundwater vulnerability; a rating of 10 indicating a high pollution potential of the parameter. Once the ratings are assigned to all classes of the parameters, the weights ranging from one to five reflecting their relative importance with respect to each other are assigned to seven parameters (Table 5). The DRASTIC Index is then computed applying a weighted linear combination of all seven parameters by multiplying each parameter rating with its weight and adding together the resulting values according to the following equation (Aller et al., 1987):

$$\text{DRASTIC}_{\text{Index}} = D_R D_W + R_R R_W + A_R A_W + S_R S_W + T_R T_W + I_R I_W + C_R C_W \quad (23)$$

Where D, R, A, S, T, I, and C are the seven parameters expressed above and the subscripts R and W are the corresponding ratings and weights, respectively.

DRASTIC provides two weight classifications (Table 5), one for general conditions and the other one for conditions with intense agricultural activity. The latter, called the Pesticide DRASTIC index (DRASTIC-P), represents a specific vulnerability assessment approach. The DRASTIC-P method is the most suitable in agricultural areas mainly due to the greater weight given to the variables of soil and slope types (Massone et al., 2007).

In recent years, the originally developed DRASTIC method has been modified by using additional parameters or factors and/or by ignoring the existing unimportant parameters according to the local characteristics of the study area (Fritch et al., 2000; Al-Adamat et al., 2003; Lee, 2003; Thirumalaivasan et al., 2003; Babiker et al., 2005; Simsek et al., 2006; Guo et al., 2007; Wang et al., 2007; Umar et al., 2009; Martínez-Bastida et al., 2010; Awawdeh and Jaradat, 2010). This reflects flexibility of the DRASTIC model to modify according to need of the study.

The well-established DRASTIC method has been applied in different parts of the world such as the United States (e.g., Rupert, 2001; Merchant, 1994; Loague and Corwin, 1998; Wade et al., 1998; Stark et al., 1999; Fritch et al., 2000), Canada (Murat et al., 2004), Europe (e.g. Stigter et al., 2006; Vias et al., 2005), South America (Tovar and Rodriguez, 2004; Herlinger and Viero, 2006), Australia (Piscopo, 2001), New Zealand (McLay et al., 2001), Asia (Al-Adamat et al., 2003; El-Naqa, 2004; Thirumalaivasan et al., 2003; Rahman, 2008; Kimand Hamm, 1999), and Africa (Lynch et al., 1997; Ibe et al., 2001). The DRASTIC model is applicable in humid climates (Babiker et al., 2005; Piscopo, 2001; Kim and Hamm, 1999; and Osborn et al., 1998) as well as in semi-arid to arid climates (Werz and Hötzl, 2007; Al-Adamat et al., 2003; Secunda et al., 1998).

Table 5. Description of DRASTIC and Pesticide DRASTIC parameters (after Aller et al., 1987)

Parameter	**Description**	**Relative Weight**	
		DRASTIC	**DRASTIC-P**
Depth to Water	Represents the depth from the ground surface to the water table, deeper water table levels imply lesser chance for contamination to occur.	5	5
Net Recharge	Represents the amount of water which penetrates the ground surface and reaches the water table, recharge water represents the vehicle for transporting pollutants.	4	4
Aquifer Media	Refers to the saturated zone material properties, which controls the pollutant attenuation processes.	3	3
Soil Media	Represents the uppermost weathered portion of the unsaturated zone and controls the amount of recharge that can infiltrate downward.	2	5
Topography	Refers to the slope of the land surface, it dictates whether the runoff will remain on the surface to allow contaminant percolation to the saturated zone.	1	3
Impact of Vadose Zone	Is defined as the unsaturated zone material, it controls the passage and attenuation of the contaminated material to the saturated zone.	5	4
Hydraulic Conductivity	Indicates the ability of the aquifer to transmit water, hence determines the rate of flow of contaminant material within the groundwater system.	3	2

In the original DRASTIC index model, semi-quantitative data layers were overlaid manually. However, the simple linear model of its combination factors expressing its vulnerability index shows the feasibility of employing the GIS for the computation of index (Fabbri and Napolitano, 1995). For past 15-20 years, the GIS technique has been widely used in groundwater vulnerability mapping (Evans and Myers, 1990; Loague et al., 1996; Hrkal, 2001; Rupert, 2001; Lake et al., 2003; Massone et al., 2010; Yin, 2013; Edet, 2014). The major advantage of GIS-based mapping is the best combination of data layers and rapid change in the data parameters used in vulnerability classification. Integration of DRASTIC method with GIS involves following four steps (Massone et al., 2010).

(i) Preparation of thematic base maps (as a polygonal entity) for each parameter under consideration using GIS software packages. Subsequently, polygon map of each parameter is transformed into raster format using the spatial analysis functions of GIS. A suitable spatial cell resolution for spatial analysis can be chosen.

(ii) Procedure indicated by methodology are applied for the assignment of weights and values to each layer of information and the application of map algebra to obtain the aquifer vulnerability maps, called DRASTIC and DRASTIC-P vulnerability maps. Conveniently, the DRASTIC index values can be discretized into suitable number of classes indicating very low, low, moderate, high and very high vulnerability, since this is the number of classes that allows one to recognize both the "best" values and the worst ones as two alternatives (high and very high or low and very low); this is better than recognizing only three classes where there is only one possible option towards each end (low or high). This is favourable to decision-making related to the use of soil in land-use planning, in environmental impact evaluations, etc.

(iii) Reclassification of the DRASTIC vulnerability maps to obtain the DRASTICpriorities, which recognize five classes from priority 1 (lower values in the series) to priority 5 (higher values).

(iv) Combining the DRASTIC vulnerability map with the DRASTIC-priorities to generate an operational vulnerability index (OVI). For this operation, both the vulnerability map and the DRASTIC-priorities are reclassified, assigning to each qualitative class a numerical value ranging from 1 (the lower class) to 5 (the higher one).

6.3.2 DRAMIC (modified DRASTIC) Method

The DRASTIC method, originally developed for rural/agricultural lands, had some limitations and/or required modifications while applying in urban areas. First, it was observed that the parameter C (hydraulic conductivity) of the DRASTIC method is closely related to the parameter A (aquifer media). Thus, impact of aquifer media has two-fold effect. Second, topography of most cities in urban areas remains relatively flat (with negligible slope), and therefore, the parameter T (topography) can be ignored from DRASTIC method. Third, in urban areas, the ground surface is mostly covered by built-up structures, concrete, etc. and it is quite difficult to obtain comparable values of the parameter S (soil media). To overcome these problems, and to improve the predictability and applicability of the DRASTIC method, Wang et al. (2007) proposed DRAMIC method. The method is expressed in Eqn. (24), where parameters of DRAMIC method and their respective assigned weights are shown. Four parameters, i.e. D, R, A and I of the DRAMIC method are same as in DRASTIC method; parameter T is deleted and the parameters S and C are replaced with two new parameters, i.e. aquifer thickness (M) and impact of contaminant (C). DRAMIC index is described as (Wang et al., 2007):

$$\mathrm{DRAMIC}_{\mathrm{index}} = 5\mathrm{D}_{\mathrm{R}} + 3\mathrm{R}_{\mathrm{R}} + 4\mathrm{A}_{\mathrm{R}} + 2\mathrm{M}_{\mathrm{R}} + 5\mathrm{I}_{\mathrm{R}} + 1\mathrm{C}_{\mathrm{R}} \quad (24)$$

Where, D, R, A, and I are the same as in the DRASTIC method; M = aquifer thickness defined by media; C = parameter showing impact of contaminant; and R = rating. The computed DRAMIC index values can be used to delineate areas, which are more susceptible to groundwater contamination compared to other areas. The higher the value of DRAMIC index is, the greater the vulnerability to groundwater pollution. The hydrogeological significance, ranges and ratings for the four factors D, R, A, and I of the DRAMIC method are the same as in DRASTIC methods. The ranges and ratings for the two new parameters of the DRAMIC method, e.g. aquifer thickness and contaminant characteristics are listed in Table 6.

Table 6: Ranges and ratings for aquifer thickness and contaminant characteristics for DRAMIC method (after Wang et al., 2007)

Aquifer Thickness		**Contaminant Characteristics**	
Range (m)	**Typical Rating**	**Characteristics**	**Rating**
0-6	9	Stable, easy to infiltrate into aquifer	9
6-15	7	Stable, relatively easy to infiltrate	7
15-25	5	Stable, uneasy to infiltrate	5
25-32	4	Relatively stable, easy to infiltrate	5
32-40	3	Relatively stable, relatively easy to infiltrate	4
40-50	2	Relatively stable, uneasy to infiltrate	3
>50	1	Unstable, easy to infiltrate	3
		Unstable, relatively easy to infiltrate	2
		Unstable, uneasy to infiltrate	1

6.3.3 GOD Scheme

GOD scheme is one of the earliest vulnerability index methods. GOD rating system is an empirical method for quick assessment of vulnerability incorporating three parameters: **G**roundwater occurrence including recharge, **O**verlying lithology, and **D**epth to groundwater (Foster, 1987). To each category of these parameters – namely the aquifer type (e.g. confined, semi-confined, unconfined), the lithology of the overlying aquitard or aquiclude (in case of a confined or semi-confined aquifer) or the aquifer unsaturated zone (in case of unconfined aquifer) and the depth to water – a rating value between 0 (not vulnerable) to 1 (highly vulnerable) is assigned. The vulnerability index is calculated using the following formula (Foster, 1987):

$$GOD_{index} = G_R \times O_R \times D_R \qquad (25)$$

Where, G = groundwater occurrence, O = overlying lithology (only in case of unconfined aquifer), and D = depth to groundwater, and subscript R indicates rating of the parameters. Schematic of the GOD system for assessing aquifer pollution vulnerability index is shown in Fig. 7. The index values also ranges from 0 to 1 and gives the overall pollution vulnerability. The vulnerability is classified into four classes according the index values as (i) low (GOD index<0.3), (ii) moderate (0.3<GOD index<0.5) and (iii) high (0.5<GOD index<0.7) and (iv) extreme (GOD index>0.7). The method could not get wide popularity, although its performance has been assessed by applying it in GIS platform in some recent past studies (Debernardi et al., 2008; Polemio et al., 2009; Kazakis and Voudouris, 2011).

6.3.4 AVI Rating System

In this method, two physical parameters are considered: the thickness of every sedimentary unit above the uppermost, saturated aquifer surface (d) and the estimated hydraulic conductivity of each of these sedimentary layers (k). Firstly, hydraulic resistance (c) is calculated by the following equation (Stempvoort et al., 1992):

$$c = \sum_{i=1}^{n} \frac{d_i}{k_i} \tag{26}$$

Where, n = number of sedimentary units above the aquifer; d = thickness of each sedimentary unit above the uppermost aquifer, and k = estimated hydraulic conductivity of each sedimentary unit. The parameter c is defined as a theoretical factor used to describe the resistance of an aquitard to vertical flow (e.g. Kruseman and de Ridder, 1990). The hydraulic resistance (c) has dimension of time, which indicates the approximate travel time for water to move by advection downward through the various porous media above the uppermost saturated aquifer surface. However, it should be noted that, in a strict sense, c is not a travel time for water or contaminants. The calculated c or log(c) values can be used directly to generate iso-resistance map by using geostatistical techniques for spatial interpolation in GIS. The parameter c is related to a qualitative Aquifer Vulnerability Index (AVI) by a relationship, shown in Table 7.

The AVI rating system of aquifer vulnerability index, originally developed and applied in Canada (Stempvoort et al., 1992), has been demonstrated by a successful application in a GIS environment by Stempvoort et al. (1993).

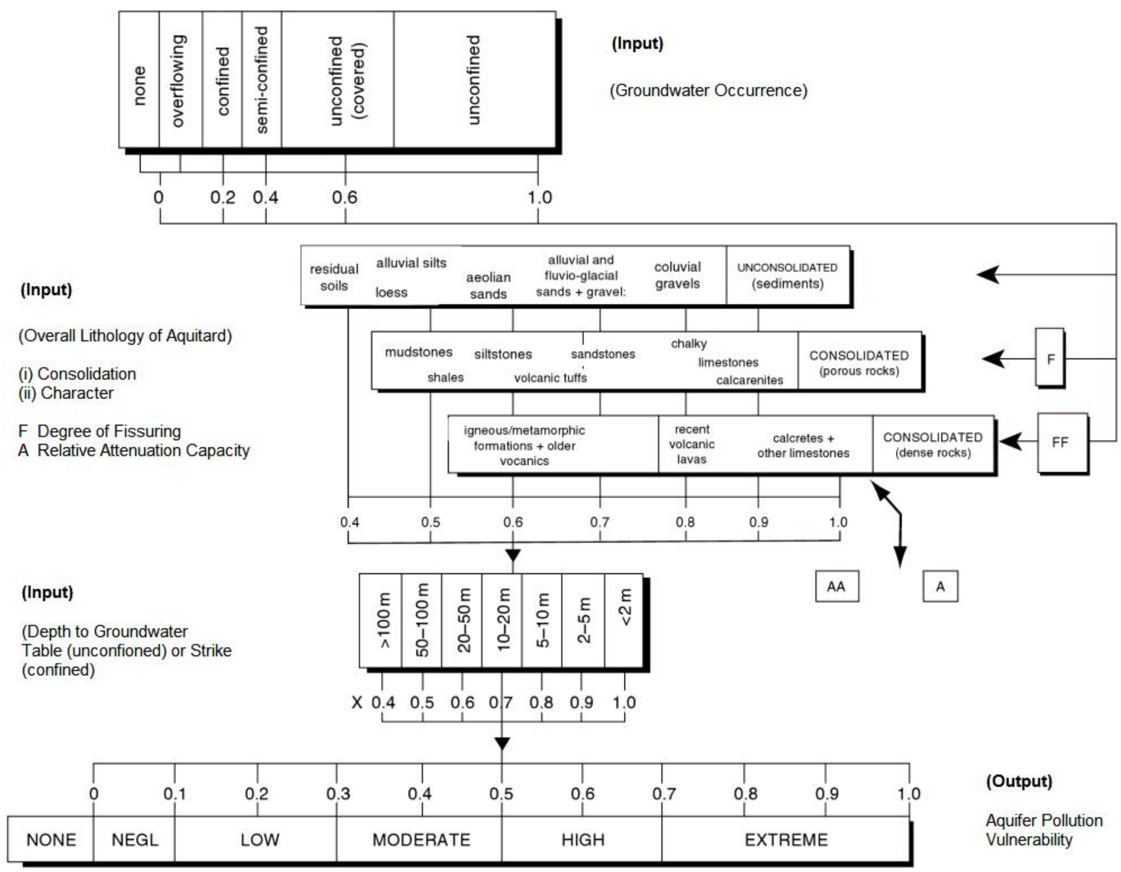

Fig. 7. Schematic of GOD method for assessing aquifer pollution vulnerability (after Foster, 1987).

Table 7. Relationship between Aquifer Vulnerability Index (AVI) and hydraulic resistance (after Stempvoort et al., 1992)

Hydraulic Resistance (c)	Log(c)	AVI
0-10 y	<1	Extremely High
10-100 y	1-2	High
100-1000 y	2-3	Moderate
1000-10000 y	3-4	Low
>10000 y	>4	Extremely Low

6.3.5 SINTACS Method

The SINTACS method (Civita, 1994; Civita and De Maio, 2000), partially derived from DRASTIC, retains only the structure of DRASTIC. It evaluates the vertical groundwater vulnerability using the same seven parameters: ***S****oggiacenza* (depth to groundwater), ***I****nfiltrazione* (recharge action), ***N****onsaturo* (attenuation potential of the vadose zone), ***T****ipologia della copertura* (attenuation potential of the soil), ***A****quifero* (hydrogeologic characteristics of the aquifer), ***C****onducibilita* (hydraulic conductivity) and ***S****uperficie topografica* (topographic slope). However, the SINTACS method is more flexible to ratings and weights of the parameters than DRASTIC method.

The SINTACS method can easily be integrated with GIS where each parameter is first computed and mapped over a space in the form of raster map. Thereafter, each mapped parameter is classified into ratings (ranging from 1 to 10), which have an impact on potential pollution. Weight multipliers are then used for each parameter to balance and enhance their importance. Then SINTACS vulnerability index (I_v), defined as weighted sum of the seven parameters, can be computed as (Civita, 1994):

$$I_v = \sum_{i=1}^{7} (P_i \times W_i) \qquad (27)$$

Where, P_i = rating of i^{th} of seven parameters, and W_i = associated weight of i^{th} parameter. The weight classes used by SINTACS depend on the hydrogeological features of each area.

6.3.6 EPIK Method

The EPIK method, through several evaluations, proved to be a suitable parametric weight and point tool to quantify the vulnerability of karstic (carbonate) aquifer zones. Considering the karst aquifer's geological, geomorphological and hydrogeological characteristics, the four parameters influencing flow and transport in karst taken into account by the method are as follows (Doerfliger and Zwahlen, 1995; Doerfliger and Zwahlen, 1998; Doerfliger et al., 1999): **E**pikarst, **P**rotective cover, **I**nfiltration condition and **K**arst network development. Descriptive information about the attribute features for each of the four parameters may be found in Barrocu et al. (2007). The parameters of the EPIK method constitute a protection index, F to be calculated for all parts of the catchments by weighted linear combination technique as follows:

$$Fp_i = a\,E_i + b\,P_i + c\,I_i + d\,K_i \qquad (28)$$

Where, i = 1,..,n is the grid cell number; E_i, P_i, I_i, K_i = weights considered for the i^{th} cell; a, b, c, d = attribute relative weights (constant for any attribute); Fp_i = i^{th} cell protection factor (pertaining to i^{th} cell). The lower the value of protection factor calculated for any i^{th} cell, the higher the vulnerability of the karst aquifer. The step-by-step methodology for applying the EPIK method is shown in Fig. 8.

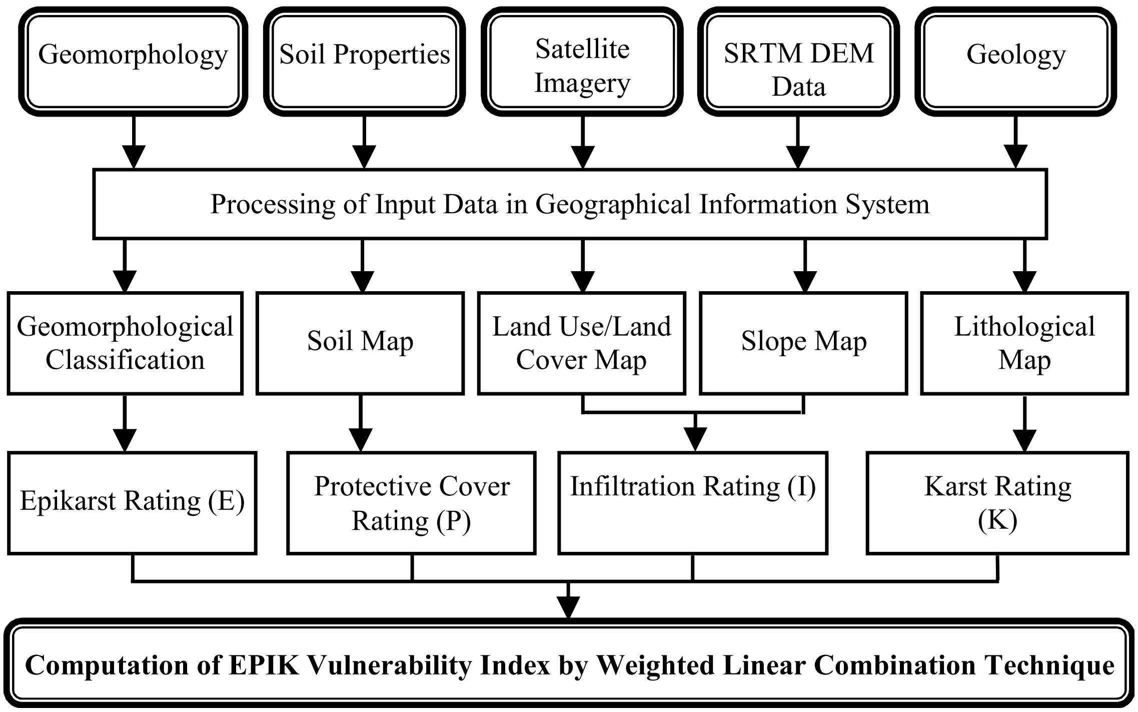

Fig. 8. Flowchart showing step-by-step methodology for applying GIS-based EPIK method.

6.3.7 GLA Method

The GLA (***G****eologisches* ***L****andesamt*) Method, first proposed by Hoelting et al. (1995), is based on a point count system similar to the DRASTIC method. The GLA method was further developed by Goldscheider (2000) into the PI-method within the framework of the European COST 620. Unlike the DRASTIC, the GLA-method only takes the unsaturated zone into consideration. Attenuation processes in the saturated zone are not included in the vulnerability concept. Perhaps, consideration of only unsaturated zone is the major reason that the method could not get wide popularity and applicability. In this method, the degree of vulnerability is specified according to the protective effectiveness of the soil cover and the unsaturated zone. Six parameters considered for the assessment of the overall protective effectiveness are as follows (Hoelting et al., 1995):

Parameter 1: S- effective field capacity of the soil (rating for FCe in mm down to 1 m depth)
Parameter 2: W- percolation rate
Parameter 3: R- rock type
Parameter 4: T- thickness of soil and rock cover above the aquifer

Parameter 5: Q- bonus points for perched aquifer systems
Parameter 6: HP- bonus points for hydraulic pressure conditions (artesian conditions)

The protective effectiveness (PT) is calculated using the following expression (Hoelting et al., 1995):

$$PT = P1 + P2 + Q + HP \tag{29}$$

Where, P1 = protective effectiveness of the soil cover; and P2 = protective effectiveness of the unsaturated zone (sediments or hard rocks)

Parameters P1 and P2 are defined as follows:

$$P1 = S \times W \tag{30}$$

$$P2 = W \times (R1 \times T1 + R2 \times T2 + \ldots\ldots + Rn \times Tn) \tag{31}$$

Based on the German mapping approach, the highest value assigned for factor W, is 1.75 for an annual groundwater recharge of less than 100 mm (Hoelting et al., 1995). A modified scale for the factor W was introduced which reflects the low amounts of groundwater recharge in many areas (Table 8).

Table 8. Modified values of the parameter W (percolation rate)

Groundwater Recharge (mm/year)	Percolation Rate (W)
>400	0.75
300-400	1
200-300	1.25
100-200	1.5
50-100	1.75
25-50	2
≤25	2.25

6.3.8 PI Method

The PI method is used for mapping the intrinsic vulnerability of groundwater resources to pollution through a GIS-based approach (Goldscheider et al., 2000). This vulnerability method is applicable to all kind of aquifers, but provides special methodological tools for the karst aquifers. Conceptually, the method is based on an origin-pathway-target model. The land surface is taken as the origin for the contaminant, the water table in the aquifer is the target which is vulnerable to contamination, and the pathway includes all geologic layers in between. Aquifer vulnerability is assessed as the product of two factors: (i) protective cover (P) and (ii) infiltration conditions (I). The detailed assessment schemes for the two factors can be found in Goldscheider et al. (2000), Goldscheider (2004) and Zwahlen (2004). PI method can be expressed as (Goldscheider et al., 2000):

$$p = P \times I \tag{32}$$

Where, p = protection factor; P = parameter representing protective cover conditions; and I = parameter describing infiltration conditions. Vrba and Zaporozec (1994) proposed five classes of vulnerability (or protectiveness, p) ranging from 1 to 5: value of p = 1 indicates a very low degree of protection and an extreme vulnerability to contamination, whereas a p = 5 indicate a very high degree of protection and a very low vulnerability.

In the PI method, the parameter P describes the protective function of all subsurface layers that may be present between the ground surface and the groundwater table: the topsoil, the subsoil, the non-karst rock and the unsaturated zone of the karst rock. Protectiveness is assessed on the basis of the effective field capacity (FC_e) of the soil, the grain size distribution (GSD) of the subsoil, the lithology, fissuring and karstification of the non-karst and karst rock, the thickness of all strata, the mean annual recharge and artesian pressure in the aquifer (Kouli et al., 2008). The parameter P is classified into five classes according to its value ranging from P=1 (extremely low degree of protection) to P=5 (very thick and protective overlying layers). A decadic (10 point) logarithmic scale is applied to make the parameter P one class higher and to show a ten times higher protectiveness (e.g. 10-m-layer thickness instead of 1 m).

The parameter I, which is very critical for karst aquifers, describes the infiltration conditions. This parameter, in particular, given an idea about the degree to which the protective cover is bypassed due to lateral surface and subsurface flows that enter the karst aquifer at some another place. Values of the parameter vary from 0 (steep slopes with low permeability soil) to 1 (for horizontal and highly permeable soil). On steep slopes of low permeability, surface runoff will be diverted towards a sinking stream while on a horizontal plane of high permeability, diffuse recharge occurs by infiltration and subsequent percolation. In such a case, the protective cover will entirely be bypassed. For rest of the situations, intermediate values (0.2, 0.4, 0.6 and 0.8) of the I parameter are assigned depending on the soil properties controlling the predominant flow process, the vegetation and slope gradient, and the position of a given point inside or outside the catchment of a sinking stream.

In GIS application of the PI method, raster maps of the parameter P and I are to be considered, which may be generated through raster-based spatial analyses performed in GIS. Finally, multiplication of P and I raster maps can be accomplished in GIS and resulted p factor map can be classified into suitable classes to identify high and low vulnerability areas.

6.3.9 COP Method

The COP method of groundwater vulnerability assessment is mainly developed for the carbonate (karst) aquifers. This method provides assessment of intrinsic vulnerability of the aquifers based on three factors: flow **C**oncentration, **O**verlying layers and **P**recipitation. According to European approach (Daly et al., 2002; Goldscheider and Popescu, 2004), the basic concept of this method is to assess the natural groundwater protection (O factor), which is determined by the properties of overlying soils and the unsaturated zone. The method also aimed at estimating how the groundwater protection can be modified by the infiltration process (i.e., diffuse or concentrated) defined by C factor and the climatic conditions (e.g., precipitation) defined by P factor (Kouli et al., 2008). Furthermore, the COP method establishes detailed guidelines, standard tables and formulae for vulnerability assessment and selects suitable variables, parameters and factors to be used according to the European Approach (Daly et al., 2002; Zwahlen, 2004). The method can have wide acceptance in most countries of the world as the geoenvironmental data required by the method is easily available with some fieldwork but no extensive input from GIS is needed. Moreover, the method is applicable in different climatic conditions and different types of carbonate aquifers, e.g. diffuse and conduit flow systems. These flexibilities associated with the COP method make the method more practical and useful for planners and decision makers framing and implementing suitable schemes of groundwater protection. The COP method, comprising of the three factors to evaluate the intrinsic vulnerability of a groundwater resource, is expressed by the following formula (Daly et al., 2002):

$$COP_{index} = C \times O \times P \quad (33)$$

Scores to all three factors are assigned according to their relative impact on the vulnerability of the karst aquifers. The numerical representations of the C, O and P factor values (or scores) are then multiplied to assess the vulnerability. In general, the final values of the COP index indicating the intrinsic vulnerability range from 0 to 15, which can be suitable classified into five vulnerability classes, i.e. very high, high, moderate, low and very low vulnerability (Vrba and Zaporozec, 1994).

The COP method is evaluated as the most effective in comparison to other methods such as DRASTIC, GOD, AVI, SINTACS, EPIK, and PI for assessing the prevailing vulnerability in the southern Spain (Longo et al., 2001; Brechenmacher, 2002; Vıas et al., 2005, 2006; Andreo et al., 2006) based on actual hydrogeological understanding of the aquifers.

7. GIS-BASED WATER QUALITY INDEX

Water Quality Index (WQI) technique is very useful for evaluating the water quality (Abassi, 1999; Adak et al., 2001; Pradhan et al., 2001), especially in resource-poor countries where cost is a major issue for water resources management. In one of the pioneer work, Horton (1965) developed general water quality indices by selecting and weighting several parameters. Although there are no hard and fast rules for constructing a water quality index, a WQI should be specific to a water use or a set of goals (Schultz, 2001). In general, two steps are required for developing a WQI. First, a set of parameters need to be selected that measure the important physical, chemical, and microbiological water characteristics. Of course, the selection of such parameters depends on the intended use of the water. Once information about that set of parameters is available, a rule is needed to summarize all the information in a unique number, i.e., 'water quality index'. The usefulness of water quality indices has been demonstrated in water quality interpretation (e.g., Melloul and Collin, 1998; Soltan, 1999; Stigter et al., 2006; Babiker et al., 2007; Ramesh et al., 2010; Machiwal et al., 2011; Machiwal et al., 2013). Provencher and Lamontagne (1977) proposed one pioneering WQI, which is based on several parameters scored using the same transformations, generally but not always linear, and a final global score is reached. In the past, a variety of water quality indices have been proposed by researchers worldwide (Table 9).

Geographic Information System (GIS) provides an efficient environment for the development of a WQI. In brief, the GIS-based WQI formulation process involves generation of representations for the spatial variability of originally scattered point measurements and the multiple transformations of water quality data into a corresponding index rating value related to water quality. The steps involved in the formulation of GIS-based WQI proposed Babiker et al. (2007) are described in the subsequent section.

Table 9. Different water quality indices developed and used in the earlier studies

S. No.	Name of Index	Country	Parameters Used in Water Quality Index	Source
1	Groundwater Contamination Index	Finland	F, NO_3, UO_2, As, B, Ba, Cd, Cr, Ni, Pb, Rn, Se, pH, $KMnO_4$ consumption, SO_4, Cl, Ag, Al, Cu, Fe, Mn, Na and Zn	Backman et al. (1998)
		Slovakia	TDS, SO_4, Cl, F, NO_3, NH_4, Al, As, Ba, Cd, Cr, Cu, Fe, Hg, Mn, Pb, Sb, Se and Zn	
2	Groundwater Quality Index	Israel	Cl and NO_3	Melloul and Collin (1998)
3	Groundwater Quality Index	Egypt	NO_3, PO_3, Cl, TDS, BOD, Cd, Cr, Ni and Pb	Soltan (1999)
4	Surface Water and Groundwater Quality Index	Croatia	Temperature, mineralization, corrosion coefficient, DO, BOD, total N, protein N, total P and total coliform	Štambuk-Giljanović (1999)
5	Groundwater Quality Index and Groundwater Composition Index	Portugal	NO_3, SO_4, Cl and Ca	Stigter et al. (2006)
6	Surface Water Quality Index	Argentina	Temperature, hardness, DO, pH, EC, alkalinity, turbidity, NO_3, NO_2, NH_3, Cl and SO_4	Vignolo et al. (2006)
7	Malaysian Department of Environment – Surface Water Quality Index	Malaysia	DO, COD, BOD, TSS, NH_3-N and pH	Shuhaimi-Othman et al. (2007)
8	Groundwater Quality Index	Japan	Cl, Ca, Na, Mg, SO_4, TDS, NO_3	Babiker et al. (2007)
9	Surface Water Quality Index	Spain	pH, EC, TSS, NH_3, NO_2, NO_3, COD, BOD, DO, temperature and total P	Sánchez et al. (2007)
10	Fuzzy Surface Water Quality Index	Brazil	Temperature, pH, DO, BOD, Coliforms, dissolved inorganic N, total P, total solids and turbidity	Lermontov et al. (2009)
11	Groundwater Quality Index	India	pH, EC, Na, Cl, SO_4, total alkalinity, total hardness, Ca, Mg, Fe, F, NO_3, NO_2, Mn, Zn, Cd, Cr, Pb, Cu, Ni, total coliform, salmonella	Ramesh et al. (2010)

7.1 Computing Normalized Difference Maps

In the first step, spatial maps (C) representing distribution of concentrations of the water quality parameters over the space are constructed for each parameter from the point sample values by spatial interpolation technique within GIS environment. Thereafter, observed spatial concentrations (C_{obs}) of the water quality parameters are related to their maximum desirable limits (C_{mdl}) prescribed by the WHO (2006) on pixel-by-pixel basis using a GIS-based normalized difference index (ND_{index}) as follows (Babiker et al., 2007):

$$\mathrm{ND}_{\mathrm{index}} = (\mathrm{C}_{\mathrm{obs}} - \mathrm{C}_{\mathrm{mdl}})/(\mathrm{C}_{\mathrm{obs}} + \mathrm{C}_{\mathrm{mdl}}) \tag{34}$$

Values of the resulted ND_{index} for each pixel range between -1 and 1.

7.2 Assigning Rank to Different Water Quality Variables

The ND_{index} maps are rated between 1 and 10 to generate a 'rank map'. The rank 1 indicates minimum impact on water quality, while the rank 10 indicates maximum impact. The minimum ND_{index} value (-1) is set equal to 1, the median value (0) is set equal to 5 and the maximum value (1) is set equal to 10. The following polynomial equation can be used to rank the contamination level (or ND_{index}) of every pixel between 1 and 10:

$$\mathrm{R} = 0.5 \times (\mathrm{ND}_{\mathrm{index}})^2 + 4.5\,(\mathrm{ND}_{\mathrm{index}}) + 5 \tag{35}$$

Where R = rank value of every pixel corresponding to its ND_{index} value.

8.3 Developing Water Quality Index Map

Water Quality Index (WQI) is calculated as follows (Babiker et al., 2007):

$$\mathrm{WQI} = 100 - \left[(\mathrm{R}_1\,\mathrm{w}_1 + \mathrm{R}_2\,\mathrm{w}_2 + \ldots + \mathrm{R}_n\,\mathrm{w}_n)/\mathrm{N}\right] \tag{36}$$

Where R = rate of the rank map (1–10), w = relative weight of the parameter which corresponds to the 'mean' rating value (R) of each rank map (1–10), and N = total number of parameters used in the suitability analysis.

The definition of WQI [Eqn. (35)] is similar to the weighted linear combination technique. The weight (w) assigned to each parameter indicates its relative importance to water quality and corresponds to the mean rating value of its 'rank map'. The total number of parameters (N) involved in the expression of WQI averages and limits the index values between 1 and 100. The '100' in the first part of the formula is incorporated to directly project the WQI value such that high index values close to 100 reflect 'high water quality' and the index values far below 100 (close to 1) indicate 'low water quality'. The entire steps of developing a water quality index map are depicted in a flowchart shown in Fig. 9.

8. GIS-COUPLED MULTIVARIATE STATISTICAL TECHNIQUES

Multivariate statistical analyses techniques such as principal component analysis (PCA) and cluster analysis (CA) are very useful for classifying aquifer groundwater quality according to the different pollution sources. It is observed that the results of the multivariate statistical analyses of water quality data can easily be combined with GIS in order to delineate the different groundwater quality zones. Mapping of groundwater contamination is often complicated by infrequent and uneven distribution of sampling locations, analytical errors in sample analyses, and large spatial variation in observed contaminants over short distances due to complex hydrogeologic conditions. Also, uncertainty may be associated with numerical modelling approach used to delineate groundwater contamination plumes due to inadequate knowledge about local hydrogeological conditions. Furthermore, managing and mapping extensive water quality datasets can be difficult due to the multiple locations, times, and analytes that may be present. An alternative to numerical modeling is to employ statistical analysis of groundwater quality data to infer zones of potential contamination.

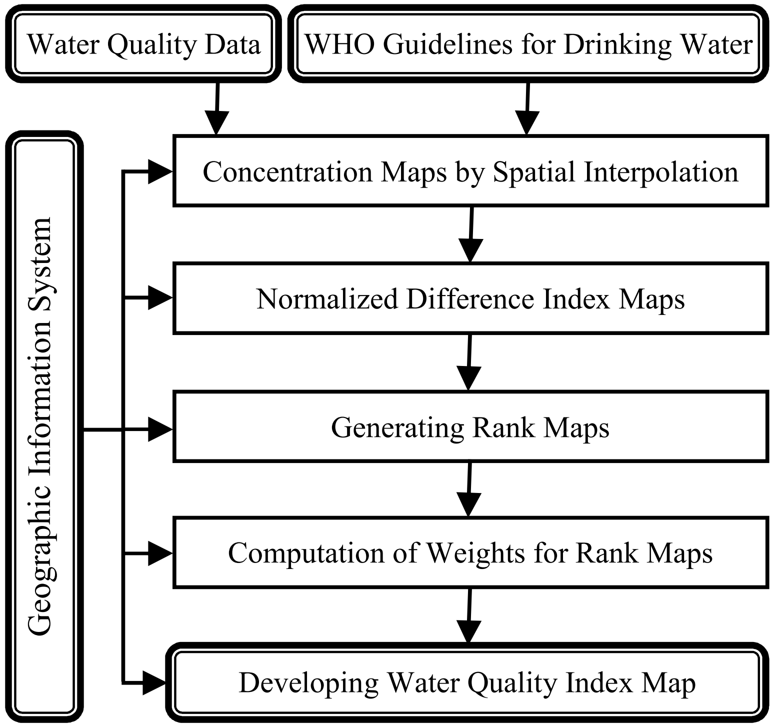

Fig. 9. Flowchart depicting methodology for developing GIS-based water quality index maps.

Principal components analysis (PCA) is a multivariate statistical technique, which classifies/groups the water quality variables based on their correlations with each other. The major aim of applying PCA and CA is to consolidate a large number of observed water quality variables into a smaller number of factors that can be more readily interpreted. Thus, the multivariate statistical techniques reduce dimensionality of the data (Dillon and Goldstein, 1984). The PCA helps identifying underlying geologic and hydrogeologic processes for individual principal components or PCs (or factors) based on the water quality variables grouped under the PCs. The more PCs extracted, the greater is the cumulative amount of variation in the original water quality data. PC loadings show how the PCs characterize strong relationships (positive or negative) between groundwater quality variable and PC describing the variable. In order to determine the number of PCs to be retained, Kaiser Normalization Criterion (Kaiser, 1958) is used. PCs, which best describe the variance of analyzed groundwater quality data (eigenvalue>1) and can be reasonably interpreted (Harman, 1960), are accepted for further analysis. The measure of how well the variance of a particular groundwater quality parameter is described by a particular set of factors is known as ‘communality’ (Jackson, 1991). Number of variables retained in principal components or communalities is obtained by squaring the elements in PC matrix and summing the total within each variable. Ideally, if a PCA is successful, number of PCs will be small, communalities are high (close to 1) and PCs will be readily interpretable in terms of particular sources or process (Dunteman, 1989). The PCA has previously been used to generate accurate maps of monitoring wells grouped by their water quality characteristics (Suk and Lee, 1999; Ceron et al., 2000; Güler et al., 2002). Suk and Lee (1999) performed multivariate statistical analysis in combination with GIS to correlate contaminant data with groundwater quality parameters for the purpose of identifying contaminated aquifer zones.

Cluster analysis (CA) is another multivariate statistical analysis technique that results in data reduction and that can be used to group monitoring sites according to aquifer water quality behaviour (Suk and Lee, 1999). The CA is an unsupervised pattern recognition technique that uncovers intrinsic structure or underlying behaviour of a dataset without making a priori assumption about the data, in order to classify the objects of the system into clusters based on their similarities (Otto, 1998). This method creates linkages between variables hierarchically in the configuration of a tree with different branches. Branches that have linkages closer to each other indicate a stronger relationship among variables or clusters of variables. Mathes and Rasmussen (2006) demonstrated the methodology for generating GIS maps of groundwater contamination using multivariate statistical analysis of water quality data. GIS is an important tool that is used to organize and manage large amounts of water quality information for use in decision support systems. Now-a-days, GIS has been started to be used routinely for displaying water quality data in map form but still use of statistical indicators of contaminant distributions are rarely seen. Presently, focus of GIS-coupled multivariate statistical analysis techniques has shifted from mapping of observed contaminant distribution to developing a map of contamination potential created using auxiliary water quality data.

Prior to applying multivariate statistical analysis, generally the observed water quality data, x_{ji} are standardized by z-scale transformation as given below:

$$z = \frac{x_{ji} - \overline{x}_j}{s_j} \tag{37}$$

Where, x_{ji} = value of the j^{th} water quality parameter measured at i^{th} site, $\overline{x}_j$ = mean (spatial) value of the j^{th} parameter, and s_j = standard deviation of the j^{th} parameter.

The analysis performed with standardized data is expected to be less influenced by small/large variance of the data. Furthermore, standardization of the data removes the influence of different measurement units of the data by making the data dimensionless.

9. CONCLUSION

Evaluation of water quality is necessary for managing water quality so as to ensure environmental sustainability. The important aspects of water quality evaluation are interpretation of water quality variables, reporting of results and recommendations for planners and decision makers. Logical sequence of any water quality evaluation programme consists of three key steps: *monitoring*, *evaluation* and *management*. There are various tools and techniques available for water quality interpretation. However, selection of appropriate tools is very crucial for making the water quality evaluation to be effective. Among the several tools used for water quality assessment, geographic information system (GIS) has been gaining a wide acceptance for past two decades among the researchers worldwide due to the capabilities of GIS such as handling, capturing, storing, analyzing and displaying large quantum of water quality data.

It is clear that with the advent of GIS technique, many conventional methods of water quality evaluation have been interfaced with GIS to enhance usefulness of the methods. The conventional methods coupled with GIS can be applied over relatively large areas. GIS-based spatial statistical analyses make it possible to explore spatial and temporal variations of the water quality. The point data of the water quality may accurately be converted to vector and raster formats through integration of geostatistical and GIS modeling techniques. Raster formats of the water quality data further enable spatial analyses to be performed under GIS platform for groundwater vulnerability mapping. Among the various overlay and index methods for mapping groundwater vulnerability, DRASTIC, GOD, AVI, and SINTACS are mostly applied in many places of the world. Later on, few specific vulnerability methods applicable to karst (carbonate) aquifers were developed, e.g. EPIK, GLA, PI and COP. Computation of water quality index is another way of evaluating the water quality where GIS technique plays a central role. With the aid of GIS, it is possible to locate areas having poor quality of the groundwater. The definition of WQI is flexible and many researchers have developed different types of water quality indices depending upon the data availability, aim of assessment, geologic condition, aquifer type, etc. Recently, GIS has also been associated with multivariate statistical analysis techniques, e.g., principal component analysis (PCA) and cluster analysis, etc.

Finally, GIS is considered as a modern powerful tool having large flexibility to be combined with conventional methods of water quality evaluation. This new era tool has great potential to play a key role in water quality evaluation to ensure sustainable management of natural resources. In future, new areas for the assessment of water quality are to be explored wherein the application of GIS technique will further strengthen the interpretation of water quality analyses ultimately leading towards more sustainable planning and utilization of the water resources.

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